Office of Radiation & Indoor Air
Radiation Protection Division (6608J)
Washington, DC 20460

EPA 402-R-05-007
August 2007

Technologically Enhanced Naturally Occurring Radioactive Materials From Uranium Mining

Volume 2: Investigation of Potential Health, Geographic, and Environmental Issues of Abandoned Uranium Mines

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Introduction

Uranium is a common element in nature, and has been used for centuries as a coloring agent in decorative glass and ceramics. Today, uranium has uses that range from metal alloys to aircraft counterweights. The most significant modern uses of uranium, however, have been for national defense and electric power generation. The advent of nuclear weapons and nuclear power in the United States resulted in a full-blown exploration and mining boom, starting immediately after World War II and making uranium the most important commodity in the mining industry. The greatest period of uranium production spanned from approximately 1948 to the early 1980s (U.S. DOE/EIA 1992). Through 2005, the industry had generated over 420,000 metric tons (MTs) of uranium to foster U.S. dominance in nuclear weapons technology, and later to feed the growing number of commercial power plants utilizing the enormous energy contained in the uranium nucleus (U.S. DOE/EIA 2003a, 2003b, 2006).

Another legacy of uranium exploration, mining, and ore processing was the creation of unreclaimed land workings wherever the uranium concentration in rock was either found or thought to be economically viable. Thousands of miners and prospectors, as well as large mining companies, searched the United States in search of veins, lenses, sedimentary deposits, and breccia pipes concentrating the valuable metal, echoing the California gold rush 100 years earlier. In many instances, they left behind unreclaimed and exposed wastes elevated in naturally occurring radioactive materials (uranium and its radioactive decay progeny), exposing people and the environment to its hazards.

In this report, Naturally Occurring Radioactive Material (NORM) is defined as: **Materials which may contain any of the primordial radionuclides or radioactive elements as they occur in nature, such as radium, uranium, thorium, potassium, and their radioactive decay products, that are undisturbed as a result of human activities.** Radiation levels presented by NORM are generally referred to as a component of "natural background radiation."

The term Technologically Enhanced Naturally Occurring Radioactive Material (TENORM) is defined as: **Naturally occurring radioactive materials that have been concentrated or exposed to the accessible environment as a result of human activities such as manufacturing, mineral extraction, or water processing. "**Technologically Enhanced" means that the radiological, physical, and chemical properties of the radioactive material have been altered by having been processed (beneficiated) or disturbed in a way that increases the potential for human and/or environmental exposures. This definition differs somewhat from other definitions provided by the National Academy of Sciences (NAS 1999a) and the Conference of Radiation Control Protection Directors (CRCPD 2004) in that it further amplifies the need to include materials which have not been modified by human activities, yet have been disturbed in

such ways that they can be misused by humans, or affect the environment[1]; it does not include a reference to Atomic Energy Act materials, as the definitions are changing (see Volume I and its Appendix VI). Uranium TENORM includes the succession of radioactive decay progeny of the parent uranium.

Under the Atomic Energy Act, the U.S. Nuclear Regulatory Commission (NRC) regulates operations that produce and concentrate uranium and thorium. In accordance with terminology of the Act, the NRC has defined in 10 CFR 40.4 "source materials" **as (1) uranium or thorium, or any combination thereof, in any physical or chemical form, or (2) ores which contain by weight one-twentieth of one percent (0.05%) or more of: (i) uranium, (ii) thorium or (iii) any combination thereof. Source material does not include special nuclear material.** It also defines the "by-product materials" (wastes) of those operations **as tailings or wastes produced by the extraction or concentration of uranium or thorium from any ore processed primarily for its source material content, including discrete surface wastes resulting from uranium solution extraction processes.** Byproduct materials are also regulated by the NRC. Underground ore bodies depleted by such solution extraction operations do not constitute "byproduct material" within this definition. Wastes from conventional uranium mining (both surface and underground) are not subject to NRC regulation, but are considered to be TENORM, and thus subject to U.S. Environmental Protection Agency (EPA) and State agency oversight.

Under the Energy Policy Act of 2005, the Atomic Energy Act was amended to place additional discrete (highly radioactive in small, defined volumes) sources of TENORM which had the potential to pose a threat to public health and safety or the common defense and security under NRC jurisdiction. The definition of byproduct materials was further modified to include discrete sources of radium-226, any material made radioactive by use of a particle accelerator for use in a commercial, medical or research activity, or materials which might pose a similar threat to public health and safety or the common defense and security. Specific requirements were provided for determining the appropriate waste disposal methods for these materials. The NRC regulatory definitions of byproduct materials to accommodate these amendments are expected to be finalized in the summer of 2007, to reflect the recent amendments as of this writing. These products and wastes are not the subject of this report.

The U.S. Environmental Protection Agency (EPA) has previously issued reports on the uranium mining industry in response to congressional mandates and programmatic needs. In 1983, EPA

[1] The National Academy of Sciences (NAS 1999a) defined TENORM as "…any naturally occurring radioactive materials not subject to regulation under the Atomic Energy Act whose radionuclide concentrations or potential for human exposure have been increased above levels encountered in the natural state by human activities." The International Atomic Energy Agency (2003), although referring to this class of wastes and products as "NORMs", defined them as encompassing "all naturally occurring radioactive materials where human activities have increased the potential for exposure in comparison with the unaltered situation. Concentrations of radionuclides (i.e. TE-NORM) may or may not have been increased." Alternatively, the Conference of Radiation Control Program Directors (CRCPD 2004) has defined them as a naturally occurring radioactive material whose radionuclide concentrations are increased by or as a result of past or present human practices. TENORM does not include background radiation or the natural radioactivity of rocks or soils. TENORM does not include "source material" or "byproduct material" as both are defined in the Atomic Energy Act of 1954, as amended (AEA 42 USC §2011 *et seq.*) and relevant regulations implemented by the NRC. EPA believes the definition should include materials which were disturbed, but not further concentrated by human activities, so that the full scope of hazards from TENORM materials can be considered.

published its *Report to Congress on the Potential Health and Environmental Hazards of Uranium Mine Wastes* (U.S. EPA 1983a, b, c), as required by the Uranium Mill Tailings Radiation Control Act of 1978. This study provided an important overview of the characteristics and generation of uranium mining TENORM wastes during a period when the uranium mining industry was still near its production peak. A subsequent 1985 *Report to Congress on Wastes from the Extraction and Beneficiation of Metallic Ores, Phosphate Rock, Asbestos, Overburden from Uranium Mining, and Oil Shale* (U.S. EPA 1985), carried out pursuant to requirements of the Resource Conservation and Recovery Act of 1976 (RCRA), as amended, provided additional risk information and characterization of uranium mining waste. In 1995, EPA issued the Technical Resource Document *Extraction and Beneficiation of Ores and Minerals: Uranium* as a technical update to provide a means of evaluating wastes that were exempt from or subject to regulation under RCRA (U.S. EPA 1995).

During the period 1989 to 1993, EPA worked on a draft scoping report (SC&A 1993) which compiled information on TENORM in several industries, including uranium mining. A preliminary risk assessment was also developed for certain public and occupational exposure scenarios involving the known radiation levels in those industries. Comments received on the draft from industry, as well as EPA's Science Advisory Board (SAB) (U.S. EPA 1994), resulted in further revisions of the scoping draft, though it was ultimately decided that a final report would not be issued.

Following a review of EPA's guidance for TENORM by the National Academy of Sciences, EPA's response to the NAS study, and discussions with EPA's Science Advisory Board, EPA's Radiation Protection Division decided that a further review of the current hazards associated with uranium mining TENORM was warranted. The SAB (U.S. EPA 2001a) agreed with EPA's intent to make TENORM documents useful to a broad audience, but also recommended that the whole life cycle of a TENORM source—in this case uranium extraction—be considered beyond regulatory or inter-agency considerations, and that the impacts of non-radiological contaminants also be examined in the Agency's technical reports. In addition to most sources of TENORM, EPA is responsible for setting environmental standards under the Uranium Mill Tailings Radiation Control Act, cleaning up hazardous waste sites that include some former uranium mines, and assisting Native Americans, including assisting in environmental reviews of proposed *in situ* leach (ISL) facilities. While this report focuses on the impacts associated with conventional surface and underground uranium mines, it provides limited background materials, in appendicies, on risks associated with uranium milling and ISL operations and wastes generated by those processes, even though they may not be considered TENORM by virtue of their regulation by the NRC and its Agreement States under the Atomic Energy Act and its amendments.

This is the second of two reports on uranium mining TENORM. The first report, *Technologically Enhanced Naturally Occurring Radioactive Materials from Uranium Mining, Volume 1: Mining and Reclamation Background* (U.S. EPA 2006a), provides background information on the occurrence of uranium, mining techniques, and reclamation of uranium mines. This report investigates the potential radiogenic cancer risks from abandoned uranium mines and evaluates which may pose the greatest hazards to members of the public and to the environment. The intent of this report is to identify who may be most likely to be exposed to wastes at small abandoned uranium mines, and where the greatest risks may lie. The specific

wastes of EPA concern from this report and study are from abandoned conventional open-pit and underground uranium mines, and include overburden, unreclaimed sub-economic ores (protore), waste rock, core hole and drill cuttings, and mine and pit (or pit lake) water. All are described in Volume I of this study. In addition, EPA has compiled and published a uranium location database (U.S. EPA 2006b).

A first draft of this report underwent an outside peer review following the Agency's peer review process. Using the comments obtained, the report has been updated and revised. Appendices have been added to this version of the report providing references and information on the risks associated with uranium mill operations and ISL operations. While some of the thousands of conventional open surface and underground uranium mines in the United States have been reclaimed, many have not. Any mine may pose such hazards as open shafts and unstable supports (rock and wood), and contain gases, such as carbon dioxide and methane, that displace oxygen and could lead to asphyxiation. In addition to the immediate physical threats that abandoned mines may pose, exposure to radiation from uranium and radium and other contaminants in abandoned mine waste can increase a person's risk of cancer.

People are exposed to naturally occurring radioactive materials in soils, as well as natural occurrences of uranium in rock outcrops. However, the primary focus in this report is on exposures to those naturally occurring radioactive materials that have been enhanced by human activities. In examining the radiological risks due to mining, the focus is on those concentrations above natural background, as recommended in the EPA *Abandoned Mine Site Characterization and Cleanup Handbook* (U.S. EPA 2000a), with emphasis on uranium and radium. Abandoned conventional uranium mines may also contain other hazardous contaminants, such as metals. For example, the carcinogen arsenic may be a problem at some uranium mines, contributing to increased risks.

This scoping report describes in Chapter 1 several previous studies supporting the risk analysis, while Chapter 2 provides a geographic location analysis of uranium mines in the western United States. Chapter 3 discusses potential scenarios and exposure pathways for the general public to hazards from uranium mines, describes the methodologies used in the analysis, and assesses cancer risks posed by human exposure to the various hazards from the mines. Chapter 4 examines the use of uranium risks in building materials, and Chapter 5 briefly discusses the potential for ecological impacts from the mines. Uncertainties and conclusions are presented in Chapters 6 and 7.

1.0 MAJOR STUDIES SUPPORTING THIS SCOPING RISK ANALYSIS

The most important period of past U.S. uranium production spanned from approximately 1948 to the early 1980s (U.S. DOE/EIA 1992). Through 2005 the industry had generated over 420,000 metric tons (MTs) of uranium for nuclear weapons and commercial power plants (U.S. DOE/EIA 2003a, 2003b, 2006). Uranium exploration, mining, and ore processing left a legacy of unreclaimed land workings wherever the uranium concentration in rock was either found or thought to be economically viable. This report investigates some potential health, geographic, and environmental issues of abandoned uranium mines.

The major studies supporting this scoping analysis include EPA's 1983 *Report to Congress on the Potential Health and Environmental Hazards of Uranium Mine Wastes* (U.S. EPA 1983a, b, c) and EPA's risk assessments for underground and surface uranium mines for Clean Air Act requirements (U.S. EPA 1989a). Other analyses considered include a report of two uranium mines on the Superfund National Priorities List (U.S. EPA 2001b) and a U.S. Department of Energy report (U.S. DOE/EIA 2000). These studies are discussed in this chapter.

1.1 1983 EPA Report to Congress

The Uranium Mill Tailings Radiation Control Act of 1978 directed EPA to conduct a study on "the location and potential health, safety and environmental hazards of uranium mine wastes," and to provide "recommendations, if any, for a program to eliminate these hazards." When EPA published its 1983 Report to Congress (U.S. EPA 1983a, b, c) (hereafter referred as the 1983 EPA report or study), there were about 340 active uranium mines in the United States. At the end of 2002, there were no active conventional uranium mining operations in the United States, and only two active operations using the *in situ* leaching process (U.S. DOE/EIA 2003a). However, with an increase in the price of uranium since 2004, additional conventional mines have begun production or will be coming on line in the near future, and some suspended mine operations have recommenced. As part of the 1983 study, EPA also made observations at a number of active and inactive uranium mine sites, collected soil and water samples, and took some external gamma and radon flux measurements at sites in Colorado, New Mexico, Texas, and Wyoming.

1.1.1 Sources and Pathways Modeled

In the 1983 report, EPA used the information discussed above to develop models for large and small mines, including an inactive surface mine hypothetically located in Wyoming and an inactive underground mine hypothetically located in New Mexico (U.S. EPA 1983b). From these model mines, which were classified as an average mine or a large mine, EPA estimated the health effects to populations within 50 miles (80 km) of each mine and on a hypothetical most exposed individual living about 1 mile from the center of a mine. The pathways considered were as follows:

- Breathing air containing windblown dust and radon decay products
- Drinking water containing uranium and its decay products

- Eating food contaminated by either air or water
- Living in homes on land covered by mine wastes (U.S. EPA 1983b)

With the exception of the last pathway, the focus of the report was on estimating risks to people who were off site. The home pathway was not explicitly modeled like the other pathways, but used estimates of indoor radon as a function of radium in the soil. While the 1983 report produced many analyses, some issues were not explicitly addressed, including the following:

- Drinking groundwater and surface water near a mine. This pathway was considered and included for the regional population, but was not included for the most exposed individual due to lack of information on radionuclides in potable water.
- Individuals spending time on mine sites.
- Using mine waste material for buildings.

In its 1983 Report to Congress, EPA identified the sources modeled and those considered, but not modeled, due to a lack of information (Table 1.1). For groundwater, the report noted that uranium mines may pose a problem, but the authors did not have enough information to consider it. The report also noted that spending time at the mine sites and using waste materials in the buildings would be a health hazard, but did not quantitatively address the issues.

Table 1-1. Sources of Contamination at Uranium Mines

In its 1983 Report to Congress, EPA identified the sources modeled (M) and those considered (C), but not modeled, due to a lack of information.

Sources of Contamination	Underground Mines		Surface Mines	
	Active	Inactive	Active	Inactive
Waste Rock (Overburden) Pile				
Wind-suspended dust	M	M	M	M
Radon-222 emanation	M	M	M	M
Precipitation runoff	C	C	C	C
Sub-Ore Pile				
Wind-suspended dust	M	M	M	M
Radon-222 emanation	M	M	M	M
Precipitation runoff	C	C	C	C
Ore Stockpile				
Wind-suspended dust	M	M	M	M
Radon-222 emanation	M	M	M	M
Precipitation runoff	C	C	C	C
Abandoned Mine Area Surfaces				
Radon-222 emanation	M	M	M	M
Mining Activities				
Dusts	M	NA	M	NA
Combustion products	M	NA	M	NA
Radon-222	M	NA	M	NA
Wastewater				
Surface discharge	M	NA	M	NA
Seepage	C	C	C	C

Note: NA = not applicable.
Source: USEPA 1983b, Table 2.

1.1.2 1983 EPA Study Findings

Using the risk methodology of the time (AIRDOS-EPA, DARTAB, and RADRISK), the study estimated that a large active underground mine posed an increased chance of a fatal lung cancer to an individual of 2×10^{-3}, primarily from breathing radon decay products, and that risks from other types of uranium mines were somewhat lower. Releases to surface water from an average underground mine one mile from an individual's home were estimated to increase his or her lifetime cancer risk by 1×10^{-3}, and that one additional cancer in several hundred years might occur in nearby populations from the normal operational releases from a mine. Although the study did not address the health effects of contaminated shallow aquifers around active or inactive mines, it recommended that they be evaluated.

For inactive mines, the study noted that radionuclide airborne emissions were smaller than for active mines, with the risks coming from radon emanating from unsealed mine vents, portals, and residual waste piles. The estimates of risks from radon emissions from inactive uranium mines were as follows:

- Individuals living for a lifetime 1 mile (1.6 km) from an inactive mine would have an increased chance of lung cancer of about $2–3 \times 10^{-5}$.
- The amount of radon-222 released each year from all inactive uranium mine sites would (cumulatively) cause about 0.1 lung cancer~~s~~ fatalities in the lifetime of the regional population living within 50 miles (80 km) of these sites.

The study found insignificant concentrations of hazardous air emissions at inactive sites and thus concluded that their health impacts would be insignificant as well. Although the study acknowledged the potential for hazards from buildings that use uranium mine wastes as construction material, it did not formally analyze the hazard. However, it did mention that building on contaminated land could increase indoor radon concentration and, thus, increase the risk of lung cancer in the residents (U.S. EPA 1983b). The study referenced an earlier study (out of print) jointly conducted by EPA and the Atomic Energy Commission in 1972, that identified about 500 buildings in several western states that exhibited anomalous gamma radiation readings that appeared to be associated with uranium mine wastes. This is further discussed in Chapter 4 of this volume. Tables 1.2 and 1.3 present the specific lifetime cancer risk estimates due to radioactive airborne emissions for one year of exposure and over a lifetime of exposure.

Table 1-2. Estimated Lifetime Fatal Cancer Risks from 1 Year of Exposure to Airborne Uranium Mine Emissions

The cancer risk from inactive uranium mine radon emissions are generally low for 1 year of exposure.

Source of Exposure	Risk to Maximum Exposed Individual[a]	Risk to Average Exposed Individual[b]	Collective Risk to Regional Population
Inactive surface mines—total	4.7×10^{-7}	8.9×10^{-10}	1.3×10^{-5}
Particulates and Radon-222	5.5×10^{-8}	6.4×10^{-11}	9.1×10^{-7}
Radon-222 daughters	4.2×10^{-7}	8.3×10^{-10}	1.2×10^{-5}
Inactive underground mines—total	2.8×10^{-7}	1.2×10^{-9}	4.5×10^{-5}
Particulates and Radon-222	1.5×10^{-8}	2.0×10^{-11}	7.4×10^{-7}
Radon-222 daughters	2.7×10^{-7}	1.2×10^{-9}	4.4×10^{-5}

a An individual living within 1 mile (1.6 km) downwind from the mine.
b The average individual in the regional population within a 50-mile (80-km) radius of the model mine.
Source: U.S. EPA 1983b, Table 6.11.

Table 1-3. Estimated Lifetime Fatal Cancer Risks from Lifetime Exposure to Airborne Uranium Mine Emissions

The risk to the average person from uranium mine emissions is low. While the risk to the maximally exposed individual is significantly larger, it is still within the Superfund 10^{-4} – 10^{-6} risk range.

Source of Exposure	Maximum Exposed Individual[a]	Average Exposed Individual[b]
Inactive surface mines—total	3.4×10^{-5}	6.3×10^{-8}
Particulates and Radon-222	3.9×10^{-6}	4.5×10^{-9}
Radon-222 daughters	3.0×10^{-5}	5.9×10^{-8}
Inactive underground mines—total	2.0×10^{-5}	8.6×10^{-8}
Particulates and Radon-222	1.1×10^{-6}	1.4×10^{-9}
Radon-222 daughters	1.9×10^{-5}	8.5×10^{-8}

a An individual living 1 mile (1.6 km) downwind from the mine.
b The average individual in the regional population within a 50-mile (80-km) radius of the model mine.
Source: U.S. EPA 1983b, Table 6.12.

1.1.3 Applicability of 1983 Risk Estimates

According to Table 6.17 of the 1983 EPA report (U.S. EPA 1983b), radon decay products account for 88 percent or more of the fatal cancer risk due to emissions of radioactive particles from inactive surface and underground mines. Risk estimates given for radon decay product releases from these two types of mines in Tables 6.11 and 6.12 of the report are consistent with the methodology used by EPA prior to 1988. At that time, 4.6×10^{-4} cancers were projected per

working-level month (WLM)[1] of exposure. An analysis of results from the recent BEIR VI report (NAS 1999b, U.S. EPA 2003a) on risks from exposure to radon suggests that the risk factor should be 5.38×10^{-4} per WLM.

Table 1.4 reproduces the working-level estimates of the model inactive surface mines and model inactive underground mines which are provided in Table 6.3 of the 1983 EPA report. The values in Table 1.5 are based on the working-level estimates in Table 1.4. Table 1.5 presents recalculated risks from 1-year, 30-year, and lifetime exposures to radon decay product emissions using the higher, current risk factor. The table does not account for exposures for the portion of time spent outdoors, and for lifetime exposures it assumes an average life span of 75.4 years, which is slightly longer than the 71-year life span used in the 1983 EPA report. The formulas used to derive the results in Table 1.5 are as follows:

$$R_1 = \text{Lifetime risk for 1-year exposure at 1 WL} = 51.56 \text{ WLM/WL-y} \times 5.38 \times 10^{-4} \text{ WLM}^{-1} \times 1 \text{ year};$$

- The risk for 30-year exposure at 1 WL = $R_1 \times 30$ years= 0.83
- The risk for lifetime exposure at 1 WL = $R_1 \times 75.4$ years = 2.09

Thus, the 1-year exposure risk estimate from radon decay products for the maximally exposed individual at an inactive surface mine using the 1.8×10^{-5} WL estimate from the model mine in Table 1.4 would be:

$$R_1 = 1.8 \times 10^{-5} \text{ WL} * 51.56 \text{ WLM/WL-y} \times 5.38 \times 10^{-4} \text{ WLM}^{-1} \times 1 \text{ year} = 4.99 * 10^{-7} \cong 5.0 \times 10^{-7}$$

Risks using this updated estimate and presented in Table 1.5 are about 17 percent higher than in the 1983 report, reflecting the increased risk per working level. One limitation relating to this conclusion is that no adjustment was made in the calculations for differences in the distribution of activity-weighted particle size for indoor and outdoor radon exposures.

Table 1-4. Annual Exposure from Radon Decay Product Emissions from Model Uranium Mines

Source of Exposure	Average Radon Daughter Concentration (Working Levels)*	
	Maximum Exposed Individual [a]	Average Exposed Individual [b]
Inactive surface mine	1.8×10^{-5}	3.5×10^{-8}
Inactive underground mine	1.1×10^{-5}	5.1×10^{-8}

* A Working Level is defined in footnote 1 of this chapter.
a An individual living 1 mile (1.6 km) downwind from the mine.
b The average individual in the regional population within a 50-mile (80-km) radius of the model mine.
Source: U.S. EPA 1983b, Table 6.3.

[1] The working level (WL) is defined as any combination of short-lived radon decay products (through polonium 214) per liter of air that will result in the potential emission of 1.3×10^5 MeV of alpha energy. A person exposed to one WL for 170 hours is said to have acquired an exposure of one working-level month (WLM) (Shapiro 1990). This 170-hour value is based on the typical number of hours underground miners worked in 1 month.

Table 1-5. Estimated Individual Lifetime Fatal Cancer Risks for Various Exposures to Radon Decay Products

With the modification for the current risk methodology, the lifetime fatal cancer risk from radon decay products is still within or below the Superfund 10^{-4} – 10^{-6} risk range. (See the discussion for additional background of the risk estimates.)

Source of Exposure	Exposure Duration	Lifetime Risk of Fatal Cancer	
		Maximum Exposed Individual [a]	Average Exposed Individual [b]
Inactive surface mine	1 year	5.0×10^{-7}	9.7×10^{-10}
	30 years	1.5×10^{-5}	2.9×10^{-8}
	75.4 years (lifetime)	3.8×10^{-5}	7.3×10^{-8}
Inactive underground mine	1 year	3.1×10^{-7}	1.4×10^{-9}
	30 years	9.2×10^{-5}	4.3×10^{-8}
	75.4 years (lifetime)	2.3×10^{-5}	1.1×10^{-7}

a An individual living 1 mile (1.6 km) downwind from the mine.
b The average individual in the regional population within a 50-mile (80-km) radius of the model mine.
Source: U.S. EPA 1983b, Table 6.17.

1.2 1989 EPA Study in Support of NESHAPs

In 1989, EPA conducted risk assessments for active underground uranium mines and surface uranium mines (U.S. EPA 1989a), in support of the National Emission Standards for Hazardous Air Pollutants (NESHAPs) for Radionuclides (U.S. EPA 1989b, c). While some of the information in this investigation was based upon U.S. EPA 1983 (a, b, c), the study also included some new field work and analysis. The study found that of all the radionuclides emitted, radon decay products posed the greatest cancer risk. The maximum exposures from underground mines would create lifetime individual fatal cancer risks of greater than 1×10^{-4}, with a maximum of 4×10^{-3}. The maximum individual risk of fatal cancer from radon decay products at surface uranium mines was estimated to be 5×10^{-5}; this risk estimate, too, would be slightly higher, given the current methodology. The 1989 study found that only a limited number of people lived within several hundred feet of the mines and would have been exposed to the maximum levels; most of the nearest residents lived several miles from the mines.

1.3 Uranium Mines on the National Priorities List

Although several uranium mill tailings sites are on the Superfund National Priorities List (NPL), only two uranium mines are on the list: Midnite Mine, near Wellpinit, Washington, and the Fremont National Forest—White King/Lucky Lass Mines, Oregon. Both sites have progressed far enough in the Superfund process to have had a cleanup remedy selected in a Record of Decision (U.S. EPA 2001b, U.S. EPA 2006c). Figures 1.1 and 1.2 are aerial images of Midnite Mine and the White King/Lucky Lass Mine sites, respectively.

Figure 1-1. Aerial Image of Midnite Mine, Washington State

Midnite Mine is on the NPL. The site has uranium and other heavy metal contamination in the disturbed area and two pit lakes.

Source: Photo courtesy of EPA Region 10 Superfund Program.

No one is currently living at the White King/Lucky Lass site, nor is a future resident anticipated, given that the site is on U.S. Forest Service property and is not near population centers. However, the risk assessment did assume a future resident as a scenario. In addition, the receptors evaluated included a site worker (e.g., timber or U.S. Forest Service employees) and a recreational user. The following areas were used as exposure points (U.S. EPA 2001b):

- The protore stockpile at the White King Mine
- The overburden stockpile at the White King and Lucky Lass mines
- Off-pile areas at the White King and Lucky Lass mines

The primary chemicals of concern at the White King/Lucky Lass site were arsenic in soil and shallow groundwater, uranium-234/238 in stockpile groundwaters, radium-226/228 in soil and shallow bedrock wells, and radon in water. Of note, and in spite of several high radon flux rates, inhalation of radon in ambient air was not an issue, since radon concentrations from the stockpiles were equivalent to background concentrations.

Figure 1-2. Aerial Image of White King and Lucky Lass Mines, Oregon

The Lucky King Mine pit lake is approximately 5,000 feet (1,500 meters) northwest (left) of the White King Mine pit lake.

Table 1.6 summarizes the risks at the mine sites for the human receptors. With the approach used in the Record of Decision, the exposure assessment indicated an extremely high risk to future potential residents and child recreational users. The high risks were primarily due to ingestion of arsenic in soils and shallow groundwater and external radiation from radium. In the ecological assessment, no adverse effects were seen from the radionuclides. However, some potential adverse ecological effects were identified due to arsenic, selenium, antimony, lead, and mercury in surface and subsurface soils at the White King Mine. At Lucky Lass, only slightly elevated risks (the noncarcinogen chemical hazard index ranging from 1 to 3) were predicted for the vagrant shrew and terrestrial plants exposed to arsenic and silver in surface soil. In contrast, Midnite Mine has a greater potential for future use, but the cancer risks were predicted to equal 8×10^{-1} for a resident of the affected area and 2×10^{-3} for recreational visitors.

Table 1-6. Potential Cancer Risks from the White King/Lucky Lass and Midnite Mine Sites

The combination of arsenic and radium produces very high cancer risks to potential on-site residents.

Receptor	Total Cancer Risk	Pathway	Notes
White King Mine current adult worker	6×10^{-5}	Ingestion of arsenic in soil and exposure to external radiation from radium-226/228 in the top 6 inches of soil.	Current exposure estimates for soil are based on 0–6 inches; future exposure estimates for soil are based on 0 - 6 feet.
Future recreational user (child) at the White King Mine	4×10^{-4}	Arsenic in soil, exposure to external radiation from radium-226/228 in soil and ingestion of arsenic in Augur Creek and White King groundwater. Ingestion of arsenic in soil and exposure to external radiation from radium-226/228 in the top 6 feet of soil, ingestion of arsenic in shallow bedrock groundwater, inhalation of radon in shallow bedrock groundwater, and exposure to arsenic in White King pond surface water and sediment.	Deep bedrock water contains high levels of naturally occurring arsenic, radon, and minerals that would preclude its use as drinking water.
Potential future resident (adult) at the White King Mine	3×10^{-1}		
Potential future resident (child) at the White King Mine	2×10^{-1} Hazard Index values for noncarcinogenic effects to current and potential future child recreational users were 4 and 11, respectively, and higher for potential future residents from ingestion of arsenic and manganese in shallow bedrock groundwater and ingestion of arsenic in soil.		
Potential future resident at the Lucky Lass Mine	1×10^{-3}		
Potential future resident at the Midnite Mine Area	1		

Note: A Hazard Index value below 1 indicates no adverse health effects are expected as a result of exposure.
Source: U.S. EPA 2001b.

1.4 DOE Report on Costs of U.S. Uranium Mine Environmental Restoration

A report commissioned by the U.S. Department of Energy (DOE) found that a number of uranium mines are undergoing or have completed remediation (U.S. DOE/EIA 2000). According to the report, 21 mines, primarily in Wyoming and Texas, were selected for analysis for one or more of the following reasons: (1) substantial output of uranium concentrates, (2) major impact on the environment, and (3) significant costs required for remediation. While the report does not specify whether these sites are undergoing risk assessments, it does specify whether a particular site has an exposure pathway of surface water, groundwater, or windblown particulates. The information lists groundwater as an exposure pathway for many of the mines, while the surface water and windblown particulate pathways are not as prevalent.

2.0 GEOGRAPHIC ANALYSIS ON THE LOCATION OF URANIUM MINES

With the exception of some phosphate mine areas in central and northern Florida, people are most likely to be exposed to uranium mining-related TENORM in the western United States. This chapter provides a geographic analysis of the spatial locations of western mines in proximity to human populations, cultural and political features and boundaries, and environmental features. The use of geographical information system (GIS) software provides a systematic means to understand the potential impacts and scenarios by which humans and the environment may be impacted by uranium mines.

Figure 2.1 was generated from uranium mining-related records from the U.S. Bureau of Mines—now U.S. Geological Survey (USGS)—Mineral Availability System/Mineral Industry Location System (MAS/MILS) database from the EPA BASINS data (U.S. EPA 2001c). While about half of the 8,234 locations are documented as producing ore, the remaining records may identify mines or simply locations with uranium. Of the 8,234 records, 4,141 are categorized as "producer" or "past producer," and these terms are being used as proxies for known mines. Another 63 records are classified as mills or processing plants, and once these are removed, the 4,078 records that are left are assumed to be former mines. Of the 4,078 mines, about 3,000 are in Colorado, Utah, Arizona, and New Mexico. Similar information comes from the Department of Energy's (DOE) Energy Information Administration database (Smith 2002), which has 3,502 records for Colorado, Utah, Arizona and New Mexico. Within this set, 2,952 mines had at least some ore production (Table 2.1), similar in number to the MAS/MILS data.

Table 2-1. Mine Sizes for Four-Corners States

Of ~3,500 uranium mines in Colorado, Utah, Arizona, and New Mexico, 2,952 mines had at least some ore production.

Ore Production (Tons)	Number of Mines
<100	1,192
100–1,000	615
1,000–100,000	952
>1,000,000	5
Data withheld as confidential business information.	188
Total	**2,952**

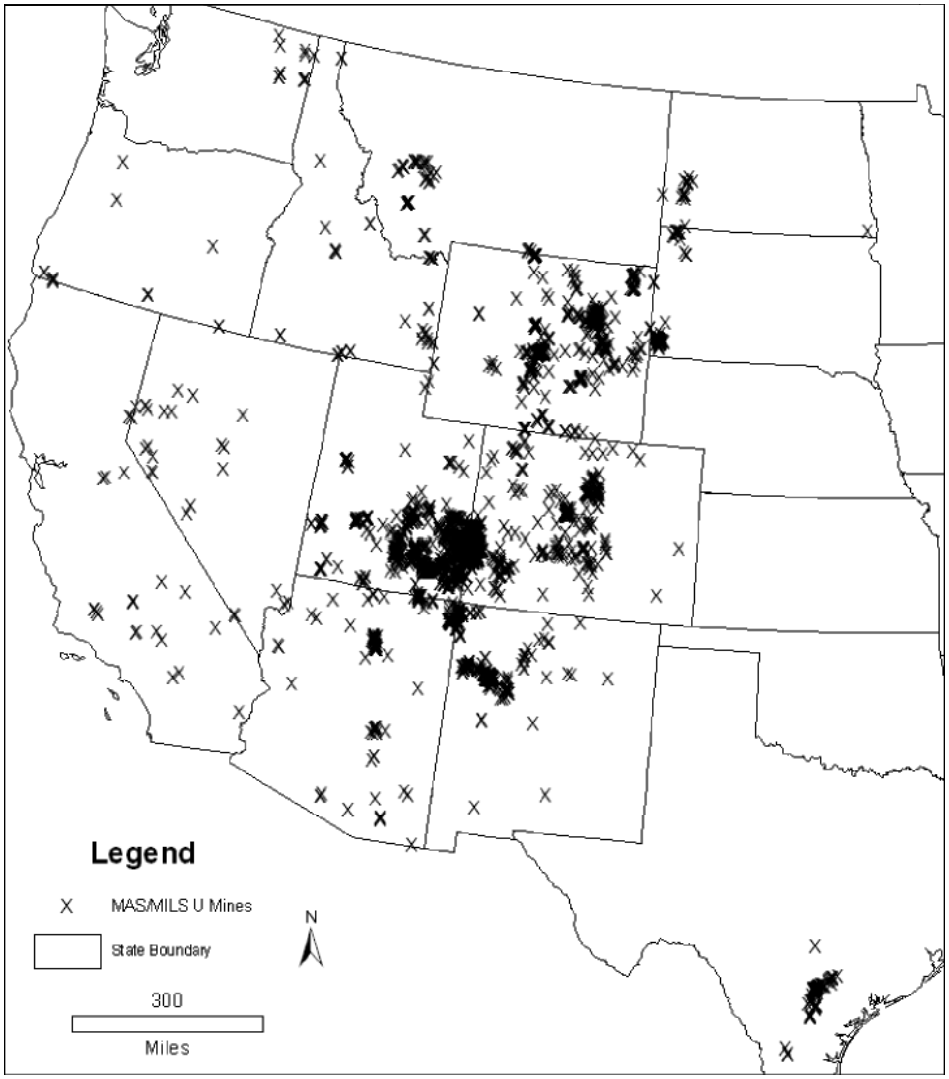
Source: Smith 2002.

The definition of a mine leads to problems with determining how many mines really exist. Even a single data set may have different interpretations for what could be considered a mine. Records may indicate multiple mine portals for an underground mine, for example. EPA has compiled a database of uranium locations from different sources totaling about 15,000 records, from which an attempt has been made to remove redundant records (U.S. EPA 2006b). The EPA database thus lists several thousand more mines than any other data set. Table 2.2 compares the number of records by state for the USGS MAS/MILS database (U.S. EPA 2001c) and unpublished USGS data sets by Finch (1998). The BASINS MAS/MILS database typically lists more mines than the Finch data set, although Finch has noted more mines in Texas and South

Dakota. The EPA ULD Compilation (U.S. EPA 2006b), as sorted for this analysis,[1] contains nearly 11,000 records, and typically has more uranium locations per state than the other data sets.

Figure 2-1. Mines and Other Locations with Uranium in the Western U.S.

Hundreds of active and abandoned uranium mines are scattered over wide areas of the western United States.



Source: MAS/MILS Database.

[1] For this comparison, the EPA ULD Compilation was sorted to delete the Mineral Resource Data System (MRDS) data, because many of the records were identified as simply drill holes, or mineral locations and also included many eastern locations not relevant to this study. In addition, location names that were variations on unknown or unnamed in the MINE NAME field in the ULD were removed so that the remaining records were more likely to be actual mining sites. For example, records with MINE NAME fields with entries such as "UNKNOWN," "UNKNOWN NAME," "UNNAMED PROSPECT," and "UNNAMED URANIUM OCCURRENCE" were deleted.

Table 2-2. Comparison of Data Compiled from Uranium Mine Records

Different data sets have different estimates of the number of uranium mines.

State	BASINS MAS/MILS		Finch	EPA ULD Compilation
	All Records	Producer or Past Producer		
Arizona	466	146	403	1,104
California	243	23	59	268
Colorado	2,286	1,631	1,262	2,268
Idaho	234	34	6	216
Missouri	2	0	0	2
Montana	195	47	31	482
Nevada	363	24	20	396
New Mexico	756	337	330	2,247
North Dakota	23	16	13	109
Oklahoma	2	0	8	0
Oregon	100	15	6	56
South Dakota	197	130	203	307
Texas	69	69	90	136
Utah	1,542	911	1,120	2,047
Washington	68	13	20	98
Wyoming	1,616	682	625	1,172
Totals	**8,162**	**4,078**	**4,196**	**10,908**

Sources: U.S. EPA 2006b, U.S. EPA 2001c, and Finch 1998.

2.1 Errors in Mine Locations

The mine record data used for most of the geospatial analyses, have two distinct error types. In addition to the definition of "mine" that was discussed above, there are errors of omission and commission (i.e., erroneous locations in the database, as well as actual mines not represented). However, accuracy of the data was checked in the EPA ULD compilation (U.S. EPA 2006b), and the mines were typically found to be within several hundred meters of mines identified on U.S. Geological Survey maps. The primary endpoint of the analyses described in this document is in terms of the radiation dose to an individual, not the collective dose to a population group. For this reason, errors in the total number of mines will not have a significant effect on the overall conclusions. There are also location precision errors (i.e., a listed mine not in its actual location as shown on USGS maps, for example). The latter are not likely to affect the analyses in this document because of the focus on risks to individuals, not populations.

2.2 Number of People Potentially Exposed to Uranium Mine Wastes

The 1983 EPA study found that, for releases to air and surface waters, the cancer risks were less than 10^{-4} and 10^{-6} for people living 1 mile or farther from active and inactive mines, respectively. Based on this information, we have assumed that the populations primarily at risk live within 1 mile (1.6 km) of uranium mines and, thus, have estimated the number of people within 1 mile of a uranium mine. We have also estimated the number of people who live nearby (within 5 miles

[8 km]) to help identify a likely population that may engage in recreational or other visitation activities in areas with unreclaimed uranium mines.

To estimate the number of people who live in proximity to uranium locations, we queried the 4,078 records in the MAS/MILS mine database in ArcView 8.2, ArcView 9.1, and Spatial Analyst (collectively, ArcView), using population data from the 2000 census (ESRI 2001). About 800,000 people are estimated to live within 5 miles of a uranium mine, and about 55,000 (or about 10 to 15 people per mine on average) are estimated to live within 1 mile of such a mine. About 10,000,000 people are estimated to be within 50 miles (80 km) of a recorded mine, with 502 of 4,078 mines located within 50 miles (80 km) of cities whose population is greater than 30,000. A search indicates that 33 of the recorded mines are within 1 mile (1.6 km) of a U.S. Bureau of Census "place" in the ArcView database, most of which are in Colorado; 141 of the mines are within 5 miles of a place (Table 2.3). In comparison, an analysis of the 10,908 "mine" locations from the ULD found that the population within 1 mile (1.6 km) and 5 miles (8.0 km) of a uranium location was 227,692 and 3,993,642, respectively.

The low number of people living within 1 mile (1.6 km) of a mine can be attributed to the fact that 7,076 of the MAS/MILS 8,234 records (86 percent) are located on federal land, while about 90% of the mines with known production are on federal land (Table 2.4). In the ULD data set, 8,124 of the 10,908 locations (74.5%) of the locations were on federal land (Figure 2.2 is a map of the ULD locations and federal lands). A query of the 7,076 mine records using ArcView revealed that 6,127 mines could be attributed to a specific federal land management agency, with most on U.S. Department of the Interior lands or Forest Service lands (Table 2.4). With the majority of the mines on federal land, people who use these sites for recreation would most likely be subjected to the greatest potential for exposure to uranium mine wastes. An exception to this would be the uranium mines on Tribal lands, where the Tribal members would receive the greatest exposure potential. Five percent (221) of the 4,078 mine records in the MAS/MILS database are on Bureau of Indian Affairs land, while eight percent (898) of the 10,908 records of the EPA ULD used in this analysis are on Bureau of Indian Affairs land.

Of the 69 mines in the MAS/MILS data identified in Texas, none are on federal lands. Over one half of the past-producer mines in Wyoming (456 of 682) are on federal lands. Of the 1,631 mines in the past-producer Colorado data set, 1,572 are on federal lands.

2.3 Mines by Watershed

One method used to view the potential for impact by mining on a region and to identify the most likely areas to be affected is on a watershed basis using geographic information system technology (Ferderer 1996). In Figure 2.3, uranium mines have been grouped in watersheds identified by 8-digit hydrologic unit codes (HUCs). Several watersheds have more than 100 uranium mines while a number of others have more than 50 mines. As might be expected from the discussion above, the highest watershed mine density is in Colorado, Utah, and Wyoming. In the watersheds with only a few mines, the mines typically produced uranium as a by-product of other mining, such as copper. One example is the Lefthand Creek mining area along the Front Range in Colorado where gold and silver were the primary metals mined, but also mined were tungsten, copper, fluorspar and uranium (U.S. EPA 2003b). Watersheds are also a unit considered in mine remediation (U.S. EPA 2003b, Buxton et al. 1997).

Table 2-3. Estimated Number of People within 1 Mile (1.6 km) and 5 Miles (8 km) of a Recorded Mine

The 4,078 mine records in the BASINS MAS/MILS database and 10,908 records from the EPA Uranium Location Database Compilation (U.S. EPA 2006b) were queried for the number of people near the uranium locations. Colorado accounts for most of the population living near current and past uranium mines.

State	People within 1 Mile	People within 1 Mile	People within 5 Miles	People within 5 Miles
	From 4,078 Records Using Producer or Past Producer and 2000 Census Data	***From 10,908 Records of EPA ULD and 2000 Census Data***	***From 4,078 Records Using Producer or Past Producer and 2000 Census Data***	***From 10,908 Records of EPA ULD and 2000 Census Data***
Arizona	1,045	21,727	12,160	438,581
California	1,068	34,867	59,437	758,545
Colorado	33,191	67,319	518,357	1,188,827
Idaho	494	5,399	5,803	89,486
Montana	891	5,954	8,233	89,573
Nevada	188	17,369	11,332	577,189
New Mexico	6,013	46,736	84,869	512,102
North Dakota	1,114	1,262	2,159	3,518
Oregon	370	1,134	6,162	30,894
South Dakota	2,889	2,956	5,954	8,538
Texas	591	871	11,700	32,640
Utah	1,387	7,169	22,376	106,015
Washington	162	5,144	3,472	79,200
Wyoming	5,196	9,785	61,701	78,534
Totals	**54,599**	**227,692**	**813,715**	**3,993,642**

Figure 2.4 illustrates one region of high-density uranium locations in drainages in southwest Colorado and eastern Utah. Figure 2.4 contains surface and underground mines, in addition to mines whose types are listed as “unknown” in the MAS/MILS database. This region typically has horizontal rock layers that have been incised by streams exposing the uranium-bearing layers, such as the Chinle Formation. In this figure, flat-lying areas appear generally featureless, whereas areas incised by streams show relief and appear to be v-shaped. Many of the mine locations are adjacent to streambeds where the mining has taken advantage of exposed uranium layers. The slopes along the canyon walls could enhance movement of radioactive materials to streambeds via mass-movement processes. Since radium and uranium may largely precipitate out of solution or adhere to particles and come to rest in sediments, benthic organisms may be the most potentially affected. However, large-magnitude events (e.g., flooding) could resuspend the material and move it around the streambeds, with higher concentrations likely developing in slack-water deposits where the water flow slows.

Figure 2-2. Uranium Locations from EPA Database and Federal Lands

About three-fourths of the uranium locations in the EPA Uranium Location Database are on Federal Lands. Thus, the most likely exposure or risk scenario for many of the uranium mine locations is the recreational scenario, such as hiking, camping, use of all-terrain vehicles or other short-term activity.

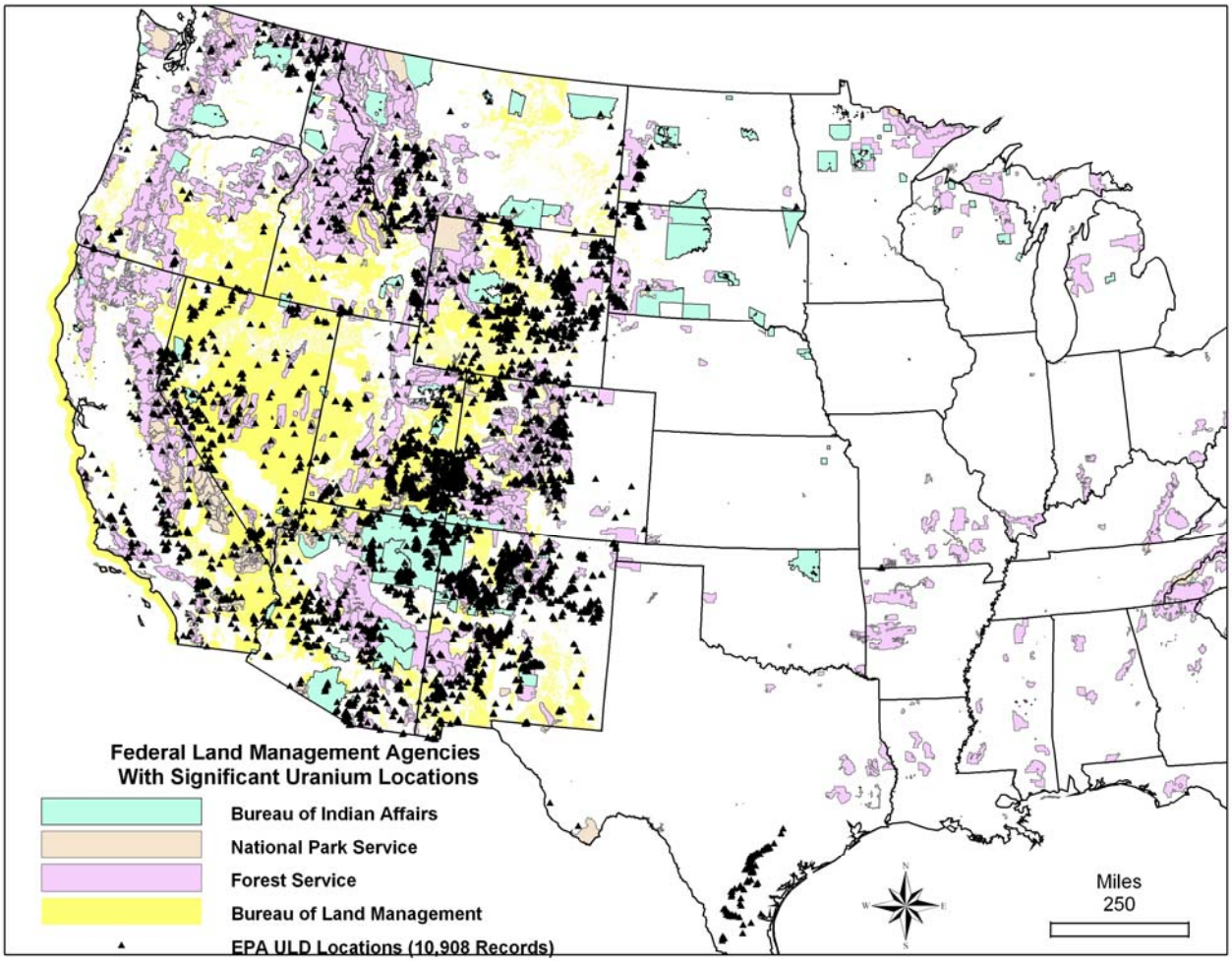


Table 2-4. Number of Mines on Federal Lands in Selected States

Most of the uranium mines on federal lands can be attributed to a specific federal land management agency; the U.S. Department of the Interior and U.S. Forest Service are the two primary land management agencies.

Federal Land Management Agency	From 8,234 Records in BASINS MAS/MILS Database	From 4,078 Records Using Producer or Past Producer and 2000 Census Data
Department of Interior		
Bureau of Land Management	4,241	2,405
Fish and Wildlife Service	7	0
Bureau of Indian Affairs	446[a]	223
National Park Service	121[b]	43
Bureau of Reclamation	3	1
Department of Defense	12	6
Forest Service (USDA)	1,297	515
Unknown	949	500
Total	**7,076**	**3,693**

a Primarily on Navajo lands.in Arizona, New Mexico, and Utah, in that order

b Primarily in Utah and California, with California primarily having unnamed prospects.

Figure 2-3. Western Uranium Mine Density by 8 Digit Hydrologic Unit Code

The greatest number of mines (745) in the MAS/MILS data is found in the Upper Dolores Watershed, located primarily in southwest Colorado with a small area in Utah. Other watersheds with more than 300 uranium mines are the Lower Dolores (Colorado and Utah) and San Miguel (Colorado) Watersheds.

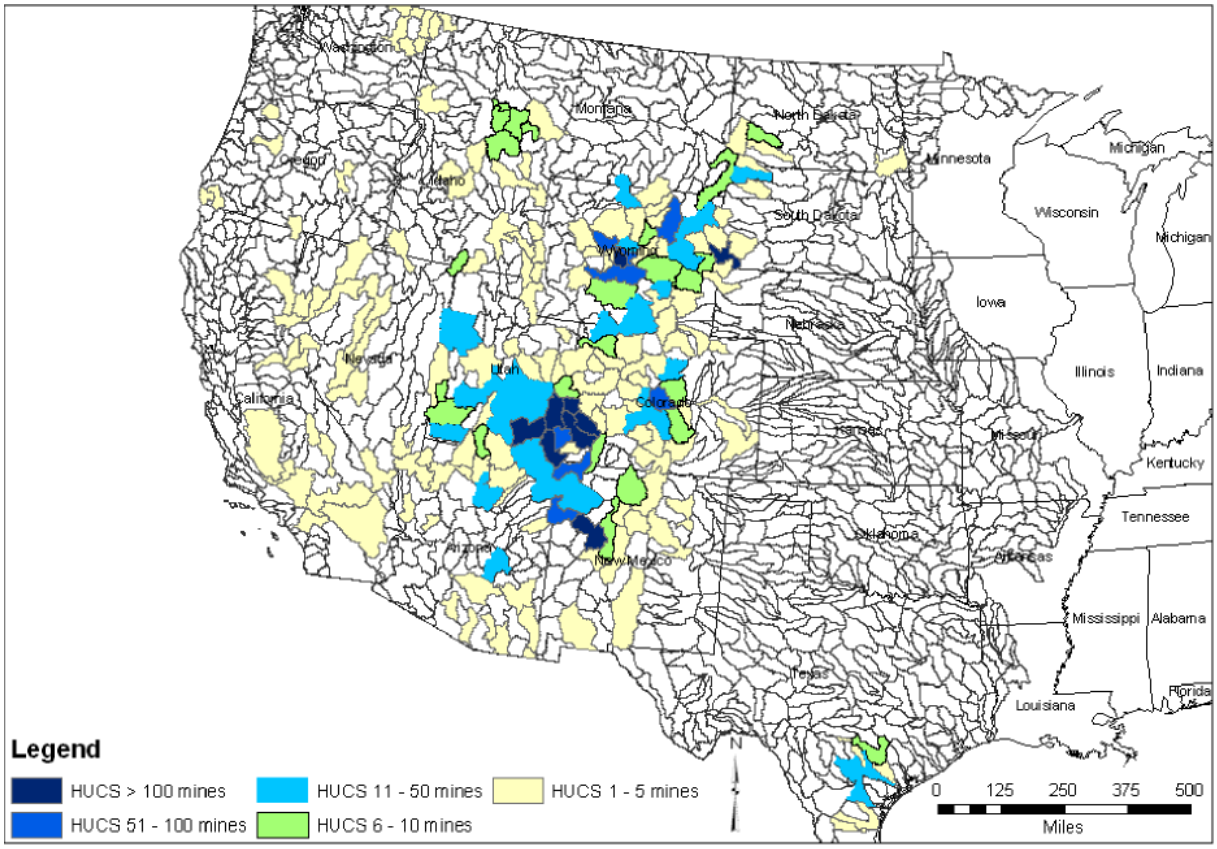
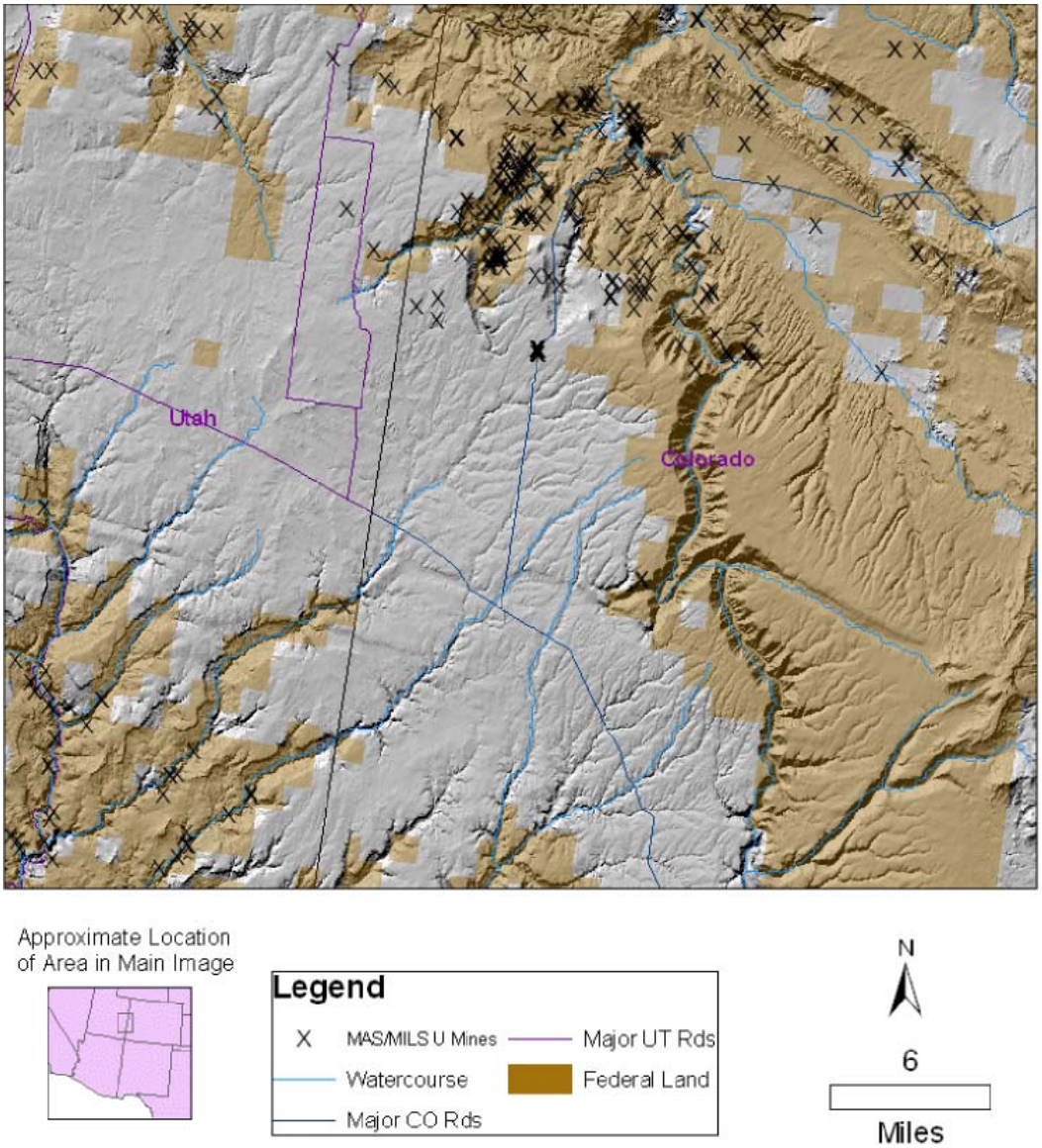


Figure 2-4. Uranium Locations in Southwest Colorado and Southeast Utah

This region typically has horizontal rock layers that have been incised by streams exposing the uranium bearing layers, such as the Chinle Formation. Flat-lying areas appear generally featureless, whereas areas incised by streams show relief and appear to be v-shaped. Many of the mine locations are adjacent to streambeds where the mining has taken advantage of exposed uranium layers. Mines from the MAS/MILS data are superimposed on digital elevation data.



3.0 CANCER RISKS FROM ON-SITE EXPOSURE

This chapter examines the potential scenarios, exposure pathways, and risks of cancer to humans that may be posed by exposure to TENORM from abandoned uranium mine wastes.

3.1 Potential Scenarios and Exposure Pathways for the General Public

Given our knowledge of uranium mine TENORM wastes,[1] there are several possible exposure scenarios for humans to the various hazards posed by these materials: on-site recreation, homes with contaminated building materials, on-site residents, and near-by residents.

3.1.1 On-Site Recreation

Since most uranium locations are on federal lands, the primary exposure scenarios to TENORM wastes at uranium mines would involve recreational use of the site, in which the abandoned mine is visited occasionally by hikers, campers, or driven through by all-terrain vehicles (ATVs). Recreational use by children may occur if a site is located near houses, as, for example, on Tribal lands in Arizona and New Mexico. A typical recreational scenario might take place at the White King and Lucky Lass mines in Oregon, which are on national Forest Service land and can be accessed only by hikers. A less common but more troubling recreational case involved the pit lake at the Yazzie-312 surface mine in Cameron, Arizona, which was approximately 300 feet (~100 meters) across and referred to by local citizens as the "swimming hole" (see Figure 3.1). The site, just off a highway, attracted swimmers because the area lacks natural lakes or streams, other than during periods of the year when the rainfall is heavy. The pit has since been filled and the area reclaimed. Users would likely visit unreclaimed uranium mines for short periods of time, such as two weeks, which is the common maximum time for which the National Park Service issues backcountry permits. Occupational workers, such as government employees or contractors performing site investigations, could also spend similar periods of time at these locations. The primary exposure pathways would be external exposure and drinking contaminated water from an adjacent spring or stream. Pathways of secondary importance include inhalation of dust, exposure to radon, ingestion of dust on dried or prepared foods, and inadvertent ingestion of soil.

3.1.2 Building Materials

A second scenario that has been known to occur, but whose frequency is unknown, is the use of uranium mine waste materials for building construction. Although most of the uranium locations are in areas where recreation is the most likely scenario, some uranium locations are near roads, including unimproved dirt roads, or near rural communities where waste material could be accessed. These materials could be transported from a nearby site and used in the construction of houses, when other building materials are difficult or too expensive for a homeowner to obtain. A discussion of risks from uranium mine wastes in building materials is presented in Chapter 4 of this report.

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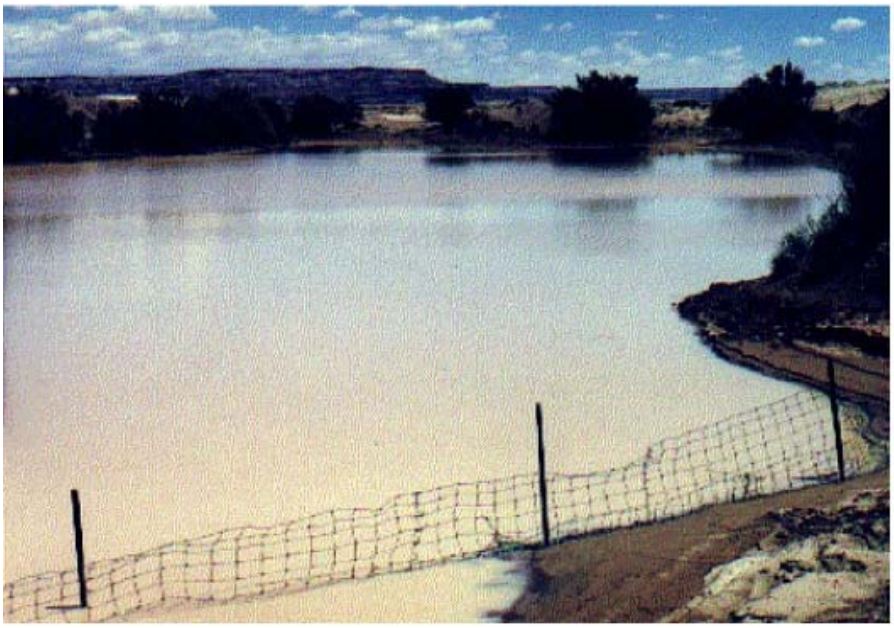
[1] Characteristics and origins of wastes mentioned in this study are more fully described in Chapter 3 of Volume I of this report (U.S. EPA 2006a).

3.1.3 On-Site Residents

A third scenario involves on-site residents. Given such factors as the nature of uranium mine waste materials, the isolation of many of the sites, the lack of potable water in many cases, and the lack of infrastructure, this scenario may have a low probability, except for some Tribal populations. The risks for such a scenario would be at the highest end of the risk spectrum and would provide an upper bound for risks. The White King Mine analysis of risks found that a future resident at the White King Superfund site would have an extremely high risk of developing cancer (see Table 1.6). Subpart B of Title 40 of the Code of Federal Regulations, Part 192 (40 CFR 192), which establishes cleanup standards for uranium processing sites, uses a radium surface soil standard of 5 pCi/g (185 Bq/kg) above background, or below, as the cleanup level, with the emphasis on preventing elevated radon levels. This radium cleanup level has been used as a relevant and appropriate requirement to establish cleanup criteria at some Superfund sites. The radon flux standards in 40 CFR 192 assume sand-like uranium mill tailings and limit the radon flux rate to 20 pCi $m^{-2}s^{-1}$. Uranium mine overburden, or protore, has elevated radon flux rates in a similar range as uranium mill tailings, although the average flux rates may be lower as described by SC&A (1989) and U.S. EPA (2006a, Chapter 3).

Figure 3.1. Uranium Mine Pit Lake

Pit lake of Yazzi-312 surface mine in Cameron, Arizona, referred to by local citizens as the "swimming hole." Suspended sediment transformed the pit water to a milky white color. The pit lake has been reclaimed.



Photograph by Loren Setlow (U.S. EPA)

3.1.4 Nearby Residents

The last scenario involves people living next to a uranium mine, which has been found to occur in the Navajo Nation. People may live within a short distance of overburden piles and be exposed to uranium from windblown particulates (inhalation of dusts), contaminated water, and external radiation.

The 1983 EPA Report to Congress studied this scenario as part of an investigation of risks to the hypothetically maximally exposed individual located 1 mile (1.6 km) from the center of average and large active and inactive mine sites (EPA 1983a, b, c). The 1983 EPA Report to Congress examined ten pathways. The study looked at risks from pathways including inhalation of radon decay products, external exposure, eating food grown in the area, fish consumption, and drinking milk and eating meat from cattle that had grazed in the area and consumed contaminated water. The study concluded that most of the pathways did not pose great risks.

The study found that radon posed the greatest risk in all scenarios, with large active underground mines emanating the highest concentrations. The maximally exposed individual's risks from radioactive airborne emissions from inactive surface and underground uranium mines were modeled and estimated to be 3.4×10^{-5} and 2.0×10^{-5}, respectively. These risk estimates assumed exposure for 71 years to inactive mine effluents. Similar results were calculated in the 1989 NESHAPs (National Emission Standards for Hazardous Air Pollutants) study (U.S. EPA 1989c). As discussed in Chapter 1, the estimated risk per working level has increased, so these risks would be slightly higher than those identified in the 1983 report and in the 1989 study (U.S. EPA 1989a). The updated risk estimates for inhalation of radon decay products from the 1983 study are listed in Table 1.5. However, since this scenario was considered in the 1983 Report to Congress and in the 1989 NESHAPs study, it is not considered further in the present analysis.

3.2 Methodology Used in This Analysis

This report focuses on risks that uranium mine TENORM wastes could pose for those people who visit inactive uranium mine sites. This analysis complements the 1983 EPA study, which looked primarily at off-site exposures from uranium mines, although it acknowledged the potential on-site health hazards. A key purpose of this approach is to help prioritize the types of uranium mine site wastes and exposures that pose the greatest risk. While some of the analysis examines residential exposure on a site, the focus is more on non-residential uses for the reasons discussed in this section.

Given the limited available data, multiple site characteristics, and the multimedia exposure pathways, multiple approaches were taken to evaluate the risks at these sites. These include reviewing existing data discussed earlier, using geographically-based queries of uranium mine and population data, the Superfund Soil Screening Guidance (SSG) approach for chemicals and radionuclides whenever applicable (U.S. EPA 1996a and 2000b), risk calculations produced for the radionuclides in drinking water regulation (U.S. EPA 2000c), and the use of RESRAD BUILD 3.21 (Yu et al. 1994) for examining building materials. This approach uses applicable peer-reviewed methodologies. The equations in the *Soil Screening Guidance: User's Guide* (U.S. EPA 1996a**)**, *Soil Screening Guidance for Radionuclides* (U.S. EPA 2000b), and *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* (U.S. EPA

2002) were used, because they are appropriate for looking at generic sites when only limited site-specific data are available. Since the intent of this analysis is meant to be scoping in nature and the information on the sites is limited, the SSG approach is appropriate for identifying the situations that may be of concern. Since this approach is for screening purposes where the intent is to ensure that potential problems are identified, the SSG methodology tends to lead to conservative risk estimates, or risks that are more likely to be overestimated. The risk estimates become more accurate with more site-specific data. Please note that all quantified risks included in this report refer to lifetime cancer risk.

An approach used at Superfund and RCRA (Resource Conservation and Recovery Act) sites is to identify preliminary remediation goals (PRGs) that are intended as initial guidelines, and not necessarily as final cleanup levels. PRGs are risk-based concentrations (assuming a target lifetime risk of 1×10^{-6}), derived from standardized equations similar to those found in the *Soil Screening Guidance for Radionuclides* (EPA 2000b). An Internet-based PRG calculator and tables of default values for radionuclides can be found at http://epa-prgs.ornl.gov/radionuclides. A major difference between the SSG methodology and the PRG approach is that the SSG methodology allows examination of an individual pathway, while the PRG uses an all-pathway approach. Since part of the intent of this analysis was to investigate individual pathways, the SSG approach was used. In addition, the PRG approach does not have a recreational scenario, which is a primary scenario identified for these mines. Although this approach was not used in this report to evaluate risks, for illustrative purposes the preliminary remediation goals for several scenarios are presented in Table 3.1.

Using the conservative SSG for radionuclides methodology, we have made some estimates of lifetime cancer risk for different exposure time periods and different concentrations for natural uranium, Ra-226, and Th-232. Natural uranium is assumed to include U-234, U-235, and U-238, in natural isotopic abundances. U-238 is in secular equilibrium with its short-lived progeny, U-234 is in secular equilibrium with Th-230, while U-235, Ra-226, and Th-232 are in secular equilibrium with their entire decay chains. The slope factors for natural uranium are expressed in terms of pCi of U-238.[2] Arsenic was evaluated using a similar approach, but using the general SSG (U.S. EPA 1996a and 1996b) methodology.

[2] For example, the inhalation slope factor (lifetime risk of cancer morbidity per pCi inhaled) for Ra-226 includes the contribution of all of its short- and long-lived progeny. This approach was employed because exposure to airborne radium particles at a mine site would most likely include most of its progeny in equilibrium. This approach slightly overestimates the risks in the case of Ra-226, because the progeny may not be in full equilibrium since some of the Rn-222 may have diffused away. The uranium slope factors do not include Ra-226 and its progeny, because separate SSLs are developed for Ra-226.

Table 3.1. Selected Radionuclide Toxicity and Preliminary Remediation Goals for Superfund for Comparison with the SSG Pathway-Specific Approach

Element and Isotope	Preliminary Remediation Goals (PRGs) (for concentrations above background)			Soil to Groundwater
	Residential Soil (pCi/g)	Agricultural Soil (pCi/g)	Outdoor Worker Soil (pCi/g)	DAF = 20 (pCi/g)
Radium 226 + D	0.012	0.0006	0.026	0.32
Thorium 232	3.1	0.0094	1.9	6.1
Uranium 238 + D	0.74	0.0015	1.8	0.12

D means that decay products are included
DAF is Dilution/Attenuation Factor
Table Source: August 4, 2004 Radionuclide Toxicity and Preliminary Remediation Goals for Superfund, at http://epa-prgs.ornl.gov/radionuclides/download.shtml.

3.3 Recreational Scenario Risk Calculations

3.3.1 Risk from External Exposure to Radium, Thorium, and Uranium

The SSG methodology assumes a linear relationship between a person's incremental cancer risk from exposure to radium (Ra-226), thorium (Th-232), and natural uranium (U-238 + U-235). The incremental lifetime cancer risk level of 10^{-6} is usually the baseline level of risk that is acceptable, and 5×10^{-4} is typically at the high end of the range of acceptability. Thus the Soil Screening Levels (SSLs) are evaluated for this range.

$$\text{Soil Screening Level (SSL)} = \frac{TR}{SFE * EF/365 * ED * ACF * [ETO + (ETI*GSF)]}$$

where:

TR	=	Target lifetime cancer risk (unitless)	variable ($1 \times 10^{-6} – 5 \times 10^{-4}$)
SFE	=	Slope factor for external exposure to soil contaminated	1.23×10^{-5} for Th-232
	=	8.49×10^{-6} for Ra-226	2.14×10^{-7} for U–natural
EF	=	Exposure frequency (days/year)	variable
ED	=	Exposure duration (years); results in risk per total number of days on site For residential exposure, ED is used to represent the exposure over a number of years, frequently 30 years.	1
ACF	=	Area correction factor for smaller sites	
	=	0.9 if area < 1,000 m^2	1
ETO	=	Estimated fraction of time outdoors on site	1
ETI	=	Estimated time indoors	0
GSF	=	Gamma-shielding factor	0

[3] Includes short- and long-lived decay products, as discussed in preceding section. Slope factors for radionuclides for all exposure pathways are based on U.S. EPA's Health Effects Assessment Summary Tables (HEAST) (http://www.epa.gov/radiation/heast/index.html). The slope factor calculations can be found in Appendix II Calculation of Slope Factors for NORM Decay Series.

Because of the nature of the recreational scenario, some of the typical assumptions have been changed. In the above equation we assume that the person spends the entire day at the site, with no indoor time—that is, the individual spends all day on the waste material and sleeps in a tent or other light structure that provides no appreciable shielding. Since no time is spent indoors, the indoor part of the equation with the gamma shielding does not come into play. For a Superfund target risk of 1×10^{-6} for 14 days of exposure and the assumptions stated above, the Ra-226 soil screening level would be ~3.1 pCi/g (~114 Bq/kg), but for one day of exposure at a 1×10^{-6} target risk, the Ra-226 soil screening level would be ~43 pCi/g (~1,590 Bq/kg). Table 3.2 and Figure 3.2 illustrate the relationship between radium concentration and risk for different times of exposure, Table 3.3 and Figure 3.3 present the corresponding data for exposure to thorium, and Table 3.4 and Figure 3.4 provide similar information for uranium. The relationship is linear, so reducing the estimated time on site by one half (from 100 percent of the time on site to 50 percent) would increase the radium screening level by a factor of two for the same target risk. In addition, if a typical residential exposure duration of 30 years is used, then the values in Table 3.2 and other tables of soil screening levels used in this chapter would need to be divided by 30; however, the assumptions used here (i.e., entire day on the waste material) would not be appropriate for a typical residential scenario. The risk estimated for a recreational exposure could also be used for occupational workers (government workers or contractors for example) who spent time at the site for their jobs.

Table 3.2. Soil Screening Levels for External Exposure to Ra-226

Table 3.2 lists the data used to generate Figure 3.2.

Exposure Frequency (days)	Target Lifetime Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Ra-226 (pCi/g)					
1	21,485	4,297	2,149	430	215	43.0
14	1,535	307	153	30.7	15.3	3.07
30	716	143	72	14.3	7.2	1.43
52	413	83	41.3	8.3	4.13	0.83
140	153	30.7	15.3	3.07	1.53	0.307
350	61.4	12.3	6.14	1.23	0.614	0.123

Figure 3.2. External Exposure - Relationship between Exposure Frequency, Radium Concentration, and Target Lifetime Cancer Risk

Figure 3.2 is derived from Table 3.2. The x-axis is the activity concentration of radium in the uranium mine waste material, and the y-axis is the incremental lifetime cancer risk as a result of exposure from the radium in the waste material for different time periods. For example, exposure to 12.3 pCi/g (454 Bq/kg) of radium, in secular equilibrium with its progeny, for 350 days, would result in a lifetime cancer risk of 10^{-4}.

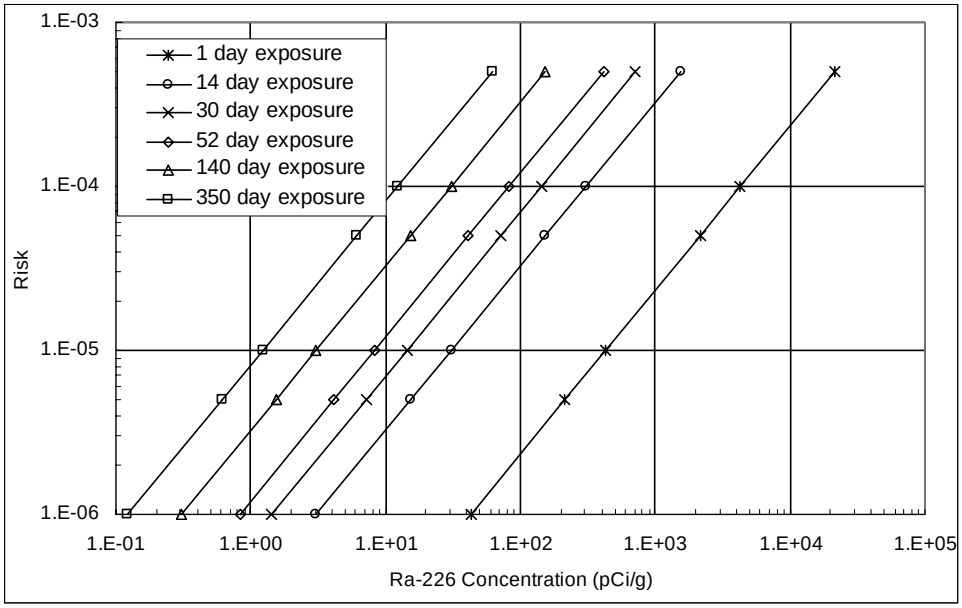


Table 3.3. Soil Screening Levels for External Exposure to Th-232

Table 3.3 lists the data used to generate Figure 3.3

Exposure Frequency (days)	Target LifetimeCancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Th-232 (pCi/g)					
1	14,849	2,970	1,485	297	148	29.7
14	1,061	212	106	21.2	10.6	2.12
30	495	99	49.5	9.9	4.95	0.99
52	286	57	28.6	5.71	2.86	0.571
140	106	21.2	10.6	2.12	1.06	0.212
350	42.4	8.5	4.24	0.85	0.424	0.085

Figure 3.3. External Exposure - Relationship between Exposure Frequency, Thorium Concentration, and Target Lifetime Cancer Risk

Figure 3.3 is derived from Table 3.3. The x-axis is the activity concentration of thorium in the uranium mine waste material, and the y-axis is the incremental lifetime cancer risk as a result of external exposure to the thorium in the waste material for different time periods. For example, exposure to 8.5 pCi/g (314 Bq/kg) of Th-232, in secular equilibrium with its progeny, for 350 days, would result in a cancer risk of 10^{-4}.

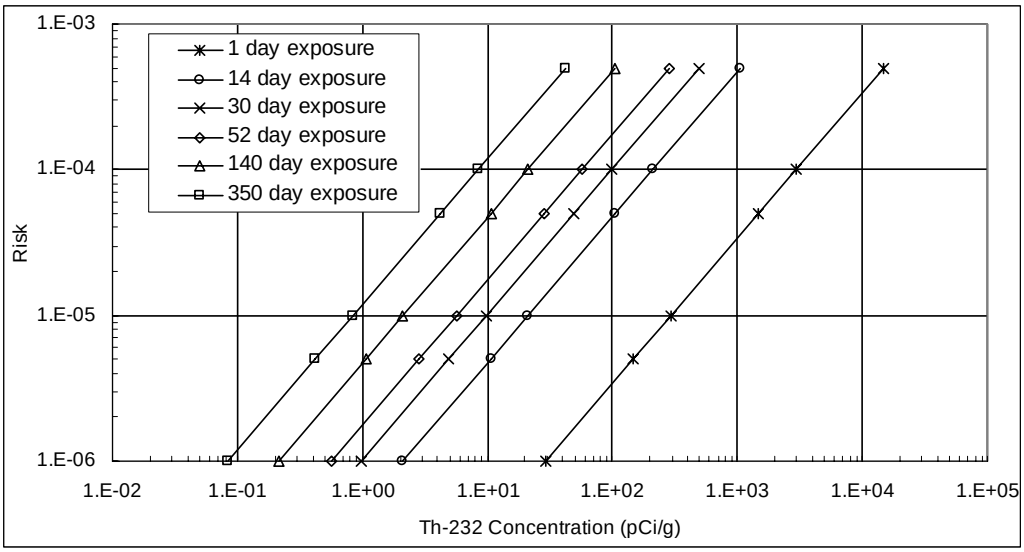


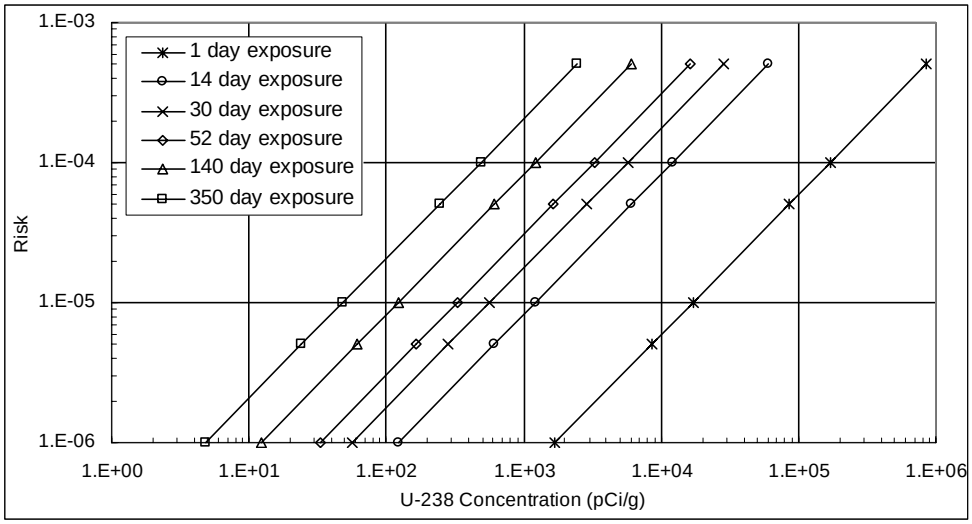
Table 3.4. Soil Screening Levels for External Exposure to Natural Uranium

Table 3.4 lists the data used to generate Figure 3.4

Exposure Frequency (days)	Target Lifetime Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Natural Uranium (pCi/g U-238)					
1	852,189	170,438	85,219	17,044	8,522	1,704
14	60,871	12,174	6,087	1,217	609	122
30	28,406	5,681	2,841	568	284	56.8
52	16,388	3,278	1,639	328	164	32.8
140	6,087	1,217	609	122	60.9	12.2
350	2,435	487	243	48.7	24.3	4.87

Figure 3.4. External Exposure - Relationship between Exposure Frequency, Uranium Concentration, and Target Lifetime Cancer Risk

Figure 3.4 is derived from Table 3.4. The x-axis is the activity concentration of U-238 in the uranium mine waste material, and the y-axis is the incremental lifetime cancer risk as a result of exposure to uranium in the waste material for different time periods. For example, 350 days of exposure on site to 487 pCi/g (18,020 Bq/kg) of U-238, in secular equilibrium with its progeny, as well as U-235 in the ratio of natural abundance (see discussion of uranium progenies earlier in this chapter) would result in a lifetime cancer risk of 10^{-4}.



3.3.2 Risk from Soil Ingestion

While the direct ingestion of soil is possible at a site, it is not likely to be a major exposure pathway for adults. The following equation uses an age-adjusted soil ingestion factor to account for the fact that children have a higher intake of soil than adults (U.S. EPA 2000b).

$$SSL = \frac{TR}{SFs * IRs * 1 \times 10^{-3} * EF * ED}$$

where:

TR	=	Target lifetime cancer risk (unitless)
SF_S	=	Soil ingestion slope factor $(pCi)^{-1}$
		Ra-226 = 3.39×10^{-9}
		Th-232 = 3.33×10^{-9}
		U-natural = 6.48×10^{-10}
IR_S	=	Soil ingestion rate (120 mg/day)
1×10^{-3}	=	Conversion factor (g/mg)
EF	=	Exposure frequency (variable)
ED	=	Exposure duration (1 year)

Sample calculation for radium, assuming a target lifetime risk of 1×10^{-6} and exposure for 14 days:

$$SSL = 1 \times 10^{-6} \div (3.39 \times 10^{-9} * 120 * 1 \times 10^{-3} * 14 * 1) = 176 \text{ pCi/g } (\sim 6{,}500 \text{ Bq/kg})$$

Table 3.5. Soil Screening Levels for Ingestion of Ra-226 in Soil

Exposure Frequency (days)	Target Lifetime Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Ra-226 (pCi/g)					
1	1.23E+06	2.46E+05	1.23E+05	2.46E+04	1.23E+04	2.46E+03
14	8.78E+04	1.76E+04	8.78E+03	1.76E+03	8.78E+02	1.76E+02
30	4.10E+04	8.19E+03	4.10E+03	8.19E+02	4.10E+02	8.19E+01
52	2.36E+04	4.73E+03	2.36E+03	4.73E+02	2.36E+02	4.73E+01
140	8.78E+03	1.76E+03	8.78E+02	1.76E+02	8.78E+01	1.76E+01
350	3.51E+03	7.02E+02	3.51E+02	7.02E+01	3.51E+01	7.02E+00

Figure 3.5. Relationship between Exposure Frequency, Radium Concentration, and Target Lifetime Cancer Risk from Soil Ingestion

Figure 3.5 is derived from Table 3.5. The x-axis is the activity concentration of Ra-226 in the uranium mine waste material, and the y-axis is the incremental lifetime cancer risk as a result of ingestion of radium in the waste material for different exposure times.

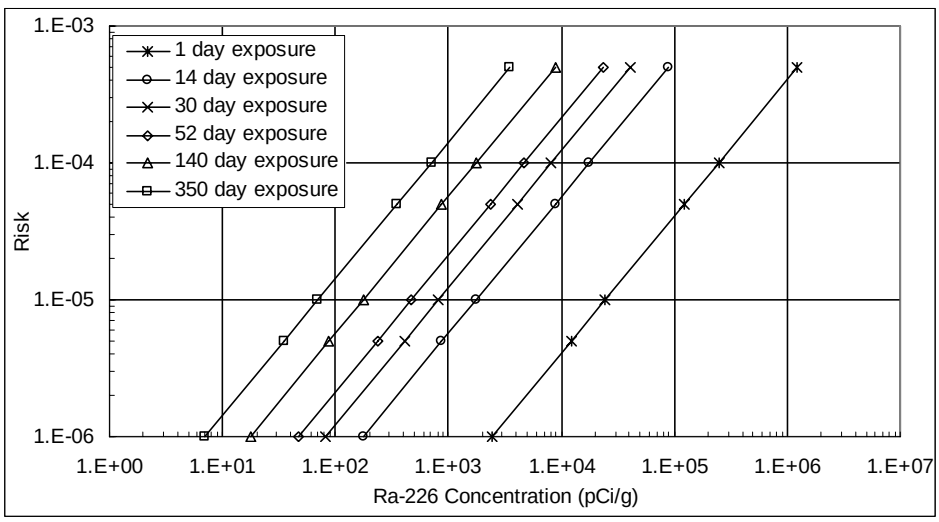


Table 3.6. Soil Screening Levels for Ingestion of Th-232 in Soil

Exposure Frequency (days)	Target Lifetime Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Th-232 (pCi/g)					
1	1.25E+06	2.50E+05	1.25E+05	2.50E+04	1.25E+04	2.50E+03
14	8.94E+04	1.79E+04	8.94E+03	1.79E+03	8.94E+02	1.79E+02
30	4.17E+04	8.34E+03	4.17E+03	8.34E+02	4.17E+02	8.34E+01
52	2.41E+04	4.81E+03	2.41E+03	4.81E+02	2.41E+02	4.81E+01
140	8.94E+03	1.79E+03	8.94E+02	1.79E+02	8.94E+01	1.79E+01
350	3.58E+03	7.15E+02	3.58E+02	7.15E+01	3.58E+01	7.15E+00

Figure 3.6. Relationship between Exposure Frequency, Thorium Concentration, and Target Lifetime Cancer Risk for Soil Ingestion

Figure 3.6 is derived from Table 3.6. The x-axis is the activity concentration of thorium in the uranium mine waste material, and the y-axis is the incremental lifetime cancer risk as a result of ingestion of thorium in the waste material for different exposure times.

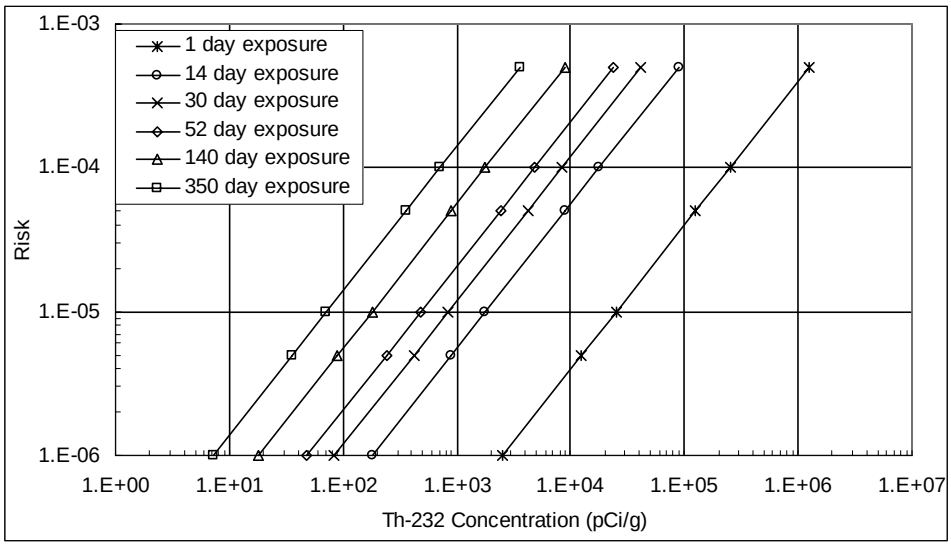
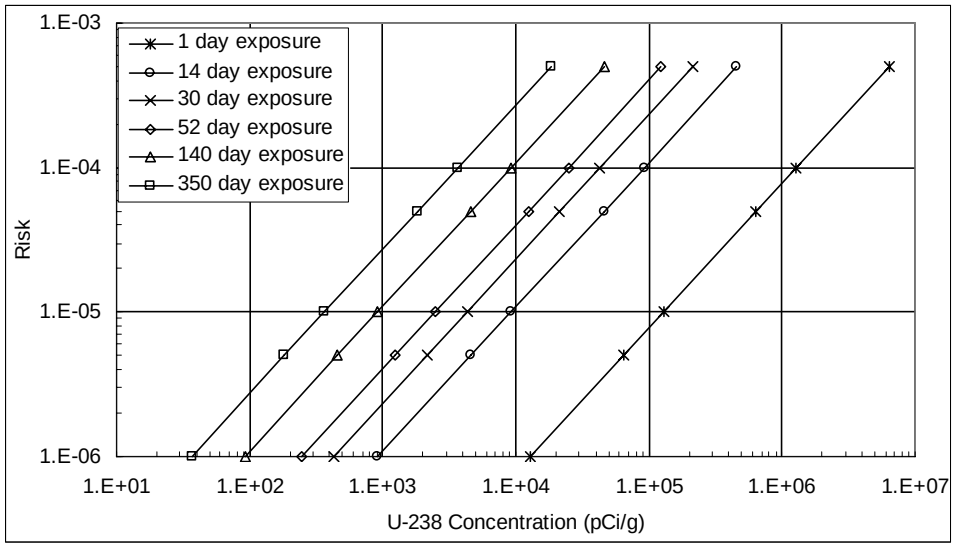


Table 3.7. Soil Screening Levels for Ingestion of Natural Uranium in Soil

Exposure Frequency (days)	Target Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Natural Uranium (pCi/g U-238)					
1	6.43E+06	1.29E+06	6.43E+05	1.29E+05	6.43E+04	1.29E+04
14	4.59E+05	9.18E+04	4.59E+04	9.18E+03	4.59E+03	9.18E+02
30	2.14E+05	4.29E+04	2.14E+04	4.29E+03	2.14E+03	4.29E+02
52	1.24E+05	2.47E+04	1.24E+04	2.47E+03	1.24E+03	2.47E+02
140	4.59E+04	9.18E+03	4.59E+03	9.18E+02	4.59E+02	9.18E+01
350	1.84E+04	3.67E+03	1.84E+03	3.67E+02	1.84E+02	3.67E+01

Figure 3.7. Relationship between Exposure Frequency, Uranium Concentration, and Target Lifetime Cancer Risk from Soil Ingestion

Figure 3.7 is derived from Table 3.7. The x-axis is the activity concentration of U-238 in the uranium mine waste material, and the y-axis is the incremental cancer risk as a result of ingestion of uranium in the waste material for different exposure times.



3.3.3 Risk from Inhalation of Radium, Thorium, and Uranium in Fugitive Dust

Windblown dust provides a pathway for radioactive materials to reach humans through inhalation. The equation for inhalation from the Superfund SSG (EPA 2000b) is:

$$SSL = \frac{TR}{SF_I * IR_I * (1/PEF) * 1 \times 10^3 * EF * ED * [ETO + (ETI * DFI)]}$$

where:

TR	=	Target lifetime cancer risk (unitless)	10^{-6}
			2.55×10^{-8} Ra-226
SF_I	=	Inhalation Slope Factor (pCi^{-1})	1.92×10^{-7} Th-232
			6.14×10^{-8} U-natural
IR_I	=	Inhalation Rate (m^3/day)	20
PEF	=	Particulate Emission Factor (m^3/kg)	1.32×10^9
1×10^3	=	Conversion factor (g/kg)	–
EF	=	Exposure frequency (days/ year)	350
ED	=	Exposure duration (year)	1
ETO	=	Exposure time fraction, outdoor (unitless)	1
ETI	=	Exposure time fraction, indoor (unitless)	0
DFI	=	Dilution factor for indoor inhalation (unitless)	NA

Using these parameters, the 350-day SSL for Ra-226 is 7,395 pCi/g (2.74×10^5 Bq/kg), 985 pCi/g (3.64×10^4 Bq/kg) for Th-232, and 3,070 pCi/g (1.14×10^5 Bq/kg) for natural uranium. This applies to exposed individuals in the vicinity of the mine.

3.3.4 Risk from Use of All-Terrain Vehicles (ATVs)

The recreational use of ATVs and dirt bikes in the western United States is very common. These vehicles allow easy access to very remote areas, so the safety provided by a mine's remote location is often negated. The soil screening levels for inhalation of dust resuspended during the operation of ATVs are estimated from empirical data on emission of dust from unpaved roads. A scoping scenario for this pathway was developed, as described below.

It is assumed that a rider of an ATV or other off-road recreational vehicle riders would participate in the sport about 60 times a year (once a week plus additional days on vacations or holidays). It is further assumed that an abandoned mine site would lie on his route, and that he would cross the site twice on each ride, going and returning over the same route. The area of the site is 463.5 hectares (ha), the average of the total disturbed areas of the 21 mines listed in DOE/EIA 2000b, Appendix C. This is a bounding condition as the estimated size of an abandoned mine is expected to be much less, on the order of two hectares (U.S. EPA 2006a). The area is assumed to be circular, and the route to be along the diameter of the circle. The riders have inhalation rates of 1.2 m^3/h, the average rate for light activity. The vehicles travel at an average speed of 40 mph. The airborne concentration of respirable dust, 5 mg/m^3, is based on the average of three measured dust concentrations at a height of 2 m taken at the side of a road composed of dirt and crushed slag, during the passage of medium-duty vehicles (3–4 tons) traveling at a speed of 15 mph (Cowherd et al. 1979). The dust had a mass-median diameter of 10–11 µm, and thus corresponds to the approximate range of respirable particles. As it happens, this concentration is also equal to the OSHA protective exposure limit (PEL) for nuisance dust set forth in 29 CFR 1910.1000, and thus constitutes a reasonable upper bound to the average dust loadings that could be comfortably tolerated by the rider. The SSLs are calculated using the preceding equation for inhalation of contaminated dust. The parameters that were changed for the ATV scenario are presented below.

The daily inhalation rate of the rider while exposed to the dust on the mine site is calculated as follows:

$$IR_1 = \frac{I_h \, 4 \sqrt{\frac{A_s}{\pi}}}{v}$$

where:

IR_I	=	inhalation rate during exposure(m^3/d)	=	0.0906
I_h	=	inhalation rate for light activity(m^3/h)	=	1.2
A_s	=	Area of site (m^2)	=	4.635×10^6
v	=	speed of vehicle(40 mi/h)	=	64,374 m/h

The emission factor is simply the reciprocal of the dust loading, and is evaluated as follows:

PEF	=	emission factor		
		= $1/\chi$ = 0.2 m^3/mg	=	2×10^5 m^3/kg
		χ = concentration of respirable dust	=	5 mg/m^3
EF	=	Exposure frequency	=	60 d/y
ETO	=	Exposure time fraction, outdoor	=	1

Based on these values, the SSLs calculated for this scenario are:

	Soil Screening Level	
Radionuclide	**pCi/g**	**Bq/kg**
Ra-226	1,445	5.35E+04
Th-232	192	7.12E+03
U-natural	600	2.22E+04

3.4 Other Recreational Use Scenarios

Other recreational use scenarios were considered as part of the present analysis. These include swimming, boating, fishing, and hunting, along with the consumption of on-site fish and game. These scenarios are either unlikely to occur, or would be an insignificant component of the risk, as reviewed in an EPA study (1983b). This study addressed related scenarios for nearby residents [within 1 mile (1.6 km)] of the mines, including cattle grazing and crop ingestion, as discussed below.

Although the pit lake at the Yazzie-312 Mine was used for swimming by local residents, the lake was drained and filled in as part of the remediation of the mine site after 40 years of abandonment. The number of other abandoned uranium mines with pit lakes is unknown. However, swimming, through water immersion and ingestion pathways, contributes little total dose (< 10 mrem or < 10^{-1} mSv) or risk. Estimates of risk from swimming are provided in Appendix 1. Fishing is not considered in this analysis. Pit lakes, being artificial and not connected to any natural bodies of water, are assumed to be devoid of fish or expected to contain minimal fish populations.

The majority of mine sites found in the uranium location database are typically in an arid environment that does not readily support plant life unless irrigated. In such arid environments, the overburden or protore piles are not expected to be able to provide much forage for animals, especially if they are covered with a desert varnish. In addition, the size of the abandoned mine sites would typically be relatively small and thus provide little forage for game animals. Consequently, any game taken on a mine site would be expected to have obtained most of its forage elsewhere. The meat from such game is thus not expected to be significantly contaminated with TENORM from a mine site.

3.5 Metals in Uranium Mines

Metals and other minerals of commercial value frequently occur in the same ore deposits with uranium (See Volume I, Chapters I and II, U.S.EPA 2006a) and, in some cases, it is economical to mine them together. The most common commodities associated with uranium in the BASINS

MAS/MILS (Mineral Availability System/Mineral Industry Location System) database are phosphate, vanadium, gold, and copper. U.S. EPA (1999) provided an extensive review of TENORM contamination, including uranium, associated with copper mines in Arizona. However, numerous other commodities are associated with uranium, including antimony, molybdenum, fluorine, rare earths, thorium, lead, mica, tantalum, and beryllium. For example, in Colorado, 83 of 2,304 records had gold associated with uranium, and 10 had silver as a secondary commodity; 38 records listing vanadium as a primary commodity also listed uranium as a secondary commodity; and vanadium was listed as the primary or secondary commodity with uranium in over 2,000 of the records. While multiple metals are associated with uranium mines, limited information is available to determine the concentrations of the metals at the different sites.

The 1983 EPA report to Congress indicated that at uranium mines, no adverse effects were expected from nonradiological constituents because of the low airborne concentrations, with the exception of fugitive dusts from operating mines (U.S. EPA 1983a, b, and c). Nevertheless, mining in general in the West has been known to generate problems with heavy metal contamination in sediments and water, and some mines are Superfund sites (U.S. EPA 2001d).

3.5.1 Risk from Exposure to Arsenic

Arsenic, a carcinogen, is a metal of special concern. This naturally occurring metal may be a common contaminant in uranium mine wastes. The presence of arsenic in extremely high amounts in soils, as well as in the water, posed a significant risk at the White King/Lucky Lass uranium mines. In the study (Portage Environmental 2005) of the Riley Pass Uranium Mines in Harding County, South Dakota, arsenic was considered to be "the primary risk driver." The primary exposure scenario at that site also involved recreational users of the site. The following equation is used to estimate the lifetime cancer risk from ingestion of arsenic:

$$\text{Arsenic SSL} = \frac{TR * AT * 365}{SF_O * 10^{-6} * EF * IF_{soil/adj}}$$

where:

TR	=	Target lifetime cancer risk	Variable
AT	=	Averaging time (years)	70
SF_O	=	Slope factor for arsenic $(mg/kg\text{-}d)^{-1}$	1.5
EF	=	Exposure frequency (days/year)	Variable
365	=	Conversion factor (days/y)	
10^{-6}	=	Conversion factor (kg/mg)	

$IF_{soil/adj}$ is the ingestion factor (age-adjusted), in units of mg y kg^{-1} d^{-1}. Because the recreational use of the mine site is assumed to be episodic—it would occur for a limited period of time during a given year—the limiting exposure would be to a child.[4] Employing the data for a child, 0 – 6

[4] The risks to a child were calculated for this chemical carcinogen because the expression for the ingestion factor is age dependent. This is unlike the calculation of risks from radionuclides, where the reference slope factors calculated by EPA are age adjusted.

years old, the ingestion factor is given by the following expression, modified from the expression for the residential scenario in U.S. EPA 1996b:

$$IF_{soil/age\ 1\text{-}6} = \frac{IR_{soil/age\ 1\text{-}6}}{BW_{age\ 1\text{-}6}}$$

where:

$IR_{soil/age\ 1\text{-}6}$	=	soil ingestion rate of child (mg/d)	200
$BW_{age\ 1\text{-}6}$	=	body weight of child (kg)	15

The results are presented in Table 3.8. For a target lifetime risk of 5×10^{-5} and an exposure of 7 days/year, the arsenic soil screening level for children would be ~8,250 mg/kg. The lowest SSL is 3 mg/kg for the 350-day exposure at 1x 10^{-6} target risk. For perspective, for the White King/Lucky Lass Superfund site, arsenic concentrations in surface soil were 769 mg/kg and 12 mg/kg, respectively, while background arsenic soil concentrations in the area were ~4 mg/kg. The Riley Pass Uranium Mines arsenic average concentrations were over 500 mg/g (Portage Environmental 2005). Although an occasional visitor to these sites does not incur much risk from arsenic, it could pose a problem for those who frequent the sites.

Table 3.8. Target Lifetime Cancer Risk for Ingestion of Arsenic by Children Up to 6 Years Old

Exposure Frequency (days/year)	Target Lifetime Cancer Risk					
	5×10^{-5}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Soil Screening Level for Arsenic (mg/kg)					
1	638,750	115,387	57,694	11,539	5,769	1,154
5	115,387	23,077	11,539	2,308	1,154	231
7	82,419	16,484	8,242	1,648	824	165
14	41,210	8,242	4,121	824	412	82
350	1,648	330	165	33	16	3

3.5.2 Risk from Drinking Mine-Contaminated Water

In addition to their potential to pose health risks on the site, uranium mines and their wastes can affect surface or groundwater. For example, the pond in the mining pit could be contaminated with radionuclides or metals, which would make the pond an exposure pathway. In addition, the overburden (or protore) waste materials could leach into the ground and move into the groundwater below. Material could also be physically transported from the waste piles by runoff or wind (see discussion and data on theYazzie-312 Mine in Volume I, Chapter 3, U.S. EPA 2006a). In another scenario, the mine workings could intersect and contaminate groundwater.

There are multiple scenarios in which people could drink water contaminated from unreclaimed uranium mining operations. For a recreational user of the site, the exposure may be short-term from a spring, stream, or pond. Others could have lifetime exposure due to proximity to a uranium mine. Also, someone who does not live on contaminated property could be exposed to radionuclides from communal wells, which occurs on the Navajo Reservation in the Four

Corners area (U.S. EPA and USACE 2000). The radionuclides in groundwater can be due to contamination from mining activities or from high natural background sources, including the uranium ore body exploited by the mining operation. However, many large uranium mining operations have reported problems of groundwater contamination (U.S. DOE/EIA 2000b, Appendix C).

EPA's 1983 Report to Congress studied concentrations in, and risks from, waters discharged from active mines to surface waters. The authors estimated that an insignificant health risk accrues to populations from waterborne radionuclides due to water discharges from an average existing active mine (U.S. EPA 1983b). However, the report acknowledged that some abandoned underground mines were probably discharging contaminated waters into streams and shallow aquifers, and the data were insufficient to determine the health risks from drinking the water. Furthermore, due to a lack of data, the authors could not determine the health hazard to individuals who drink from contaminated surface or underground sources. However, Volume I of this study (U.S. EPA 2006a) reports on concentrations of radionuclides in ponds and streams associated with open pit uranium mines, and case studies where shallow groundwater and surface springs or streams were contaminated by uranium mine discharges.

EPA has established maximum contaminant levels (MCLs) for several radionuclides in community water supplies that serve more than 25 customers (Table 3.9). These MCLs can be used to help establish soil cleanup levels at a site. The SSG approach is used to conservatively identify a soil level that would prevent a site contaminant from attaining the MCL in groundwater. The drinking water MCL for uranium is based primarily on kidney toxicity, rather than radiological effects.

Table 3.9. Radionuclide Maximum Contaminant Levels for Public Water Supplies

EPA has established drinking-water maximum contaminant levels for several radionuclides. Although these values are for public water supplies, the Superfund program has applied them to site cleanups.

Radionuclide	Maximum Contaminant Level
Uranium	30 µg/L
Man-made beta/photon emitters	4 mrem/y (0.04 mSv/y) to whole body or any organ
Alpha emitters (excluding radon and uranium)	15 pCi/L (555 Bq/m^3)
Combined radium-226 and radium-228	5 pCi/L (185 Bq/m^3)

Source: Modified from EPA 2000c.

While the number of people who drink water contaminated by uranium mining activities is unknown, it is possible to calculate an individual lifetime risk for various concentrations of radionuclides. The numbers in Table 3.10 are based on the risk calculations presented in the technical support document for the radionuclides in drinking-water regulation (U.S. EPA 2000d).

Table 3.10. Radionuclide Mortality and Morbidity Risk Coefficients[5]

While the number of people who drink water contaminated by uranium mining activities is unknown, it is possible to calculate an individual lifetime risk for various concentrations of radionuclides.

Radionuclide	Mortality Risk Coefficient per pCi Consumed	Morbidity Risk Coefficient per pCi Consumed
Radium[a]	5.66×10^{-10}	8.03×10^{-10}
Th-232[b]	6.92×10^{-11}	1.01×10^{-10}
Ra-228[b]	7.40×10^{-10}	1.04×10^{-9}
Th-228[b]	6.73×10^{-11}	1.07×10^{-10}
Ra-224[b]	1.01×10^{-10}	1.67×10^{-10}
Uranium[c]	4.4×10^{-11}	6.81×10^{-11}
Gross alpha[d]	1.14×10^{-10}	1.83×10^{-10}

[a] Average weighted by relative prevalence of Ra-226 and Ra-228
[b] Principal members of Th-232 decay chain
[c] Arithmetic average for natural uranium isotopes: U-234, U-235, U-238
[d] Average weighted by relative prevalence of Ra-224 and Ra-226
Source: U.S. EPA 2000d.

The equation used to calculate the risks from these radionuclides is:

Risk = Concentration (pCi/L) * Risk coefficient * Water consumed (L/day) * Exposure frequency (days/year) * Number of years

Figures 3.8–3.10 depict the risks from radium, gross alpha, and uranium for (1) 70 years of exposure, 365 days a year, drinking 2 liters of water a day from the contaminated source, representing lifelong consumption; and (2) 10 years of exposure, 14 days a year, drinking 2 liters a day, representing recreational consumption. For the first situation, long-term exposure produces risks of up to 1×10^{-3} for some of the higher concentrations. However, for the long-term recreational user consuming contaminated water, the lifetime risk remains less than 6×10^{-6}.

[5] Morbidity risk is the risk of getting cancer, and mortality risk is the risk of dying from cancer.

Figure 3.8. Cancer Risks from Lifetime and Recreational Exposures to Radium in Drinking Water: 70 Years, 365 Days/Year & 10 Years, 14 Days/Year Exposure

Long-term exposure to radionuclide-contaminated water produces risks of up to 1×10^{-3} for some of the higher concentrations. However, the risk from long-term recreational consumption is less than 6×10^{-6}.

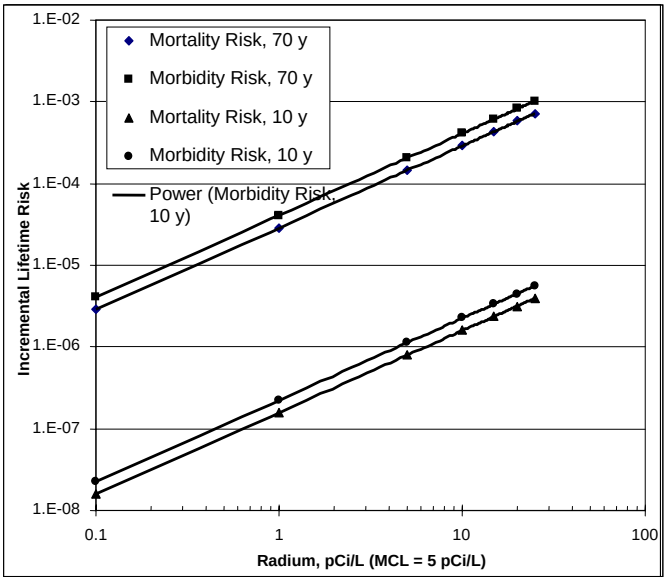


Figure 3.9. Cancer Risks from Lifetime and Recreational Exposures to Gross Alpha in Drinking Water: 70 Years, 365 Days/Year & 10 Years, 14 Days/Year Exposure

Long-term exposure to radionuclide-contaminated water produces risks of up to 1×10^{-3} for some of the higher concentrations. However, the risk from long-term recreational consumption is less than 6×10^{-6}.

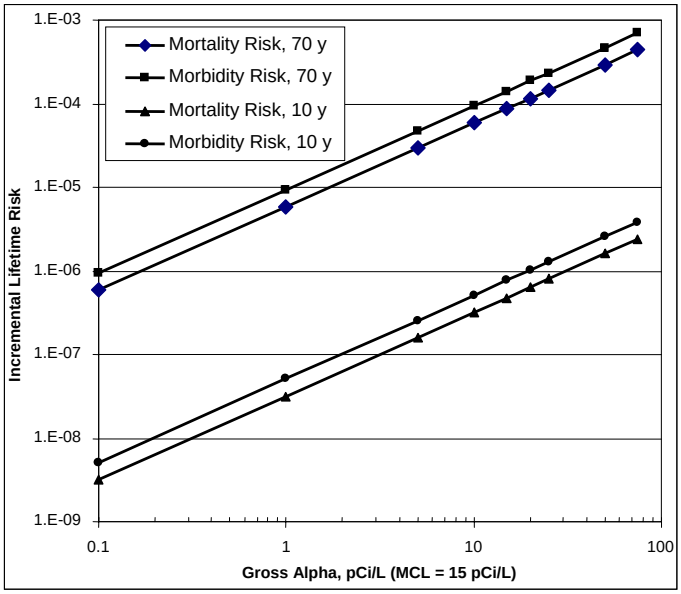


Figure 3.10. Cancer Risks from Lifetime and Recreational Exposures to Uranium in Drinking Water: 70 Years, 365 Days/Year and 10 Years 14 Days/Year Exposure

Long-term exposure to radionuclide-contaminated water produces risks of up to 1×10^{-3} for some of the higher concentrations. However, the risk from long-term recreational consumption is less than 6×10^{-6}.

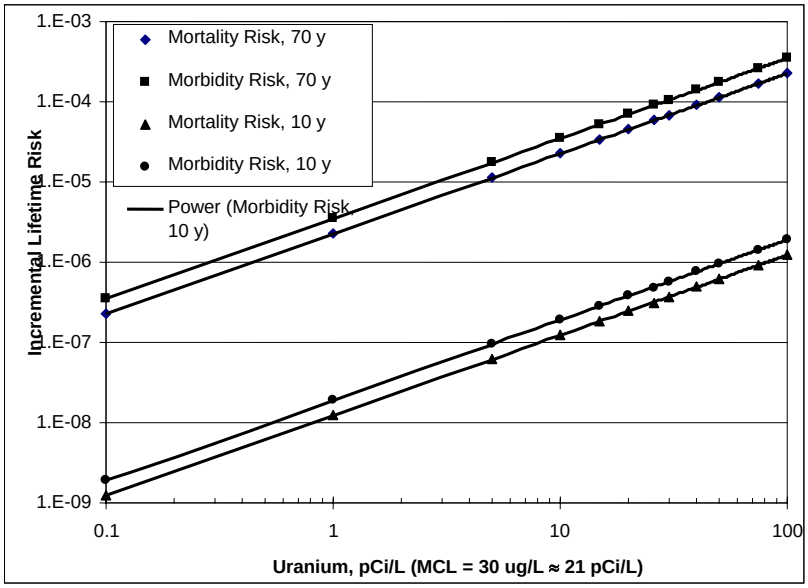


Table 3.11 estimates the potential lifetime cancer risk from radionuclides in the shallow Yazzie-312 Mine pit water (Panacea 2002), at concentrations measured before the pit was remediated. At these levels, long-term consumption of drinking water containing the radionuclides would be a significant health risk, but shorter-term exposures would not.

Table 3.11. Lifetime Risks Estimated from Drinking Unremediated Yazzie-312 Mine Pit Water

While long-term consumption of drinking pit water from the Yazzie-312 Mine posed a significant health risk, shorter-term exposures would not.

Contaminant	Average Concentration	Exposure Duration			
		70 Years, 365 Days/Year		10 Years, 14 Days/Year	
		Mortality	Morbidity	Mortality	Morbidity
Total Radium	2.3 pCi/L	7×10^{-5}	9×10^{-5}	4×10^{-7}	5×10^{-7}
Total Uranium	173 pCi/L	4×10^{-4}	6×10^{-4}	2×10^{-6}	3×10^{-6}
Gross alpha[a]	84 pCi/L	5×10^{-4}	8×10^{-4}	3×10^{-6}	4×10^{-6}
Total Risk	--	9.7×10^{-4}	1.5×10^{-4}	5.4×10^{-6}	7.5×10^{-6}

[a] Without uranium and radon

Note: Other periods of exposure may be of interest, such as a 30-year period, often used in Superfund calculations. Since the relationship between concentration and risk is linear, a ratio can be used to calculate risks at different time periods. To estimate the risk for 30 years of exposure, divide the 70-year risk number by 2.33 (70 y/30 y). Arsenic was measured in the pit at

an average concentration of 55 µg/L, just over the MCL in effect in 2005 and five times higher than the 10 µg/L MCL that became effective in 2006. In calculating the risk from arsenic in the water, the following equation and defaults from Superfund Risk Assessment Guidance (U.S.EPA 1989b) were used:

Target lifetime cancer risk, TR = (SF_O * C * IRW * EF * ED) / (BW * AT * 365 days/year)

where:

SF_O	=	Slope factor for arsenic $(mg/kg\text{-}d)^{-1}$	1.5
C	=	Pit water arsenic concentration (mg/L)	0.055
IRW	=	Daily water ingestion rate (L/day)	2
EF	=	Exposure frequency (days/y)	350
ED	=	Exposure duration (years)	30
BW	=	Body weight (kg)	70
AT	=	Averaging time (years)	70

Using the default values listed above, we estimate the risk for drinking arsenic to be about 1 in 1,000, assuming 30 years of exposure (1.5 * 0.055 * 2 * 350 * 30 /[70 * 70 * 365] = 1,732.5 / 1,788,500 = $9.7 \times 10^{-4} \sim = 1 \times 10^{-3}$). For an exposure of 14 days/year for 10 years, the risk estimate is 1.3×10^{-5} or $\sim = 1 \times 10^{-5}$. Thus, the pit water at the Yazzie-312 Mine could have posed a high risk from both radionuclides and arsenic, if the water were consumed over long periods of time.

The 1983 EPA report to Congress also reported Wyoming and New Mexico field studies of trace elements and radionuclides from inactive mining areas at off-site locations (U.S. EPA 1983c). In both cases, precipitation is seasonal and adjacent streams are dry much of the year. The general observations were that concentrations of Ra-226 and U-238 from spoils piles and in stream channels decreased rapidly with distance from the mines. However, the migration of trace metals did not show as distinct a trend. The transport processes were believed to be wind erosion and sheet erosion from cloudbursts, and they appeared to move mine spoils material up to 2,000 feet (~600 m) in 10 years. Preliminary data from recent sampling by Burghardt (2003) at several uranium mines have identified decreasing uranium and arsenic concentrations from the toe of the pile to background levels within several hundred meters.

3.6 Migration of Uranium Waste into Groundwater

Chemical and physical processes can enhance or retard the movement of the contaminants into and through an aquifer. Infiltration of water into soil is an example of a physical process, while partitioning of the contaminant between the soil and water is an important chemical process (which gives rise to the soil–water distribution coefficient, K_d). On the Colorado Plateau, where many uranium mines are located, the dry climate limits the available water for transporting the radionuclides and for drinking. Much of the precipitation is lost to evapotranspiration, thus limiting the infiltration, although high intensity precipitation events may contribute to increased infiltration at times. In large parts of the Colorado Plateau, the only usable water available in quantity is from groundwater (U.S. EPA 1983b), particularly in relatively deep confined aquifers, but near-surface aquifers are present in some areas. The impact of small surface

uranium mines on most of the groundwater in this region is expected to be minimal. As an example described in more detail below, drilling and sample analysis of a groundwater aquifer located under theYazzie-312 pit lake found no direct communication or correlation of water chemistry with the overlying lake (Panacea 2002). However, underground mines that intersect an aquifer could contaminate the aquifer, as could large surface mines with deep pits. Also, in areas with greater precipitation or near-surface unconfined aquifers, including higher elevations in the Colorado Plateau, contaminated water may more easily reach the groundwater, where it could be transported and pose significant cancer risks to people who obtain their drinking water from the aquifer.

3.7 Mobility of Uranium and Radium through Groundwater

EPA's *Soil Screening Guidance for Radionuclides* is one method that can be used to conservatively estimate the potential for a radionuclide to move into groundwater and to develop a general understanding of the resulting health risks (U.S. EPA 1996a, 2000b). This approach, which is modified as site-specific conditions are understood, relies on the use of distribution coefficients. This generalized approach is useful for this scoping analysis, since many potentially different site conditions and parameters would need to be considered otherwise. Indeed, for an individual site it is important to gather site-specific information before decisions are made for the particular site. A goal in establishing a soil contaminant concentration is to avoid future contamination of groundwater above the maximum concentration level (MCL) established for the contaminant in potable water. This general approach is also applicable to metals, but the focus here is on key radionuclides.

In calculating the SSL, in pCi/g, for groundwater the equation is:[6]

$$C_t = C_w * (1 \times 10^{-3}) * (K_d + \theta_w/\rho_b)$$

where:

C_t	=	Total concentration in soil (pCi/g)	1.5
C_w	=	Target concentration in leachate (pCi/L)	element-specific, ~20 pCi/L for uranium
1×10^{-3}	=	Conversion factor (kg/g)	-
K_d	=	Soil-water partition coefficient (mL/g)	Element-specific
θ_w	=	Water-filled porosity (unitless)	0.3
ρ_b	=	Dry soil bulk density (kg/L)	1.5

C_w, the target concentration in the leachate, is derived by multiplying the MCL by a dilution factor of 20[7], the soil-water partition coefficient is specific to the contaminant of concern, and default values are used for the unitless water-filled porosity, and the dry soil bulk density (U.S. EPA 2000b, Equation 6).

[6] There are additional variations on this equation, including a mass-limit version that includes infiltration. More detail on this and alternative ground-water transport models are discussed in the EPA Soil Screening Guidance Technical Background Document (U.S. EPA 1996b).

[7] Default value from U.S. EPA 1996b, Part 2.

The SSL generally corresponds to a risk of 1×10^{-6}, and the actual cleanup goal is modified from there; however, for groundwater it is based on achieving the MCL. Tables 3.13 and 3.15 provide the soil screening levels for uranium and radium, respectively, assuming varying soil-water partition coefficients with the target concentration as the MCL. Thus, C_w= 600 µg/L of uranium for an MCL of 30 µg/L [or ~20 pCi/L using the uranium specific conversion 0.67 pCi/µg (U.S. EPA 2000d)].

In using this equation, it is important to note the following simplifying assumptions applied in the Soil Screening Guidance methodology. The assumption that soil contamination extends from the surface to the water table adds a conservative element to the equations, since this condition would be uncommon in the Colorado Plateau, where the depth to water can be tens of meters or more, precipitation is limited, and the aquifer is typically confined. However, in other areas where water is near the surface, this list of simplifying assumptions may not be as conservative.

Simplifying Assumptions for the Migration of Radionuclides to Groundwater

- The source is infinite (i.e., steady-state concentrations will be maintained in groundwater).
- Contaminants are uniformly distributed throughout the zone of contamination.
- Soil contamination extends from the surface to the water table (i.e., adsorption sites are filled in the unsaturated zone beneath the area of contamination).
- There is no chemical or biological degradation in the unsaturated zone.
- Equations do not account for radioactive decay.
- Equilibrium soil/water partitioning is instantaneous and linear in the contaminated soil.
- The receptor well is at the edge of the source (i.e., there is no dilution from recharge downgradient of the site) and is screened within the plume.
- The aquifer is unconsolidated and unconfined (surficial).
- Aquifer properties are homogeneous and isotropic.
- Chelating or complexing agents are not present.
- No facilitated transport (e.g., colloidal transport) of inorganic contaminants occurs in the aquifer.

Source: U.S. EPA 2000b.

3.7.1 Uranium

Depending on the environmental conditions, uranium can be mobile enough to leach into and move through groundwater, especially in the oxidizing conditions at low pH levels that are present in acid mine drainage. Uranium tends to be relatively immobile under reducing conditions. Table 3.12 illustrates the range of uranium mobility as a function of pH, and Table 3.13 indicates the soil screening level above background needed to achieve the MCL of 30 µg/L. A higher partition coefficient (K_d) means that the movement of uranium would be slower relative

to the movement of water. In the White King monitoring wells, the ore pile area pH was between 4.2 and 6.9, the mine spoil area pH was between 5.6 and 7.0, and the pH in unaffected areas in the vicinity was between 6.3 and 7.7 (USFS 1991).

Although no K_ds were calculated at the White King site, no downgradient uranium was detected, even though pore water samples in the protore stockpile were over 27,000 pCi/L (10^6 Bq/m^3). The overburden stockpile activities were less than 18 pCi/L (670 Bq/m^3), with a concentration of only 75 pCi/L (2775 Bq/m^3) immediately under the protore stockpile (Weston 1997). Thus, the uranium appears to be immobile, with a high K_d, at this site. Radium, in the form of radium sulfate, apparently had not migrated at all. In the 1983 EPA report to Congress, soil profiles obtained at a uranium mine in Wyoming also showed no downward migration of radionuclides (U.S. EPA 1983c).

Table 3.12. Look-up Table for Estimated Range of K_d Values for Uranium Based on pH

K_d (mL/g)	pH Levels							
	3	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Minimum	< 1	0.4	25	100	63	0.4	< 1	< 1
Maximum	32	5,000	160,000	1,000,000	630,000	250,000	7,900	5

Source: U.S. EPA 1999

Table 3.13. Soil Screening Values for Uranium as a Function of K_d
Uranium (MCL = 30 µg/L ~ 20 pCi/L ~ 760 Bq/m^3)

Target Concentration	Assumed Partition Coefficient (K_d) (L/kg)				
	1	10	25	50	100
	Soil Screening Values (pCi/g Above Background Levels) Resulting in Groundwater Target Concentration Using the Groundwater Soil Screening Approach				
30 µg/L[8]	0.5	4	10	21	41

In contrast to the White King/Lucky Lass site, at Midnite Mine the groundwater indicator map from preliminary investigation work (U.S. EPA 2003c) plots concentration exceedances for shallow and deep wells. Uranium and other metals have been detected in several of the downgradient alluvial wells and in a couple of shallow bedrock aquifer wells adjacent to a pit and a stockpile.

The Yazzie-312 Mine has no near-surface water table because of the dry Arizona climate. There is a confined aquifer at 105 feet (32 m) below ground surface in the southern part of the site in a sand-and-gravel unit, with a static water level of 27 feet (8.2 m) below ground surface. This unit was thought to be part of a former alluvial channel, since no water was found in another well north of the mine. Since only 2.6 pCi/L (96 Bq/m^3) uranium was in the well water while 173 pCi/L (6,400 Bq/m^3) was in the pit water, the interpretation is that the pit water is not contributing to the radionuclide concentration in the aquifer. On the other hand, Longsworth

[8] Conversion factor for naturally occurring uranium from µg/L to pCi/L (U.S. EPA 2000d): 0.67 pCi/µg

(1994) measured shallow groundwater in the vicinity of mines in the Monument Valley area of Arizona and Utah with significant levels of uranium, radium, and radon (up to 14,000 pCi/L U-238, 110 pCi/L Ra-226, and 250,000 pCi/L of Rn-222). The impact on groundwater from surface and near-surface uranium mines would appear to be highly dependent on local geological and hydrological conditions.

3.7.2 Uranium Plume Migration

In a review of uranium plumes in groundwater from natural analogues, in-situ leaching operations, and uranium mill tailings sites, Colon et al. (2001) identified a "clear and reasonably consistent picture of [uranium] plume behavior" in which plumes appear to reach a steady-state condition: the plumes rarely exceed 1.25 miles (2 km) in length and exhibit natural attenuation under different circumstances, with the low-pH in-situ leaching process contributing to the greatest plume distances. Of the natural analogues, the maximum axial[9] plume length was 1 mile (1.6 km) from the Oklo uranium deposit that acted as a natural reactor ~ 2 billion years ago. If this attenuation were to hold true at uranium mines, the distance of influence on uranium transport from an abandoned uranium mine (in the absence of added acids) in the groundwater could be less than 1.25 miles (2 km). Fracture networks, facilitated (colloid) transport, or other site-specific characteristics may act to limit this attenuation.

3.7.3 Radium

Information on radium soil-water distribution coefficients is less common, but radium K_d values that span a large range are found in the literature. U.S. EPA (2004) cautions the reader that many of the high values are suspect, because they may be the result of co-precipitation of radium with other ionic species, rather than absorption of radium itself. One EPA study indicates that very little radium is available for transport, and strong acids were necessary to extract the radium (DeLaune et al. 1996). Tachi et al. (2001) calculated K_ds of 10^2–10^4 mL/g for bentonite clays with a dependence on pH. U.S. EPA (2004) mentions one study of four sandy soils from Utah with a range of radium K_d values from 214 to 354 ml/g for pH that varies between 7.6 and 8.0. EPA (2004) confirms the paucity of K_d data, stating: "Development of K_d look-up tables for radium is not possible given the minimal number of adsorption studies." U.S. EPA (ibid.) then goes on to suggest the use of the K_d table for strontium presented by U.S. EPA (1999, Vol. 2) as general guidance for radium. This table is reproduced as Table 3.14. Table 3.15 provides SSLs for radium as a function of K_d for a range of K_ds from 1 to 500.

[9] Along the center line of the contamination where the greatest concentration would be expected.

Table 3.14. Relationship Between pH Levels and Strontium Mobility as a Surrogate for Radium

Look-up table for estimated range of K_d values for strontium as a surrogate for radium based on clay content and pH. [Tabulated values pertain to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength (< 0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.]

	Soil Clay Content (wt.%)								
	< 4%			**4 - 20%**			**20 - 60%**		
	pH			**pH**			**pH**		
K_d (ml/g)	**< 5**	**5 - 8**	**8 – 10**	**< 5**	**5 – 8**	**8 - 10**	**< 5**	**5 - 8**	**8 – 10**
Minimum	1	2	3	10	15	20	100	200	300
Maximum	40	60	120	150	200	300	1,500	1,600	1,700

Table 3.15. Soil Screening Values for Radium as a Function of K_d

Radium (MCL = 5 pCi/L [185 Bq/m^3])

Target Concentration	**Assumed Partition Coefficient**					
	1	10	25	50	100	500
	Soil Screening Levels Concentration Values (pCi/g) Above Background Resulting in Target Groundwater Concentration					
5 pCi/L	0.12	1.0	2.5	5	10	50

3.7.4 Potential for Groundwater Infiltration and Contamination

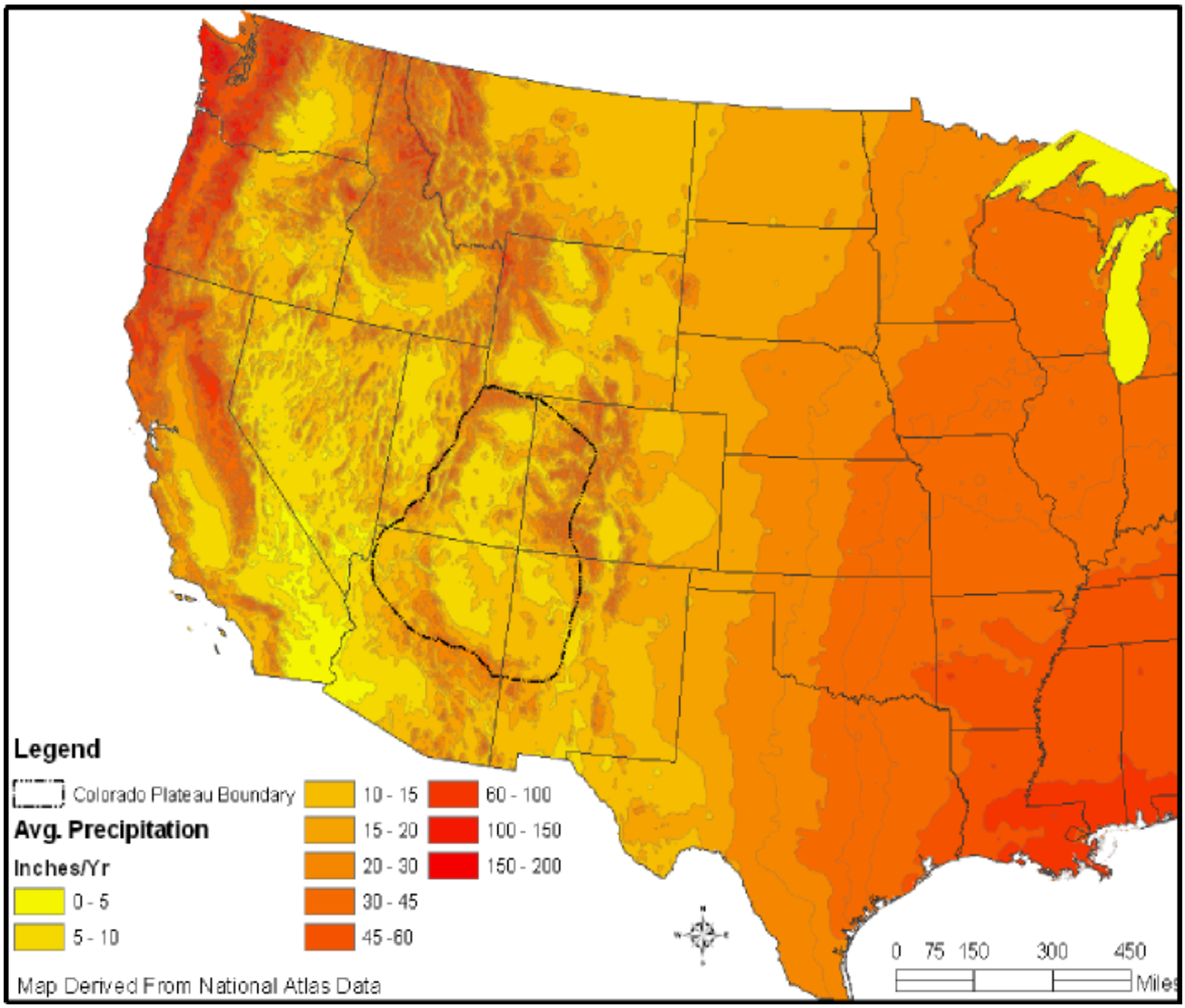
From Figure 3.11 below, the general annual precipitation range for the Colorado Plateau area is 5-15 inches (13-38 cm). This area also has high evapotranspiration rates. The 1983 EPA Report to Congress (U.S. EPA 1983a, b, and c) estimated that about 97 percent of the precipitation was lost to evapotranspiration. Evaporation tables indicate that the general area experiences greater than 75 inches (190 cm) of evapotranspiration annually. Thus, very little precipitation infiltrates. The Maxey-Eakin empirical method for estimating recharge in the southwest (Maxey and Eakin, 1949) assumes recharge would be zero if precipitation was less than 8 inches (20.3 cm/y), and only 3% if precipitation was between 8-12 inches (20.3-30.4 cm/y). Flint et al. (2002) modified this for areas of shallow soil, so that the minimum precipitation threshold for recharge to occur was 10 cm/y.

Thus, for 15 inches/y (38.1 cm/y) of precipitation, or the maximum of the range of annual precipitation in the Colorado Plateau, the average recharge would be ~0.5 inches/y (1.1 cm/y). If this average value is assumed to be a simple velocity estimate to an aquifer and assuming no retardation, it would take hundreds of years or longer to reach an aquifer at depth. Doubling the velocity (i.e., infiltration rate) would reduce the travel time by one-half. Thus, abandoned uranium mines in the proximity of shallow aquifers may contaminate the aquifer within tens of years, but this process would take longer for the deeper mines. This simple analysis suggests that the abandoned uranium mines that don't intersect aquifers pose a greater immediate risk from surface pathways and use than from the groundwater pathway.

Site-specific characteristics such as precipitation, depth to water, soil characteristics (e.g., permeability or pH), or presence or absence of fractures, would dictate the actual infiltration, potential recharge and potential to contaminate an aquifer, and the time frame over which such contamination could occur. Once the radionuclide enters an aquifer, its transport would be dependent on several site-specific factors—including the aquifer's permeability, water velocity, and chemistry (e.g., pH)—that affect retardation. Although much of the discussion in this section has focused on radionuclides, similar concepts apply if metals are also present at a site.

Figure 3.11. Average Precipitation (inches/year) for the Western United States

The Colorado Plateau, where many of the uranium mines are or were located, is a region characterized, in general, by low precipitation and high evapotranspiration.



3.8 Consideration of Multiple Exposure Pathways

The fundamental criterion for applying the SSLs to a single exposure pathway is that $C_i \leq SSL_{i,k}$; that is, the concentration of pollutant i, C_i, is less than the SSL for pathway k, $SSK_{i,k}$. This implies that, for multiple exposure pathways, the SSL should be reduced to account for additive contributions to the pollutant intake from these additional pathways such that:

$$C_i/SSL_{i,1} + C_i/SSL_{i,2} + C_i/SSL_{i,3} + \ldots C_i/SSL_{i,k} \leq 1.0.$$

Dividing both sides by the concentration term C_i and inverting the equation gives:

$$C_i \leq 1/[1/SSL_{i,1} + 1/SSL_{i,2} + 1/SSL_{i,3} + \ldots 1/SSL_{i,k}].$$

The term on the right side may be viewed as a multi-pathway SSL. Tables 3.16, 3.17, and 3.18 show the application of this methodology to the external exposure, soil ingestion, and inhalation of fugitive dust pathways for the on-site exposure scenario discussed earlier in this chapter. The SSLs for external exposure and soil ingestion are listed in Tables 3.2 – 3.7. The calculation of SSLs for the inhalation of fugitive dust is discussed in the text. The risk from recreational use of off-road vehicles is not included, because the riders of these vehicles will not, in general, be the same individuals exposed in the other on-site scenarios. Likewise, the consumption of drinking water from a well would affect residents on or off the site many years in the future, after the activity has percolated into the groundwater. These would not be the same individuals exposed to the radioactivity in the surface soil due to recreational use of the site at the present time. However, for a particular site the risk from drinking surface or near-surface water could be added to risks from the other pathways. However, risk estimates conducted for this chapter indicate that the risks in the recreational scenario from external exposure are much greater than from drinking water contaminated with radionuclides.

A comparison of the multi-pathway SSLs for Ra-226 listed in Table 3.16 with the SSLs for external exposure shown in Table 3.2 shows a difference of about 1.75%; thus, the external exposure pathway for this nuclide and its progeny is dominant, and the other pathways make minor contributions to the total risk. A similar comparison for Th-232, using the SSLs in Tables 3.3 and 3.17 shows an even smaller difference—about 1.2%—indicating that the external exposure pathway is dominant for this nuclide and its progeny. This is not the case for natural uranium; although external exposure constitutes over 86% of the risk, soil ingestion makes a significant contribution. The inhalation of fugitive dust makes a minor contribution. Figures 3.12 through 3.14 portray the same data in graphical form.

Table 3.16. Multi-pathway Soil Screening Levels for Ra-226

Exposure Frequency (days/year)	Target Lifetime Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Ra-226 (pCi/g)					
1	21,116	4,223	2,112	422	211	42.2
14	1,508	302	151	30.2	15.1	3.02
30	704	141	70.4	14.1	7.04	1.41
52	406	81.2	40.6	8.12	4.06	0.812
140	151	30.2	15.1	3.02	1.51	0.302
350	60.3	12.1	6.03	1.21	0.603	0.121

Table 3.17. Multi-pathway Soil Screening Levels for Th-232

Exposure Frequency (days/year)	Target Lifetime Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Th-232 (pCi/g)					
1	14,674	2,935	1,467	293	146.7	29.3
14	1,048	210	105	21	10.5	2.10
30	489	97.8	48.9	9.78	4.89	0.978
52	282	56.4	28.2	5.64	2.82	0.564
140	105	21.0	10.5	2.10	1.05	0.210
350	41.9	8.38	4.19	0.838	0.419	0.0838

Table 3.18. Multi-pathway Soil Screening Levels for Natural Uranium

Exposure Frequency (days/year)	Target Lifetime Cancer Risk					
	5×10^{-4}	1×10^{-4}	5×10^{-5}	1×10^{-5}	5×10^{-6}	1×10^{-6}
	Concentration of Natural Uranium (pCi/g U-238)					
1	751,392	150,278	75,139	15,028	7,514	1,503
14	53,671	10,734	5,367	1,073	537	107
30	25,046	5,009	2,505	501	250	50.1
52	14,450	2,890	1,445	289	144	28.9
140	5,367	1,073	537	107	53.7	10.7
350	2,147	429	215	42.9	21.5	4.29

Figure 3.12. Multi-pathway Soil Screening Levels for Ra-226

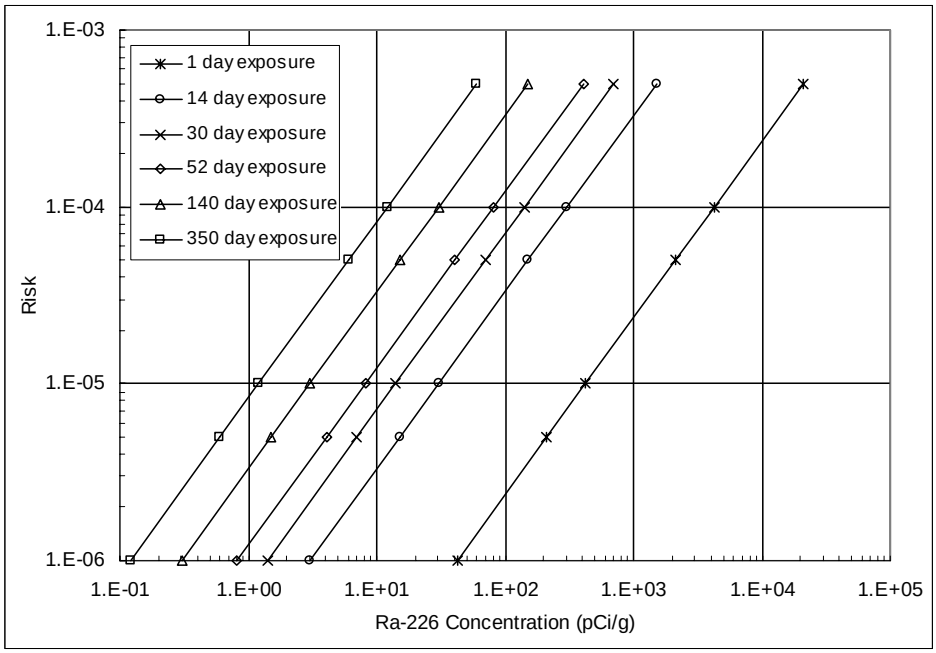


Figure 3.13. Multi-pathway Soil Screening Levels for Th-232

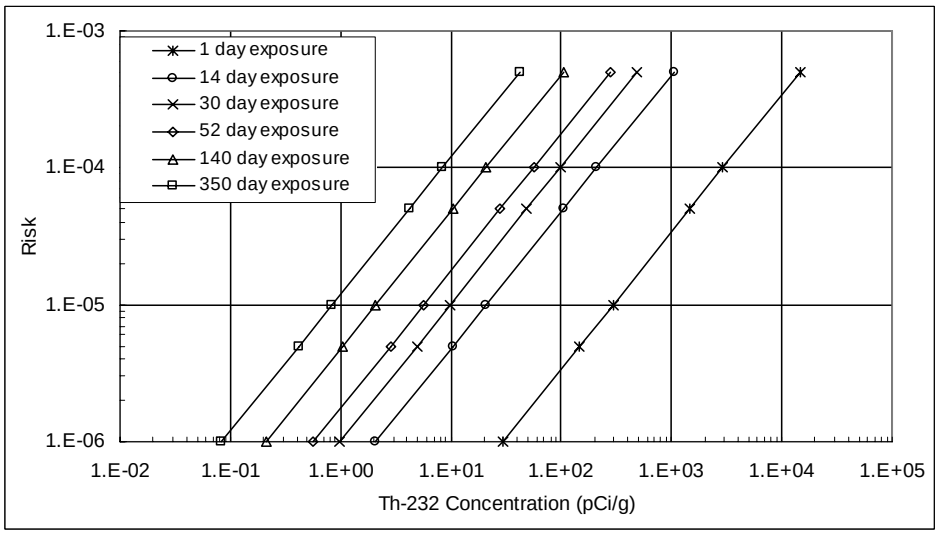
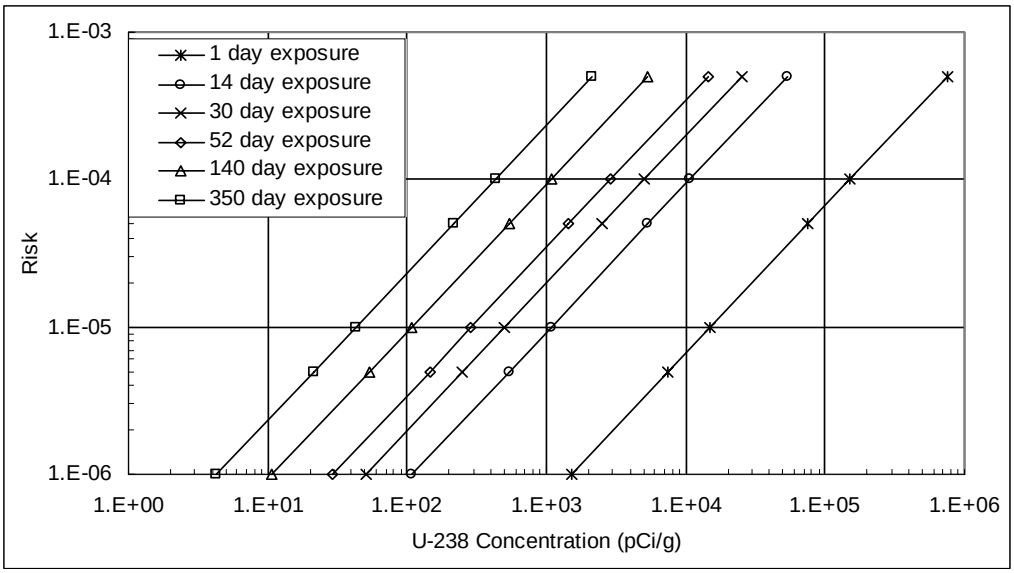


Figure 3.14. Multi-pathway Soil Screening Levels for U-238



4.0 RISK FROM URANIUM MINING WASTE IN BUILDING MATERIALS

In general, building materials contain low levels of radioactivity. For example, the range of natural uranium concentrations may average as low as about 0.5 ppm (0.34 pCi/g or 13 Bq/kg) total uranium activity in sandstone building materials to as high as 5 ppm (3.4 pCi/g or 130 Bq/kg) in granitic building materials. Concrete and brick buildings are estimated to contribute an average of about 10 mrem (0.1 mSv) annual effective dose equivalent (NCRP 1987) to the average person's background exposure to radiation. However, exceptions can occur to this generalization, especially in buildings constructed with materials containing uranium TENORM mine wastes. In the Grand Junction, Colorado area, thousands of homes and properties were constructed using uranium mill tailings (U.S. EPA 1983a, b, c) in the past as a source of construction sand, gravel, and clays. However, a number of homes have also been built with materials that have been attributed to "uranium ore" that are not considered to be mill tailings. In a 1972 EPA and Atomic Energy Commission (AEC) survey intended to locate building materials contaminated with mill tailings, 537 buildings were identified where uranium ore may have been the source of gamma-ray exposure anomalies (U.S. EPA 1983b):

> *We do not know to what extent the wastes from uranium mines have been removed from mining sites and used in local and nearby communities. However, while surveying in 1972 for locations with higher-than-normal gamma radiation in the Western States to locate uranium mill tailings used in local communities, EPA and AEC identified more than 500 locations where "uranium ore" was believed to be the source of the elevated gamma radiation. The specific type of ore (mill-grade, sub-ore, low-grade waste rock) was not determined as this was beyond the scope of the survey. At some locations, however, surveyors attempted to characterize the ore by using such terms as "ore spillage," "ore specimens," "low-grade crushed ore," or "mine waste dump material." Some locations were identified as sites of former ore-buying stations* [U.S. EPA 1973].

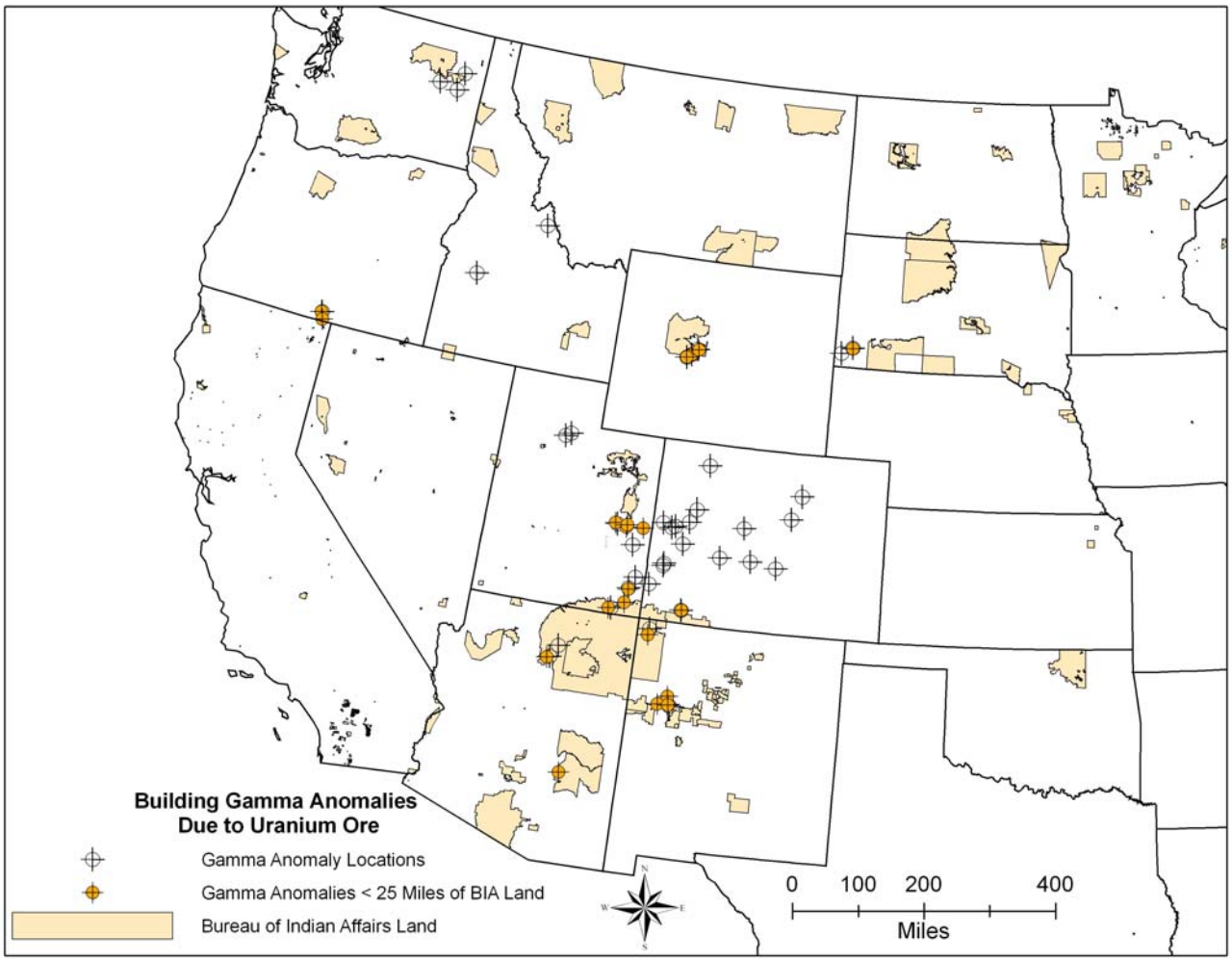
Since it is unlikely that valuable mill-grade ore would have been widely available for off-site use, we suspect that uranium mine waste (perhaps protore) may be the source of the elevated gamma radiation levels at many of the locations where large quantities of ore material are present.

About three-fourths of the 537 buildings were in Colorado and Utah, with the rest distributed among several other states. Figure 4.1 identifies the localities from the 1972 survey that had at least one building thought to have used "uranium ore" construction materials. Many of these same localities also had additional anomalies attributed to either a radioactive source or natural radioactivity. The original report that discusses the survey is unavailable, so it is not possible at this time to determine the basis used for the attribution of the cause. Of the 53 localities with at least one anomaly attributed to uranium ore, 20 are on or within approximately 25 miles (40 km) of Bureau of Indian Affairs (BIA) Reservations. Without knowing the design of the study, it is not possible to determine the statistical significance of the survey. Nevertheless, the survey does indicate the potential problem of contaminated buildings in uranium mining areas, especially on and around Tribal lands. EPA has provided support to Tribal authorities since that time to identify buildings on Tribal lands constructed with uranium mine wastes.

Contaminated buildings are among the mine waste issues that have been publicized regarding the Navajo Nation. Although not specifically addressed herein, anecdotal information is amenable to the methods and models for dose and risk estimates contained in this report. For mattresses placed directly on a contaminated slab (reported in the *Los Angeles Times* on November 19, 2006), a geometrical variation would be applied to the analysis that follows. (As an example involving other exposure pathways, for children who "dug caves in piles of mill tailings and played in the spent mines," variations in the recreational scenario of Chapter 3 would be applied.)

A specific case of the potential problem on Tribal lands is illustrated by hogans with elevated radioactivity found in the Monument Valley area of Utah. In April 2001, EPA razed and removed a building that had been used as a hogan (sacred home) by a Navajo family. As shown in the photograph in Figure 4.2, the hogan was a small, one-room round structure with a concrete slab for a floor and stucco walls, although the building originally had a dirt floor. Figure 4.3 is a picture of another house taken from the vantage point of uranium mine workings.

Figure 4-1. Locations of Building Gamma Anomalies Due to Uranium Ore from 1973 EPA-AEC Study

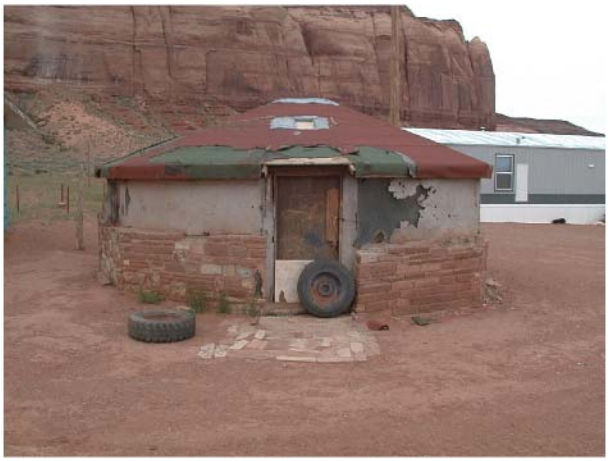


Source: U.S. EPA.

Short-term gamma-ray exposure rates and radon concentrations were measured prior to the demolition of the hogan (Sowder et al. 2001). Radiation exposures were between 370 µR/h and 600 µR/h. This is equivalent to doses in air of 325–525 µrad/h (~3–5 µGy/h). (Typical indoor background dose rates are in the range of 1.2–16 µrad/h [12–160 nGy/h]). Several stones in the hogan exhibited levels of 1,000 µR/hour on contact. Short-term indoor radon measurements using multiple methods averaged 50–90 pCi/L (1,850–3,300 Bq/m^3) under pseudo-closed conditions. Outdoor exposure rates as high as 75 µR /hour at 3.3 feet (1 m) from the structure were observed. Stones used in the exterior construction produced exposure rates of 500–1,000 µR/hour. Inspection of the floor after demolition revealed that uranium ore had been used as aggregate for the concrete. Apparently, the source of the sand and stones in the building material was a nearby uranium mine or outcrop adjacent to the mine (possibly the Skylight Mine). Other possibilities for the material include mine-waste material debris piles alongside roads, such as the one in Figure 4.4, which is on Navajo Nation land. Readily available construction materials, including clay, sand, gravel, cobbles, and boulders in above-ground piles, make them attractive for houses, stoves, chimneys, and barbecues, and for stucco, cement for log houses, driveways, walkways, and fill dirt.

Figure 4-2. Monument Valley Navajo Hogan

Monument Valley Navajo family hogan razed due to high gamma readings. Note the talus in back, much of which originated from Skylight Mine on top of the mesa directly above.



Photograph by Andrew Sowder (U.S. EPA)

Figure 4-3. Navajo Home in Proximity to Uranium Mine

This picture is another example of the proximity of some homes to uranium mines. A New Mexico mine, now reclaimed, lies in the foreground of the picture, while the house in the background was originally constructed with mine waste but has since be reconstructed to remove the contaminated material.

Photograph by Loren Setlow (U.S. EPA)

Figure 4-4. Uranium Mine Debris Pile

Debris pile of uranium mine wastes just off a road on Navajo Nation land.

Photograph by Andrew Sowder, (U.S. EPA)

4.1 Building Materials Analysis

Given that some homes incorporate uranium mine waste building material, the question arises as to the radium and uranium concentrations in these materials that would result in exposure levels of concern. To identify potential gamma and radon exposures over a range of uranium and

radium concentrations from contaminated concrete used as building materials for the floor and each wall, we used the RESRAD-BUILD 3.21 computer code (Yu et al. 2001).

The building we used for our modeling was based on the concrete Monument Valley Navajo hogan. The building modeled had one room with a floor area of 16.4 × 16.4 feet or 269 ft^2 (5 × 5 m or 25 m^2). Each wall is assumed to be 8.2 feet (2.5 m) high, 16.4 feet (5 m) long, with an area of 134 ft^2 (12.5 m^2) (Figure 4.5). Occupancy is assumed to be 70 percent for 365 days a year (NAS 1999). Since the calculations were scoping in nature, we used the RESRAD-BUILD default parameters. We assumed that the floors and walls were made of concrete, the radium and uranium concentrations were equal, and the receptor was at a height of 3.28 feet (1 m). However, RESRAD-BUILD calculates the contribution of the floor and the wall, so that the contribution from each part can be separated. The calculations assume no contribution from the soil beneath the concrete floor. The concrete was assumed to be 6 inches (15 cm) thick, with a density of 2.4 g/cm^3. Results are presented in doses, which are calculated by RESRAD-BUILD.

4.1.1 Results of Building Materials Analysis

From the modeling conducted using RESRAD-BUILD, we calculated doses from external exposures to U-238 and Ra-226 in full secular equilibrium with their short-lived progenies.[1] These doses are listed in Tables 4.1 and 4.2 and are presented graphically in Figures 4.6 and 4.7.

[1] This is somewhat different from the way uranium was characterized in the analyses presented in Chapter 3. In the latter case, all uranium isotopes were assumed to be present in proportion to their natural abundance, and all long-lived progenies except Ra-226 and its decay chain were included, whereas the analysis in this chapter addresses only U-238, the dominant isotope, and its short-lived progeny.

Figure 4-5. Navajo Hogan Building Model

This 3-D schematic of the Navajo hogan indicates the sources of exposure modeled, with the floor as source 1, and the walls as sources 2 - 5. The origin of the geometry is at the lower left-hand corner, where z represents the vertical extent of the room and x and y represent the lateral extent of the walls.

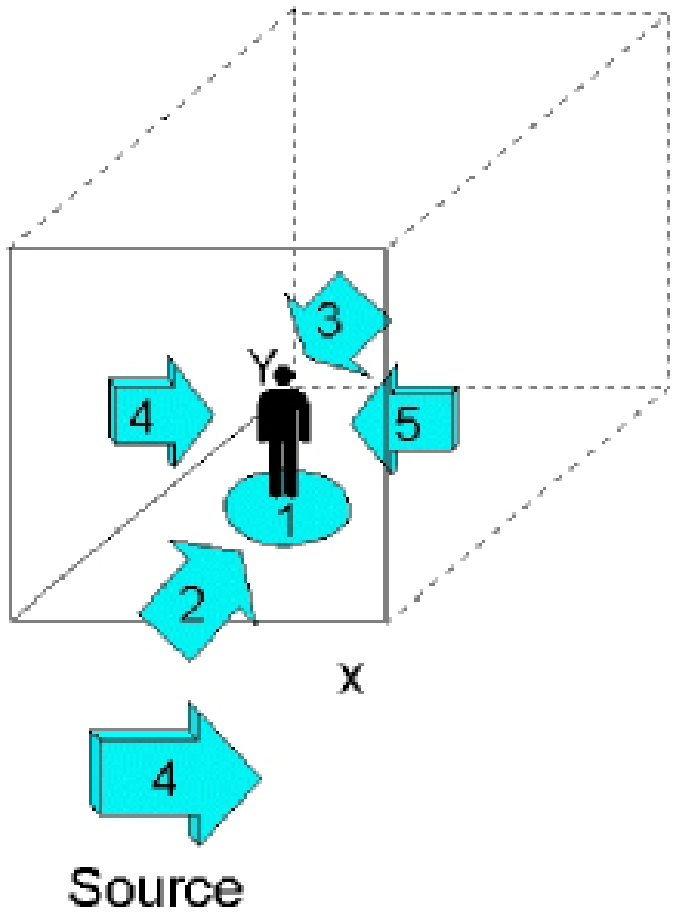


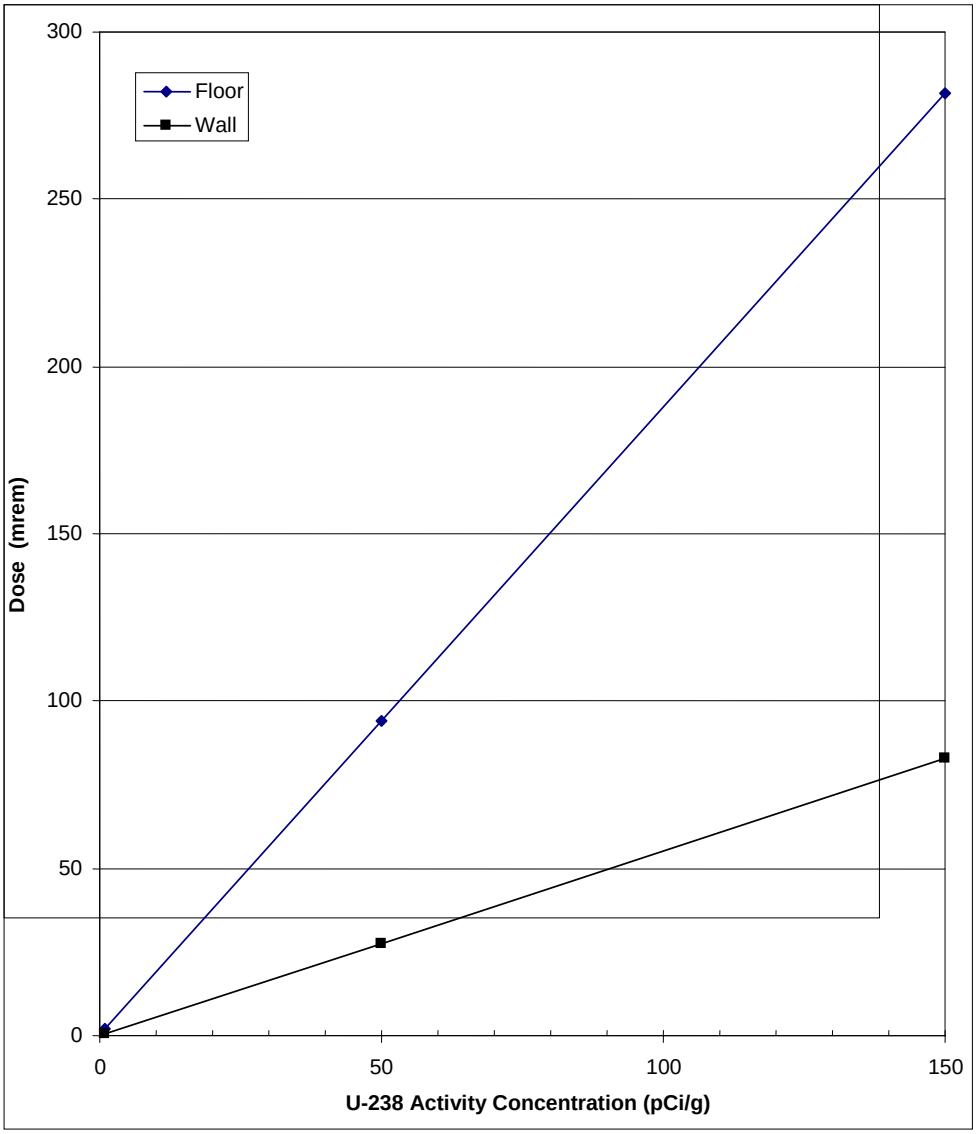
Table 4-1. Doses from 30 Years of External Exposure to U-238 in a Navajo Hogan

The dose from the floor is about equal to all of the walls combined.

Activity Concentration (pCi/g) (Bq/kg)	Dose from Floor (mrem) (mSv)	Dose from One Wall (mrem) (mSv)
1 (37)	1.88 (.02)	0.554 (.006)
50 (1850)	93.9 (.9)	27.7 (.3)
150 (5550)	282 (2.8)	83.1 (.8)

Figure 4-6. Doses from 30 Years of External Exposure to U-238 in a Navajo Hogan

The floor in the Navajo hogan contributed the most gamma exposure.



Doses are listed from external exposure to the floor and to a single wall to allow for an estimate of the dose if just a concrete slab is contaminated. The calculated dose from a single wall is between one-fourth and one-third the calculated dose from the floor. The total dose from the entire structure may be estimated by multiplying the dose from one wall by a factor of four and adding the result to the dose from the floor.

In order that the uranium in building materials could pose a significant risk from external exposure, the uranium concentrations in the building materials must be quite high relative to background concentrations. For example, for a dose of 300 mrem (3 mSv) from the uranium in the floor over a 30-year period, the U-238 activity would need to be about 180 pCi/g

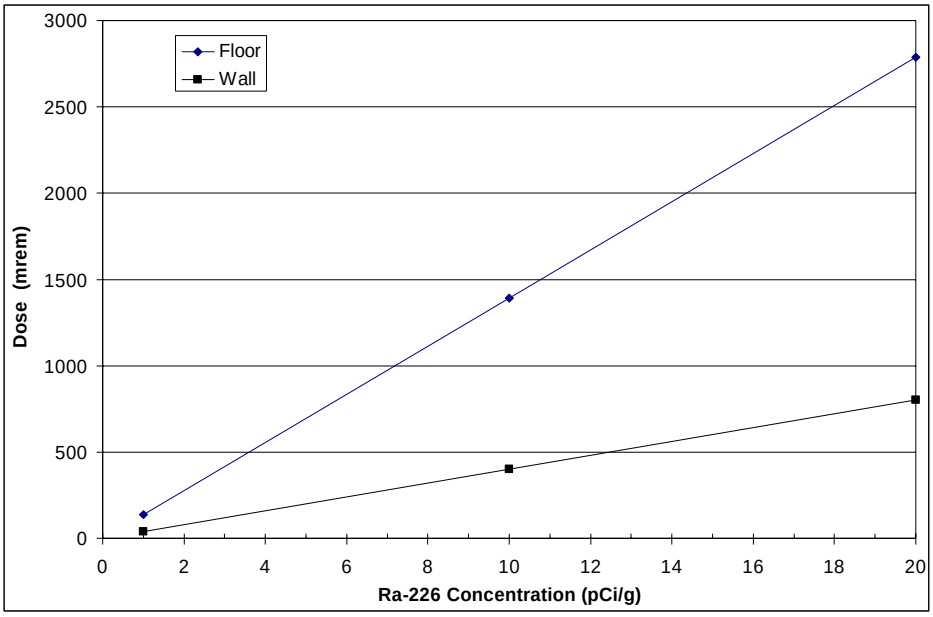
(6,660 Bq/kg or about 540 ppm). However, this level could be found in uranium overburden, and especially in protore.

Table 4-2. Doses from 30 years of External Exposure to Ra-226 in a Navajo Hogan

The dose from the floor is about equal to all of the walls combined.

Activity Concentration (pCi/g) (Bq/Kg)	Dose from Floor (mrem)(mSv)	Dose from One Wall (mrem)(mSv)
1 (37)	139 (0.14)	40 (.4)
10 (370)	1394 (14)	401 (4)
20 (740)	2787 (28)	801 (8)

Figure 4-7. Doses from 30 years of External Exposure to Ra-226 in a Navajo Hogan



Although U-238 would contribute to the overall radiation exposure, the Ra-226 in the mining waste materials is the more hazardous of the two radionuclides. A concentration of 1 pCi/g (37 Bq/kg) of Ra-226 in the floor is estimated to result in a dose of about 140 mrem (1.4 mSv) during 30 years of external exposure. According to the 1985 EPA report to Congress, most of the uranium mines sampled had Ra-226 concentrations of 20 pCi/g (740 Bq/kg) or more in the waste. If waste with this radium activity were incorporated into a concrete floor slab, it would result in a 30-year dose of about 2.8 rem (28 mSv). Figure 4.7 illustrates the relationship between Ra-226 concentrations and doses from external exposure calculated with RESRAD-BUILD.

The dose rate from the floor and four walls is approximately 50 µrem/h per pCi/g of Ra-226 (1.4×10^{-4} mSv/hr per Bq/kg). If the exposure rates measured in the Monument Valley Navajo hogan above were primarily from radium in the floor and walls, and the measurements were

made in the center of the hogan, we estimate that the materials in the hogan contained up to about 10 pCi/g of Ra-226 (370 Bq/kg).

In addition to direct radiation exposure, radon generation from radioactive decay could also contribute to risk posed by living in buildings constructed with uranium mine waste, depending on frequency of air exchange and other factors. As mentioned above in the Sowder et al. (2001) study of the hogan in Monument Valley, Utah, short-term indoor radon measurements using multiple methods averaged 50–90 pCi/L (1,850–3,300 Bq/m3) under pseudo-closed conditions. This greatly exceeds EPA's radon action level of 4 pCi/L (U.S. EPA 2004). However, studies of other houses constructed with uranium mine waste on Navajo Lands found many had much lower concentrations of radon, which may have been the result of construction methods and chimneys which allowed inside air to quickly exit the buildings (L. Setlow, U.S.EPA, personal communication, 2007)

4.2 Risk of Exposure of On-site Residents to Uranium Mining Waste

As described in Volume 1, the overburden and protore are typically left as piles, and consist of poorly sorted materials ranging from clay-sized fractions to boulders. Thus, it is not likely that the material would have a building located on it unless it has been flattened by erosion, was accessible from a higher elevation, or had been disposed off a hillside to create a terrace. In populated areas, however, it could be possible for the material to be spread out and a home subsequently built upon the leveled material. This scenario is included here as an upper bound on the potential risks from uranium mines, but it is not a focus of this scoping analysis because there are already guidelines for the amount of radium that is acceptable for Superfund remediation actions (U.S. EPA 1997a) and in the standards at 40 CFR 192 promulgated by EPA under the Uranium Mill Tailings Radiation Control Act (UMTRCA).[2]

The results of a study in Florida (U.S. EPA 1979) developed a relationship between Ra-226 in soil and indoor working levels (WL). The 1983 EPA report to Congress (U.S. EPA 1983b) references this document and assumed a similar relationship for a home built on uranium mine waste material. These data indicate that a concentration of 1 pCi/g (37 Bq/Kg) of Ra-226 in soil produces an indoor concentration of 1 pCi/L (0.03 Bq/L) of Rn-222, which is equal to 0.004 WL, assuming an equilibrium factor of 0.4 (UNSCEAR 2000). Thus, a concentration of 5 pCi/g (185 Bq/kg) of Ra-226 in the soil would produce an indoor radon concentration that is above the current recommended action level of 4 pCi/L (148 Bq/m^3).

The lifetime risk from the indoor radon decay products using current risk estimates is included in Table 4.3, along with the original estimate from 1983. Since the 1983 report was published, numerous studies have concluded that indoor radon concentrations are influenced by a

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[2] EPA regulations at 40 CFR 192 include limitations for radium and radon at UMTRCA sites: The disposal areas must be designed to limit releases of radon-222 from uranium byproduct materials to the atmosphere so as not to exceed an average release rate of 20 pCi/m^2/s. This requirement, however, applies only to a portion of a disposal site that contains a concentration of radium-226 that, as a result of uranium byproduct material, exceeds the background level by more than 5 pCi/g (185 Bq/Kg) averaged over the first 15 cm below the surface, or more than 15 pCi/g (555 Bq/Kg), averaged over 15 cm thick layers more than 15 cm below the surface.

combination of factors, including foundation slab integrity and permeability, indoor and soil pressure differentials, and the soil radium concentration. Thus, it is difficult to predict the indoor radon concentration based on soil parameters. However, modeling can provide a general indication of the radium/radon relationship.

Table 4-3. Estimated Lifetime Risk of Fatal Lung Cancer from Living on Contaminated Land

This table assumes an average individual is inside the home 75 percent of the time for the 1983 estimate, and 70 percent occupancy for the 2006 estimate. Because the estimate of risk per working level has increased from that used in 1983, and it is greater than the decrease in occupancy, the estimated cancer risk is higher in 2006.

Rasium-226 in Soil (pCi/g) (Bq/Kg)	Indoor Working Levels (WL)	Lifetime Risk of Fatal Lung Cancer	
		1983	2006*
5 (185)	0.02	0.025	0.029
10 (370)	0.04	0.050	0.059
20 (740)	0.08	0.100	0.117
30 (1110)	0.12	0.150	0.176

* The 2006 risk estimate is calculated using the equation presented in Chapter 1 of this volume, under the Applicability of 1983 Risk Estimates section.

Source: U.S. EPA 1983b.

Additional modeling was conducted using the RESRAD computer code, which embodies a one-dimensional multi-pathway model for residual radioactivity at sites (Yu et al. 2001). This code was chosen because of its applicability, widespread use, testing and review, and ease of use. Most of the RESRAD default values were chosen for this scoping analysis. For the Colorado Plateau on-site resident scenario, we initially assumed that consumption of groundwater was not an exposure pathway. We assumed a ventilation rate of 0.5/h, that the foundation was at the surface with no basement, and that 70 percent of the time was spent indoors and 30 percent outdoors. With these assumptions, the model predicted indoor radon and external exposure to direct penetrating radiation to be the major source of radiation exposure, with the indoor radon exposure higher than the external exposure. Most of the risk from living on contaminated materials is from the decay of indoor radon. When the ventilation rate is reduced to 0.25/h, the working levels increase (~0.031 WL for 5 pCi/g (185 Bq/Kg) radium). When we repeated the analysis with the drinking-water pathway included, using a value of ~82 feet (25 meters) for depth to the aquifer and conservative parameters, such as an evaporation coefficient of 0.5, and K_ds of 10 mL/g for uranium and radium, the indoor radon and external exposure pathways are still dominant. Pending any consideration of the food chain, which is of most potential importance for subsistence ranching and hunting, the risks from uranium are dwarfed by the risks posed by radium and radon.

Uranium mine wastes have the potential to create very high risks to an on-site resident, as indicated by this analysis and the analysis of the White King/Lucky Lass mine site. Ra-226 is the primary contributor to risk from the external exposure and indoor radon inhalation pathways. While the indoor radon concentrations and corresponding working levels resulting from a given concentration of Ra-226 depend on multiple factors, it is possible to estimate approximate relationships among these quantities.

5.0 POTENTIAL ECOLOGICAL IMPACTS FROM URANIUM MINES

This document has focused on the potential risks to humans from exposures to unreclaimed uranium mining materials. The potential effects on relevant ecosystems have not been addressed, because they are beyond the scope of this report. Although not analyzed here, ecosystem effects are briefly mentioned because of the potential importance of the topic in the consideration of unreclaimed uranium mines. Although the Superfund characterization process includes radionuclides in the ecological risk assessment and for some individual species, the lack of an accepted standard methodology for demonstrating protection of ecosystems from radiation makes the identification of potential effects due to uranium mining difficult. There is, however, a general framework for ecological risk assessment. As defined in the 1992 Framework for Ecological Risk Assessment (U.S. EPA 1992), an ecological risk assessment (ERA) is a process for evaluating the likelihood that adverse ecological effects may occur, or are occurring, as a result of exposure to one or more stressors.[1] This framework was applied in the Superfund guidance, *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments*, Interim Final (U.S. EPA 1997b).

Ecological risk assessment addresses two major elements, characterization of effects and characterization of exposure, which provide the focus for three primary phases of activities: problem formulation, analysis, and risk characterization (U.S. EPA 1998). In these three phases, the risk assessment process provides a way to develop, organize and present scientific information so that it is relevant to environmental decisions. Issues to consider are spatial and temporal, along with assessment endpoints, and whether it is the terrestrial or aquatic environments that are of concern (U.S. EPA 2000a). When conducted for a particular area such as a watershed, the ecological risk assessment process can be used to identify vulnerable and valued resources, prioritize data collection activity, and link human activities with their potential effects. However, a risk does not exist unless: (1) the stressor has the ability to cause one or more adverse effects, and (2) it co-occurs with or contacts an ecological component long enough and at a sufficient intensity to elicit the identified adverse effect (U.S. EPA 1997b). As discussed in this chapter, it is very possible that the stressors to the surrounding ecosystem may not be the radioactive materials, but rather the other hazardous constituents that may be associated with uranium mine sites.

Efforts are underway to extend the ecological risk assessment approach to radiation. In recent work, Jones et al. (2003) state that, "potentially susceptible receptors [to radiation] include vertebrates and terrestrial plants." EPA has no radiation dose standards for the protection of flora and fauna, but the Department of Energy (DOE) (Jones et al. 2003) has suggested levels of exposure for the protection for the following: natural populations of aquatic biota (1 rad d-1 or 10 mGy d-1), terrestrial plants (1 rad d-1 or 10 mGy d-1) and animals (0.1 rad d-1 or 1 mGy d-1).[2] The question remains whether these levels are indeed protective.

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[1] A "stressor" is any chemical, physical, or biological entity that can induce adverse effects on individuals, populations, communities, or ecosystems.

[2] 1 gray = 100 rad; thus 1 mGy = 0.001 Gy = 0.1 rad or 100 mrad.

DOE has recently issued a technical standard on applying these levels in the document *A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota* (U.S. DOE 2002). The graded screening approach uses three tiers, becoming progressively more rigorous and detailed: a scoping assessment, a screening ERA, and a more detailed ERA that uses site-specific information (Jones et al. 2003). As the tiers become more site-specific, the assumptions become less conservative. In the screening phase, this process uses biota concentration guides (BCGs) for water and sediment for evaluating aquatic systems, and water and soil for evaluating a terrestrial system. These BCGs are set "so that doses received by real biota exposed to such concentrations are not expected ever to exceed the biota dose limits" (Higley et al. 2003). The BCGs for aquatic and terrestrial systems are reproduced in Tables 5.1 and 5.2. The radiation levels found at some of the uranium mines where sub-ore and ore-grade materials have been left on site could exceed the levels identified by DOE, especially for radium.

Table 5-1. Biota Concentration Guides (BCGs) for Water and Sediment for Evaluation of an Aquatic System

Nuclide	BCG for Water		Organism Responsible for Limiting Dose in Water	BCG for Sediment		Organism Responsible for Limiting Dose in Water
	Bq/m³	*pCi/L*		*Bq/kg*	*pCi/g*	
^{226}Ra	2×10^2	5.4×10^0	Riparian[a] Animal	4×10^3	1.1×10^2	Riparian Animal
^{228}Ra	2×10^2	5.4×10^0	Riparian Animal	3×10^3	8.1×10^1	Riparian Animal
^{232}Th	1×10^4	2.7×10^2	Aquatic Animal	5×10^4	1.4×10^3	Riparian Animal
^{233}U	7×10^3	1.9×10^2	Aquatic Animal	2×10^5	5.4×10^3	Riparian Animal
^{234}U	7×10^3	1.9×10^2	Aquatic Animal	2×10^5	5.4×10^3	Riparian Animal
^{235}U	8×10^3	2.2×10^2	Aquatic Animal	1×10^5	2.7×10^3	Riparian Animal
^{238}U	8×10^3	2.2×10^2	Aquatic Animal	9×10^4	2.4×10^3	Riparian Animal

a A "Riparian Animal" is an animal that lives on a riverbank and hence spends time on land and in water, e.g., a muskrat.

Source: Reproduced from Higley et al. 2003.

Table 5-2. Biota Concentration Guides for Water and Soil for Evaluation of a Terrestrial System

Nuclide	BCG for Water		Organism Responsible for Limiting Dose in Water	BCG for Sediment		Organism Responsible for Limiting Dose in Water
	Bq/m³	pCi/L		Bq/kg	Bq/m³	
^{226}Ra	3×10^5	$8.1 \text{ x } 10^3$	Terrestrial Animal	$2 \text{ x } 10^3$	$5.4 \text{ X } 10^1$	Terrestrial Animal
^{228}Ra	3×10^5	8.1×10^3	Terrestrial Animal	2×10^3	5.4×10^1	Terrestrial Animal
^{232}Th	2×10^6	5.4×10^4	Terrestrial Animal	6×10^4	1.6×10^3	Terrestrial Animal
^{233}U	1×10^7	2.7×10^5	Terrestrial Animal	2×10^5	5.4×10^3	Terrestrial Animal
^{234}U	1×10^7	2.7×10^5	Terrestrial Animal	2×10^5	5.4×10^3	Terrestrial Animal
^{235}U	2×10^7	5.4×10^5	Terrestrial Animal	1×10^5	2.7×10^3	Terrestrial Animal
^{238}U	2×10^7	5.4×10^5	Terrestrial Animal	6×10^4	1.6×10^3	Terrestrial Animal

Note: 1 pCi/L = 37 Bq/m3, 1 pCi/g = 37 Bq/kg

Source: Reproduced from Higley et al. 2003.

5.1 Other Metals

There could be multiple stressors from uranium mining, especially in watersheds where a high density of uranium mines could have a larger effect than a single mine. The metals associated with uranium may cause adverse ecological effects, depending on the concentration and bioavailability. Arsenic, a human carcinogen, is one and it was discussed in Chapter 3. Other common associations include copper, phosphate, molybdenum, and vanadium. Lead and selenium are additional metals noted in some Arizona mines in the EPA Abandoned Mine Lands portion of the CERCLIS[3] database. See Table 5.3 for mineral ores with which uranium (and radium) may be associated. Vanadium and uranium are commonly mined together on the Colorado Plateau (U.S. EPA 2006a).

Most of the mines located in the sedimentary sandstone deposits of the southwestern United States are not in pyritic formations, and the resulting runoff waters or pit lakes are generally neutral to alkaline in character (pH of 7 or higher). Low precipitation rates and the resultant lack of water may further reduce the potential for generation of acid mine or rock drainage (AMD or ARD) from waste rock, for example, in both the Colorado Plateau and the Shirley Basin of Wyoming (U.S. EPA 2006a). For mines elsewhere, AMD/ARD can be a problem. Midnite Mine in Washington State is an example of a large uranium mine in which AMD did occur. While AMD/ARD can enhance contaminant mobility by promoting leaching from exposed wastes and mine structures, releases can also occur under neutral pH conditions (U.S. EPA 2000a).

The effects of the metals can be assessed within the Superfund methodology. An example of this was mentioned as part of the discussion of the White King/Lucky Lass Superfund site. In that ecological risk assessment, no adverse ecological effects were seen from the radioactive materials, but the associated metals did pose a potential ecological threat to a local shrew species. Other mining sites have created environmental problems, and some are on the National Priorities List for cleanup. Midnite Mine, for example, underwent a preliminary ecological risk assessment (URS 2003), and a number of metals were examined, including copper, lead, arsenic, selenium, uranium, vanadium, zinc, molybdenum, and chromium. Uranium-235, uranium-238 and thorium-232 were also evaluated. According to the final ecological risk assessment, there were situations where both the radioactivity and the metals exceeded guidelines (Lockheed Martin 2005). The record of decision concludes that, "Contaminants in surface water, ground water, surface materials, and air represent a threat to human and ecological receptors" (U.S. EPA 2006c).

Although not analyzed here, there may be environmental effects, in addition to potential human health effects, from unreclaimed uranium mines. While many of the mines are remote and may not be visited by humans, the flora and fauna would be exposed for much longer periods of time, and thus could be affected by unreclaimed mines. Issues to be considered for an ecological risk

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[3] The Comprehensive Environmental Response, Compensation and Liability Information System (CERCLIS) Database contains general information on sites across the nation and U.S. territories including location, contaminants, and cleanup actions taken. The database can be downloaded from the web at http://www.epa.gov/superfund/programs/aml/amlsite/nonnpl.htm.

assessment of unreclaimed mines could include the identification of stressors for the different types of uranium mines, affected species at different sites, the potential exposures, and the endpoints for determining effects.

Although radiological and chemical toxicity should be treated as concerns, the closure of mine shafts that have long remained unreclaimed must also be considered carefully. In parts of the country where open mine shafts have long been part of the landscape, animal species—most notably bats—may rely on those mines shafts as critical habitat. Endangered bat species have been documented nesting in unreclaimed mines. If a survey by a biologist determines the presence of bat species in an abandoned mine, adequate closure may be accomplished by means of a "bat gate," a metal grate that prevents humans from entering but allows the free passage of bats (Burghardt 2003).

Table 5-3. Mineral Commodities with Uranium Associations

Several mineral ores often, though not always, have TENORM-associated wastes resulting from co-occurrence of uranium and radium.

Aluminum (bauxite)
Coal (and coal ash)
Copper
Fluorospar (fluorite)
Gypsum
Molybdenum
Niobium
Phosphate (phosphorus)
Potassium (potash)
Precious metals (gold, silver)
Rare earths: yttrium, lanthanum, monazite, bastanite, etc.
Tin
Titanium (leucoxene, ilmenite, rutile)
Tungsten
Vanadium
Zircon

Source: U.S. EPA 2003d.

6.0 UNCERTAINTIES

The major uncertainty in this analysis is the actual exposure that people will experience. Because many abandoned uranium mines are on federal lands, the most likely exposure scenario is recreational use, but the true nature and extent of the recreational use is unknown. Exceptions to this are Native Americans who live around the uranium mines and personnel who may work around the sites. In addition, the number of people exposed would depend on the number of mines that have been reclaimed. Some mines have been reclaimed, but the total number is unknown. In the 1983 EPA study (U.S. EPA 1983b), the authors noted that many of the mines from the 1950s and 1960s had not been reclaimed at that time. Survey work done by Otten for EPA (1998) found that in many uranium-producing states, perhaps half of the mines or more had been reclaimed. No other survey has been conducted since that time. In the 1970s, surveys identified hundreds of potential buildings constructed from what was believed to have been uranium mining-related material. However, little is known about the extent of building contamination or the level of contamination in the building materials, or whether they remain or are occupied.

Another uncertainty is the true effect uranium mines have on the ground water and the subsequent use of the water. In many parts of the Southwest, where many of the mines are located, the primary sources of drinking water are deep-lying aquifers, so shallow open-pit or underground mines may not contaminate the water because of the limited infiltration. Furthermore, since uranium mines are in mineralized areas, it can be difficult to differentiate between a groundwater problem caused by a uranium mine and naturally occurring uranium. In other instances, in areas with surface water flow, such as the Ross-Adams Mine in Alaska, or Orphan Mine in Arizona (see U.S. EPA 2006a), a local source of drinking water may be contaminated by water flowing through uranium mine waste or the mine itself, and serve as a possible ingestion pathway for radiation exposure.

The other major uncertainty involves the concentrations of contaminants. The primary radiological contaminant of concern is Ra-226, which would contribute the greatest risk—from external exposure—to the occasional recreationalist. Uranium may also be a contaminant of concern, especially if it can migrate to a drinking-water source where its chemical toxicity becomes the health hazard. There is information that can be used to bound the potential exposures to both of these radionuclides, but the concentrations vary within a site and between the true overburden and amount of protore at a specific mining location. Arsenic, a carcinogen, has been shown to be associated with uranium mine wastes and can reach high levels at mine sites, but arsenic concentrations can be highly variable. At some sites, the risk from arsenic may dominate the radiological risk, and other metals may also contribute some uncertain level of hazard. Since this analysis was done on a generic, scoping basis, site-specific analyses would remove much of the uncertainties encountered here.

7.0 CONCLUSIONS

7.1 Summary

The majority of uranium production in the U.S. has come from several hundred underground and open-pit mines out of the thousands of mines and exploration workings known to exist. Some of these have been closed and remediated, at least two have been placed on the National Priorities List (Superfund) for cleanup, and others have been in standby mode where the owners have been waiting for the price of uranium to increase, as it has in 2006. The focus of this scoping report, however, has been on an investigation of potential risks from the thousands of relatively small mines and exploration sites that were abandoned over the years. With this report we have tried to identify the most likely exposure scenario for the abandoned mines, develop a first order estimate of cancer risks using some conservative assumptions, and identify if there are potential ecological effects that may develop around these mines.

Of the thousands of uranium mines in the continental United States, most are concentrated in Colorado, Utah, Wyoming, and New Mexico, and to a lesser extent, Arizona and Texas. For the small number of uranium mines in other regions, uranium is typically a byproduct of other mineral production. Many of the Four Corners States' mines are concentrated in a small number of watersheds. Though some Superfund removal actions have taken place within the Colorado Plateau, the two uranium mines on the National Priorities List are outside of the major uranium-producing states.

Most abandoned uranium mines are likely to have elevated radium and uranium concentrations, and possibly elevated levels of other contaminants such as arsenic. An analysis of the location of uranium mine records indicates that many are on federal lands, so a primary exposure scenario pertains to short-term recreational activities, including short-term occupation. Another scenario of concern is the use of mine waste material as building materials for those situations where the mines are not remote and material can be transported by nearby populations. In the recreation scenario, short-term exposure to radium, uranium, and arsenic appears to create only minimal additional cancer risk. This additional risk is dominated by external gamma exposure associated with radium in the waste material. The radioactivity in sub-ore grade uranium mine waste can be very high, so longer-term exposures from repeated visits to a high radium/high gamma site could begin to create a higher risk, even to a recreational user. The highest end of the risk spectrum is the scenario in which abandoned mine areas are used as home sites, which could pose a significant cancer risk to any long-term inhabitant. Long-term inhabitants who live near the mine sites might also use uranium mine waste material in building materials, and they would face additional risk from those radioactive building materials. It appears that those living on western Tribal lands appear to be most at risk as potential residents on or near abandoned uranium mine sites, or from the frequent visiting or passing through contaminated sites and wastes.

In general, the risks from these sites are primarily from occasional exposures and are likely to be minimal, even with conservative assumptions. The risk resulting from frequent use of a site, however, approaches a resident's exposure. Due to the predominant recreation scenario, the risk analysis examined risks in terms of days of exposure instead of the typical annual exposure,

although 350-day scenarios have been included to represent the exposure level for one year. From the estimates of the risk provided in this document, it is possible to quickly determine a first order estimate of the risks from a site, given the predominant contaminants, with the caveat that specific site conditions and site use would need to be factored in for a more realistic risk estimate.

Many of the abandoned uranium mines occur in areas with low precipitation and deep groundwater so that risk to ground-water drinking water sources is often low for at least the short-term (tens of years). However, some abandoned uranium mines occur in areas with higher precipitation. Abandoned uranium mines that are the most likely to affect groundwater are those that intersect groundwater (e.g., underground mines or deep surface mines) or are above shallow aquifers. Both radium and uranium have had MCLs established for them in drinking water supplies, but uranium is the most likely candidate to contaminate groundwater, since radium-226 is typically more immobile. In the case of uranium, the MCL is based on the limiting effect of chemical toxicity, not the radiological properties.

Ecological effects were not a focus of this report, but they were considered. Radionuclide and other heavy metal concentrations could be high enough to affect flora and fauna around abandoned mines, especially in watersheds with a high mine density. Indeed, it may be the flora and fauna that are affected much more than human health, and it may be the non-radioactive metals that produce the more significant ecological effects. This may be especially true where uranium is a secondary commodity, such as in the Lefthand Creek watershed in Colorado. At the same time, however, species may have grown accustomed to the presence of mine shafts that remain unreclaimed, and may, in fact, rely on them for habitat.

7.2 Potential Considerations for Site Prioritization

Ideally, all abandoned uranium mine sites would be remediated; however, given budget restraints, it is recognized that the most likely sites to be remediated are those that pose the greatest threat to human health and the environment. There are a number of items that could be considered when trying to prioritize the mines to be remediated. For example, in the cases where the radionuclides are likely to reach the groundwater, surface water, or springs, uranium may be the limiting radionuclide, because it is typically more mobile than radium. Radium may most often be the limiting factor in other cases because of the risk from external exposure. Less information is known about thorium values and the importance of thorium relative to radium. In some cases, the non-radiological metals may be the most hazardous of the mine waste constituents.

7.2.1 Depth to Groundwater and Annual Precipitation

EPA considers groundwater a resource for which it is easier to prevent pollution than to treat pollution after the fact. Those uranium mines that are located in areas with shallow (<50-60 feet or <~20 meters) groundwater resources have the potential to contaminate underlying aquifers within decades. Coupled with moderate amounts of precipitation (>~20 inches or >~50 cm), radioactive and metal contaminants at uranium mines could create a groundwater problem if not addressed. Large mines and underground mines that intersect aquifers have caused groundwater contamination. A scoping study such as this can identify some potential issues in this area, but it

cannot account for all the variations in site characteristics, so the geology and hydrology of a particular site would have to be examined (e.g., pH) when making remediation decisions.

7.2.2 Frequency of Use

The main tenets of protection from radioactive materials are time, distance and shielding. At one end of the spectrum, if one were to live on the mine waste materials or be exposed to mine wastes as part of construction materials, the risk of cancer from doing so could be relatively high. The scoping analysis in this report indicates that people who spend only small amounts of time at these abandoned mines probably have low risk. This low-risk consequence changes if one of these sites gets frequent use, creating a continuum of risk which we have tried to illustrate through the use of exposure calculations based on days per year of exposure.

In addition to current uses (e.g., recreation), the potential for future population growth and use could also be considered. The West and Southwest have experienced population growth in recent years, and second homes have also recently become popular in areas that were formerly primarily recreational. Anecdotal information suggests that home developments may be encroaching on areas of abandoned mines or mine wastes. In these cases, nearby populations may increase the potential use of these properties, with a concurrent increase in potential exposures.

The frequency of use may be related to their distance from roads. In other words, how remote are the mines? With the mines located on federal property, access may depend on fire roads or roads previously used during the mine's operation which are likely in disrepair so that access would be by foot, all terrain vehicles or possibly four-wheel drive vehicles. Some mines, however, may be located along well developed roads with easy access which may lead to more frequent visits or visits of longer duration.

7.2.3 Presence and Concentrations of Contaminants in Soils, Water, and Sediments

A major driver for the overall risk is the presence of contaminants. In the case of abandoned uranium mines, the contaminants would be both radioactive and stable metals. Radium, uranium, and possibly thorium could pose risks from external gamma exposures, but arsenic and other heavy metals (e.g., vanadium, selenium, copper, molybdenum) could pose a risk as well, especially to flora and fauna if there are enough waste materials. Some of the waste material quantities may be so minimal in area or volume that they do not pose a problem.

7.2.4 Density of Mines

One observation from this analysis is that the uranium mines are often along drainages where there can be a high density of mines or mine portals and associated wastes (see Figure 2.4 for example). While one mine may not pose a problem, a number of mines close together may increase the potential for adverse health or ecological effects, which may be seen at some distance from an individual mine site.

7.2.5 Level of Acceptable Risk

Lastly, the level of acceptable risk will also be important to determining how to prioritize the mines. The level of cancer risk typically used by EPA in the Superfund program is the risk range of 1 in 10,000 (10^{-4}) to 1 in 1,000,000 (10^{-6}) and the level of acceptable risk for non-carcinogens (i.e., some metals) is a hazard ranking less than 1. Sites which get frequent visitation may approach the upper end of the cancer risk range, while other sites would be at the lower end of the risk range. Residential exposure to uranium mine wastes, if it were to occur, would most likely be at the high end of risk range or even above.

The scoping analysis presented in this report indicates that at least some of the abandoned uranium mines have the potential to pose health and ecological hazards from both radioactive and non-radioactive materials. Data indicate that the concentrations of contaminants can be high enough to create adverse health effects if people were to spend substantial time on the sites. Non-radiological contaminants may be the most significant hazard, especially for flora and fauna. Since many of the sites are on federal lands, the largest exposure would be from recreational visits, or occupational use by a government employee or contractor, where the relatively short period of exposures would minimize the impact of high concentrations of contaminants. For the occasional visitor to abandoned mines, the mine wastes typically do not produce a significant radiation risk. However, individuals who visit a site frequently or for long periods of time can incur substantial risks. Residential exposure through on-site exposure or through the use of contaminated building material is not likely in most cases, except for some Tribal members, such as in the Navajo Nation, or other nearby residents. Where it does occur, the risks from these situations could be quite high.

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Appendix I. Swimming Risk

This appendix provides supplemental information on the swimming risks discussed in Chapter 3. Swimming risks were assumed to come from two sources; (1) ingestion and (2) immersion. In the drinking water discussion of Chapter 3 of the main report, we identified a recreational exposure scenario of 10 years of exposure, 14 days a year, and drinking 2 liters of water a day. For this recreational scenario, the lifetime cancer risk from drinking water contaminated with a range of uranium concentrations was in the 10^{-5} to 10^{-6} risk range. However, the analysis in Chapter 3 did identify that long-term use of pit-lake water could pose cancer risks. The potential hazard from the pit lakes may be greater from metals, such as arsenic, than from radionuclides. Since ingestion risks from ranges of radionuclide concentrations were discussed in Chapter 3, they are not discussed further here.

To calculate the immersion risks from exposure to radionuclides, we first calculated a dose using the formula modified from Whelan et al. (2006), and then applied a dose to risk coefficient from Tables 7.3 and 7.6 of Federal Guidance Report No. 13 (U.S. EPA 1999) to develop age-averaged site-specific cancer mortality and morbidity risk estimates. The formula for the immersion dose is as follows:

$$\text{swimming external dose} = Cw * EDFS * T\ \text{exposure}$$

where Cw is the radionuclide concentration in the water in pCi/L, the EDFS is the External Dose Factor for Swimming in rem/hr per pCi/L, and the time of exposure is length of time a swimmer would be in the water in hours. To calculate the EDFS for the uranium and thorium decay series, we used the DCAL program (Eckerman et al. 2006), a comprehensive software system for the calculation of tissue dose and subsequent health risk from intakes of radionuclides or exposure to radionuclides present in environmental media. The results are listed below in Tables AI.1 and AI.2 for the U-238 and Th-232 decay series. Note that Ra-226 is included in the U-238 dose and risk calculations. The totals would apply if secular equilibrium were assumed; this is an unlikely case, because of the tendency for the radionuclides to settle into the sediment, as well as being dissolved in the water column. Table AI.3 shows the dose equivalent and risks per pCi/L for both decay series combined as a function of time spent immersed in the water.

In summary, the cancer risks from immersion due to swimming are very small per pCi/L from the U-238 and Th-232 decay series. Even if secular equilibrium were assumed and all the radionuclides in either series were present, the cancer risks from immersion while swimming are negligible for the recreational scenario, even at concentrations of 10s to 100s of pCi/L. The cancer risks from ingesting water while swimming are also anticipated to be low, based on the drinking water discussion in Chapter 3 of the main report.

Table AI.1. Uranium-238 Dose Equivalent Rate and Risk per pCi/L per hour (EDFS): Water Immersion

Nuclide	Dose Equiv. Rate [(rem/hr) per (pCi/L)]	Mortality Risk[1]	Morbidity Risk
U-238	7.8E-14	4.5E-17	6.6E-17
Th-234	8.8E-12	5.0E-15	7.4E-15
Pa-234m	2.7E-11	1.5E-14	2.2E-14
Pa-234	2.5E-09	1.4E-12	2.1E-12
U-234	1.9E-13	1.1E-16	1.6E-16
Th-230	4.5E-13	2.6E-16	3.8E-16
Ra-226	8.3E-12	4.8E-15	7.1E-15
Rn-222	5.1E-13	3.0E-16	4.3E-16
Po-218	1.2E-14	7.0E-18	1.0E-17
At-218	3.0E-12	1.7E-15	2.5E-15
Pb-214	3.2E-10	1.8E-13	2.7E-13
Bi-214	2.1E-09	1.2E-12	1.8E-12
Po-214	1.1E-13	6.3E-17	9.3E-17
Pb-210	1.4E-12	8.0E-16	1.2E-15
Bi-210	4.0E-12	2.3E-15	3.4E-15
Po-210	1.1E-14	6.5E-18	9.5E-18
Total	**5.0E-09**	**2.9E-12**	**4.2E-12**

Table AI.2. Thorium-232 Dose Equivalent Rate per pCi/L per hour(EDFS): Water Immersion

Nuclide	Dose Equiv. Rate [(rem/hr) per (pCi/L)]	Mortality Risk	Morbidity Risk
Th-232	2.20E-13	1.26E-16	1.86E-16
Ra-228	0.00E+00	0.00E+00	0.00E+00
Ac-228	1.29E-09	7.44E-13	1.10E-12
Th-228	2.41E-12	1.39E-15	2.04E-15
Ra-224	1.25E-11	7.21E-15	1.06E-14
Rn-220	5.00E-13	2.87E-16	4.23E-16
Po-216	2.24E-14	1.29E-17	1.89E-17
Pb-212	1.82E-10	1.05E-13	1.54E-13
Bi-212	2.54E-10	1.46E-13	2.15E-13
Po-212	0.00E+00	0.00E+00	0.00E+00
Tl-208	4.86E-09	2.80E-12	4.11E-12
Total	**6.61E-09**	**3.80E-12**	**5.59E-12**

[1] Mortality risk is 5.575 E-4 per rem; Morbidity risk is 8.46 E-4 per rem.

Table AI.3. Total Dose Equivalent and Risk per pCi/L per hour (EDFS): Water Immersion

Time Spent Swimming (hours)	Dose Equivalent (rem per pCi/L)	Mortality Risk (per pCi/L)	Morbidity Risk (per pCi/L)
1	1.2E-08	6.5E-12	9.8E-12
2	2.3E-08	1.3E-11	2.0E-11
3	3.5E-08	1.9E-11	2.9E-11
4	4.6E-08	2.6E-11	3.9E-11
5	5.8E-08	3.2E-11	4.9E-11
10	1.2E-07	6.5E-11	9.8E-11
15	1.7E-07	9.7E-11	1.5E-10
20	2.3E-07	1.3E-10	2.0E-10
25	2.9E-07	1.6E-10	2.5E-10
30	3.5E-07	1.9E-10	2.9E-10
40	4.6E-07	2.6E-10	3.9E-10
50	5.8E-07	3.2E-10	4.9E-10
75	8.7E-07	4.9E-10	7.4E-10
100	1.2E-06	6.5E-10	9.8E-10

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Appendix II. Calculation of Slope Factors for Naturally Occurring Radionuclides

In developing the target soil screening levels found in Chapter 3 of this report, the slope factors for external exposure to, ingestion of, and inhalation of soil at an infinite depth must be considered. This appendix calculates the slope factors for the naturally occurring radionuclides under consideration. The Radionuclide Table, Radionuclide Carcinogenicity, formerly the Health Effects Assessment Summary Tables or HEAST Tables (http://www.epa.gov/radiation/heast/index.html), lists slope factors for individual radionuclides or for decay chains consisting of a parent nuclide and its short-lived progeny (i.e., radioactive daughter products with half-lives of less than 6 months). As explained below, naturally occurring radionuclides are often associated with their long-lived decay products. The slope factors for three naturally occurring radioactive decay series─natural uranium, Ra-226, and Th-232─used in the present report include the contributions from these decay products. This appendix explains the methodology used to calculate these combined slope factors.

The following guidance is excerpted from U.S. EPA 1996a:

> *Selected radionuclides and radioactive decay chain products are designated with the suffix "+D" (e.g., U-238+D, Ra-226+D, Cs-137+D) to indicate that cancer risk estimates for these radionuclides include the contributions from their short-lived decay products, assuming equal activity concentrations (i.e., secular equilibrium) with the principal or parent nuclide in the environment.*
>
> . . . .
>
> *Note that there may be circumstances, such as long disposal times or technologically enhanced concentrations of naturally occurring radionuclides, that may necessitate the combination of the risks of a parent radionuclide and its decay products over several contiguous subchains. For example, Ra-226 soil analyses at a site might show that all radium decay products are present in secular equilibrium down to stable Pb-206. In this case, Ra-226 risk calculations should be based on the ingestion, inhalation and external exposure slope factors for the Ra-226+D subchain, plus the ingestion, inhalation and external exposure factors for the Pb-210+D subchain.*

Radium-226 slope factors for the external exposure, soil ingestion, and inhalation pathways used in this analysis were calculated according to the guidance cited above. The same logic was applied to Th-232, whose progeny includes Ra-228, which has a half-life of 5.75 y, and Th-228, with a half-life of 1.91 y. Since the naturally-occurring thorium at the uranium mines will be in equilibrium with this progeny, the thorium slope factors are calculated as the sum of the slope factors for Th-232, Ra-228+D, and Th-228+D that are listed in the HEAST tables. Natural uranium is assumed to consist of U-234, U-235, and U-238, in ratios corresponding to natural isotopic abundances. We first calculated a slope factor for the U-238 decay series, which we will call $U\text{-}238_{series}$, by taking the sum of the slope factors for U-238+D, U+234, and Th-230. Radium-226 was not included, because separate soil analyses are normally performed for radium which, due to its different chemical properties, is often not in equilibrium with uranium. In

similar fashion, we calculated a slope factor for the U-235 decay series ($U\text{-}235_{series}$) as the sum of the slope factors for U-235+D, Pa-231, and Ac-227+D. We then calculated slope factors for natural uranium, by multiplying the slope factor for the U-235 decay series by the ratio of the specific activities of U-235 to U-238 and adding this product to the slope factor for the U-238 decay series, as shown by the following expression:

$$Unat = U\text{-}235_{series} \times f_{235} + U\text{-}238_{series}$$

where f_{235} is the ratio of the specific activities of U-235 and U-238 in natural uranium and is shown in the following expression:

$$f_{235} = \frac{[U-235]}{[U-238]} = 0.046$$

Because uranium concentrations in soil are commonly reported as pCi/g of U-238, the natural uranium slope factors are normalized to a unit activity concentration of U-238. To apply these slope factors, multiply by the actual activity of U-238, not the total activity of the uranium isotopes.

Details of these calculations are shown in the Table AII-1 below.

Table AII-1. Calculation of Slope Factors for NORM Decay Series

Series	Nuclide	Activity Fraction	SF		
			External (risk/y per pCi/g)	Ingestion (risk/pCi)	Inhalation (risk/pCi)
U-Series	U-238+D	1	1.14E-07	2.10E-10	9.35E-09
	U-234	1	2.52E-10	1.58E-10	1.14E-08
	Th-230	1	8.19E-10	2.02E-10	2.85E-08
	U-235+D	0.046	5.43E-07	1.63E-10	1.01E-08
	Pa-231	0.046	1.39E-07	3.74E-10	4.55E-08
	Ac-227+D	0.046	1.47E-06	1.16E-09	2.09E-07
	Total[a]		2.14E-07	6.48E-10	6.14E-08
Ra-Series	Ra-226+D	1	8.49E-06	7.30E-10	1.16E-08
	Pb-210+D	1	4.21E-09	2.66E-09	1.39E-08
	Total		8.49E-06	3.39E-09	2.55E-08
Th-Series	Th-232	1	3.42E-10	2.31E-10	4.33E-08
	Ra-228+D	1	4.53E-06	2.29E-09	5.23E-09
	Th-228+D	1	7.76E-06	8.09E-10	1.43E-07
	Total		1.23E-05	3.33E-09	1.92E-07

[a] Sum, weighted by fractional activities

Appendix II References

U.S. EPA (U.S. Environmental Protection Agency). n/d. "Radionuclide Table: Radionuclide Carcinogenicity – Slope Factors" http://www.epa.gov/radiation/heast/docs/heast2_table_4-d2_0401.pdf

U.S. EPA (U.S. Environmental Protection Agency). 1996a. *Soil Screening Guidance: User's Guide.* EPA/540-R-96-018. Washington, DC: U.S. EPA, Office of Solid Waste and Emergency Response, July 1996.

Appendix III. Occupational and Public Risks Associated with In-Situ Leaching

Introduction

EPA's Science Advisory Board (SAB) recommended that EPA present information on in situ leaching (ISL) mining operations and uranium mill operations to provide a more complete picture of uranium production. This appendix summarizes information on environmental and health-related aspects of ISL operations. The primary sources used for this review are, *Technologically Enhanced Naturally Occurring Radioactive Materials from Uranium Mining. Volume 1: Mining and Reclamation Background* by U.S. EPA (2006), *An Environmental Overview of Unconventional Extraction of Uranium* by Marlowe (1984) and *A Baseline Risk-Informed Performance Approach for In Situ Leach Uranium Extraction Licensees* by Mackin et al. (2001).

Background

In situ leaching is an extraction process that is regulated by the NRC or its Agreement States; the waste materials and fluids are considered byproduct material (see Appendix VI of U.S. EPA 2006). However, ISL operation wells are subject to permitting under EPA's Underground Injection Control (UIC) program (U.S. EPA 2006, Appendix VI). ISL operations, also known within the uranium industry as "in situ recovery," or ISR, are discussed here to provide a more complete representation of the impacts from uranium production.

ISL is used when specific conditions exist, such as the following:

- The ore is too deep to be mined economically by conventional means
- The uranium is present in multiple-layered roll fronts that may be offset by faulting
- The ore body is below the water table
- Considerable methane and hydrogen sulfide are associated with the ore
- The ore grade is low, and the ore body is too thin to mine by conventional means
- A highly permeable rock formation exists in which uranium can be economically produced using in situ leaching

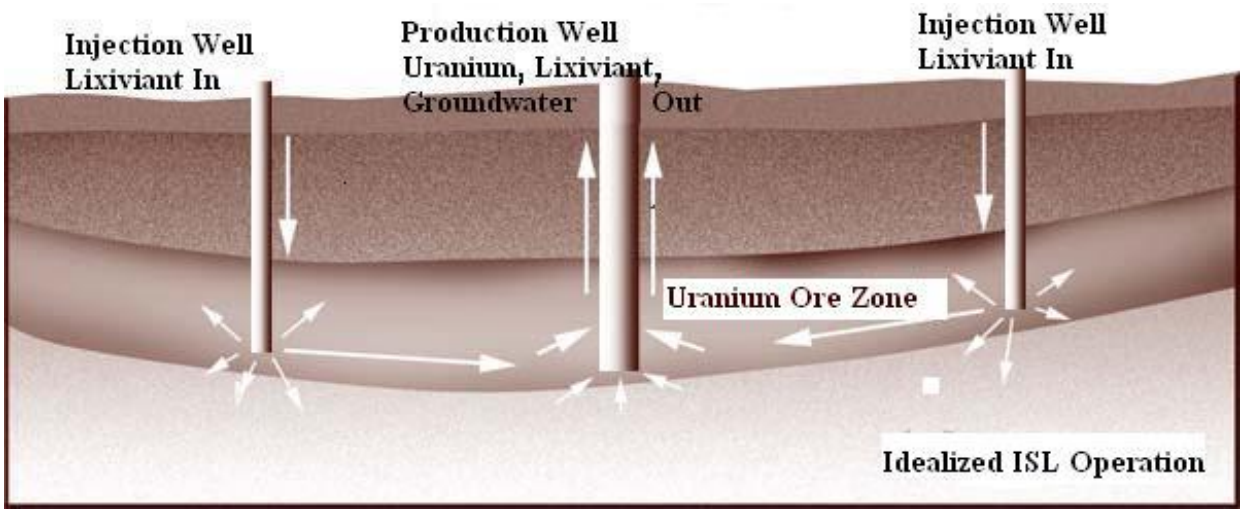
In this method of extraction, uranium ores are leached underground by the introduction of a solvent solution, called a lixiviant, through injection wells drilled into the ore body. The process does not require the physical extraction of ore from the ground, which makes it a much more economical option in many cases. Lixiviants for uranium mining commonly consist of water containing added oxygen and carbon dioxide or sodium bicarbonate, which mobilize uranium. Other ISL facilities, especially in Eastern Europe, employ an acid-based lixiviant, though this method is rarely, if ever, utilized in the United States. The lixiviant is injected, passes through the ore body, and mobilizes the uranium. The uranium-bearing solution is pumped to the surface from production wells.

The pregnant leach solution is processed to extract the uranium, usually by ion exchange or by solvent extraction. The ion exchange process employs a resin that, once fully saturated with

uranium, is flushed with a highly concentrated salt (e.g., sodium chloride) solution. This reverses the exchange process and releases uranium into the solution. The uranium solution is then sent to another process for concentration, precipitation, and drying as yellowcake. The solvent extraction process relies on unmixable properties between the pregnant leach solution and (uranium) solute. Normally, the solvents are organic compounds that can combine with either cationic or anionic solutes. For example, anionic solutions include amine chains and ammonium compounds, and cationic solutions are phosphoric acid-based. Figure AIII-1 shows a simplified version of the ISL process.

Figure AIII-1. Illustration of ISL Process

This figure shows a simplified version of how ISL solution mining works. Lixiviant is injected into the ground through wells on the left and far right, the fluid flows underground, dissolving uranium and carrying it in solution until it reaches a production well in the center. The fluid carrying dissolved uranium is returned to the surface from the production well, and piped to a production facility for refinement into yellowcake.



Source: Modified after ANAWA : http://www.anawa.org.au/mining/isl-diagram.html

When the ISL process is completed, the ore body and aquifer are placed in a restoration phase, as required by mine permits and NRC and Agreement State regulatory programs. Typically, the aquifer must be restored to background levels where possible or practical, or to its prior classification for water use in terms of the presence of metals, organics, pH level, and radioactivity. Therefore, in some cases, restoring it to the pre-operation level does not necessarily make it potable. Through the aquifer exemption process, EPA and its Delegated States determine if an aquifer or part of an aquifer is exempt from protection as an underground source of drinking water, because it is currently unusable as a source of drinking water and will not serve as a source of drinking water in the future. Approval of this exemption is necessary before a UIC permit may be issued for ISL mineral extraction wells. The aquifer exemption is permanent, and so for some operations in some states, there is no requirement for restoration of an aquifer, or part of an aquifer depending on the UIC permit, once it is exempted. EPA requires, however, that non-exempted groundwater sources be protected from contaminates migrating from the exempted portion of the aquifer.

According to Commission Order CLI-00-22, in situ leach mining (ISL) produces two categories of waste; (1) gaseous emissions and airborne particulates resulting from drying of yellowcake,

and (2) liquid waste associated with operations including well field processing and aquifer restoration (Dicus et al. 1999). A variety of methods exist to address liquid waste disposal and storage at ISL facilities, including the use of evaporation ponds, deep-well injection, land application, and surface discharge under a National Pollution Elimination System (NPDES) permit.

Figure AIII-2. Picture of an in situ leach field

Unlike a conventional mine, ISL operations produce minimal solid waste. This picture from the Wyoming Association web site shows an ISL well field.

Source: http://www.wma-minelife.com/uranium/insitu/insitufr.htm

Potential Environmental and Health Issues

While the primary environmental concern from ISL operations may be related to groundwater, Mackin et al. (2001) identify four primary risks from ISL operations in three categories:

(i) Surface environment chemical hazards
(ii) Surface environment radiological hazards
(iii) Groundwater chemical and radiological contamination hazards

The main risks to the worker are from the surface chemical and radiological hazards associated with various types of accidents at the site. Conversely, the risks to the general public pertain to the contamination of drinking water sources. Therefore, site-specific accidents would not affect the public unless a large prolonged release of hazardous chemicals and/or radionuclides were allowed to contaminate the local water supplies. In addition to hazards during ISL operation, site rehabilitation presents environmental and health concerns. Each of these issues is discussed in the sections to follow.

(i) Surface Environment Chemical Hazards (Mackin et al. 2001):

Twelve chemicals are commonly used in ISL operations and could pose hazards to ISL workers, but are unlikely to affect the general public. These chemicals, along with their intended purpose at ISL facilities, are summarized below in Table AIII-1. Potential hazardous situations involving each of these chemicals are discussed in the paragraphs following the table.

Table AIII-1: Typical Chemicals Found at ISL Operations

Chemical	Formula	Purpose at ISL Operations
Anhydrous Ammonia	NH_3	pH adjustment during uranium precipitation phase
Sulfuric Acid	H_2SO_4	Used to separate the uranium from the carbonate complex pumped from below the surface
Oxygen (gaseous and liquid)	O_2	Oxidant added to lixiviant used for extraction of uranium forming UO_3
Hydrogen Peroxide	H_2O_2	Oxidant used during the precipitation phase of uranium
Sodium Hydroxide	NaOH	pH adjustment during radium removal phase
Barium Chloride	$BaCl_2$	Used as a precipitant for radium during restoration and wastewater treatment
Carbon Dioxide	CO_2	Carbonate used to keep oxidized uranium in solution, also used for pH adjustment of lixiviant
Hydrochloric Acid	HCl	pH adjustment during radium precipitation phase
Sodium Carbonate	Na_2CO_3	Carbonate used to keep oxidized uranium in solution, also used in the regeneration/recycling resin
Sodium Chloride	NaCl	Used to regenerate/recycle the resin for further use in uranium extraction
Hydrogen Sulfide	H_2S	Used in groundwater restoration to decrease the solubility of various heavy metals
Sodium Sulfide	Na_2S	Used in groundwater restoration to decrease the solubility of various heavy metals

The main hazard posed by ammonia would be if a pipe were to break inside the processing plant. The liquid ammonia, assumed to be under high pressure, would likely have a significant spray in such an event and would pose a risk to the skin and eyes of any localized worker. In addition, as the ammonia quickly evaporates, an inhalation hazard would exist that would be exacerbated by poor ventilation. The possibility also exists for a leak in the primary holding tank or associated piping which transfers the ammonia from outside the plant to its application site.

Similar to ammonia, a break in the pipes used to transfer sulfuric acid, sodium hydroxide, and hydrochloric acid inside the plant would pose a hazard, as it is highly corrosive to the skin. Sulfuric acid and sodium hydroxide would not pose a significant inhalation hazard unless the ventilation systems in the plant were not in operation or if a worker encountered a “spray” caused by smaller leaks in the piping system. A hydrochloric acid leak could lead to a vapor inhalation hazard, especially in confined spaces. These chemicals are also highly reactive with one another and so multiple localized failures, as might be the case with fire or explosions, would cause an even greater hazard.

Oxygen gas poses a significant hazard because of its combustible properties. Similarly, hydrogen sulfide and sodium sulfide also pose risks because of their flammable properties, in addition to an inhalation as well as an eye/skin irritation hazard.

Hydrogen peroxide's main risk pertains to the degradation of the chemical into hydrogen and oxygen gas which can be caused by mechanical shock, incompatible materials, light, ignition sources, excess heat, strong oxidants, rust, dust, or a pH greater than 4.0. Also, if the chemical is contained within an especially rigid tank, the casual degradation of the H_2O_2 into water and oxygen gas would cause expansion which could rupture the holding tank. A pipe failure event within the processing plant holds similar risks to that of ammonia and sulfuric acid.

Barium chloride is only considered a hazard if it is inhaled or ingested. Since the chemical is in solution form at an ISL plant, this would only become significant if the worker encountered a "spray" from a leaky piping system. Carbon dioxide from a leak can pose a risk of asphyxiation if it occurs in a sufficiently confined space. This can be avoided if a self-contained breathing apparatus were used when entering confined spaces where the displacement of oxygen with carbon dioxide is possible.

Sodium chloride and sodium carbonate both are very irritating to the eyes and the skin. In addition, sodium carbonate can pose an inhalation hazard when it is in its salt stage (dust inhalation) or from small leaks which form a spray of the sodium chloride/carbonate solution. Sodium carbonate also reacts readily with HCl and H_2SO_4.

(ii) Radiological risks

Thickener Tank Failure

The thickener tank stores wet yellowcake slurry before it is sent to a precipitation operation and dried into U_3O_8 yellowcake. Thickener tank failure can pose an inhalation risk to workers if spills are not cleaned up before the contaminants are allowed to dry. This accident scenario would not be a significant risk to off-site residents.

The thickener tank itself does not pose any external exposure risk, as most of the uranium progeny have been removed and the alpha component would be significantly attenuated by the slurry. Annual external exposures have been calculated to be 120 mrem for the limiting case of a worker standing directly next to the thickener tank for an entire 2,000 hour work year (Mackin et al. 2001).

If the yellowcake slurry is allowed to dry after a spill incident, it would pose a significant risk of uranium inhalation. Conservative treatments indicate that the dose to the public from a massive spill and subsequent airborne contamination event remain below the radiation dose limits established by 10 CFR 20 for members of the general public, however, the intake to an unprotected worker has the potential to exceed the 5 rem annual occupational limits (Mackin et al. 2001).

Yellowcake Dryer Accident

As discussed above, the dried yellowcake which consists of quantities of U_3O_8, can pose a significant inhalation hazard to the onsite worker when it is allowed to dry. Failure of the dryer cake systems can stem from a number of accidents, including fire/explosion (worst case), spill over of dryer contents due to a faulty discharge valve, failure of offgas treatment systems causing the gases to release into the dryer area, and damage to the facility via natural disasters. It is important to note that the failure of the yellowcake dryer systems due to natural disasters is effectively bounded by the fire and explosion scenario. Exposures from a yellowcake dryer accident would presumably be of similar magnitude to that of the thickener tank scenario. (Mackin et al. 2001)

Exposure to Pregnant Lixiviant or Loaded Resin

Pregnant lixiviant and loaded uranium resin may pose a radiological hazard as an external exposure source, and present the possibility of inhaling elevated levels of radon-222. The most likely indoor exposure incident would occur if the pregnant lixiviant/resin were released due to a pipe or valve failure during the ion-exchange process, at which point the solution would drain from the ion-exchange column and the radon gas would be released to the air.

In addition to the inhalation hazard from radon, the pregnant lixiviant contains some other radioisotopes of interest that may also cause a significant exposure. These radioisotopes are shown in Table AIII-2, along with typical activity concentrations (Mackin et al. 2001).

Table AIII-2: Radionuclides with Typical Activity Concentrations* found in Pregnant Lixiviant/Loaded Resin

Radionuclide	Activity Concentration (pCi/L)
^{222}Rn	8.0×10^5
^{226}Ra	3.4×10^3
Natural Uranium (^{234}U, ^{235}U, ^{238}U)	1.7×10^5
^{218}Po	3.4×10^3
^{214}Bi	3.4×10^3
^{214}Po	3.4×10^3

*Progeny assumed to be in equilibrium

Conservative treatments of a possible spill incident have been modeled to show that a maximum annual exposure would be 27 mrem to a subject standing on a spill of infinite area and depth; with the consideration of loaded resin, this value becomes much lower. Since such a spilling event would likely be cleaned up expeditiously, such an exposure is not likely and is also well within the limits established in 20 CFR 20 for the general public, as well as the site worker (Mackin et al. 2001).

Exposures from the failure of near surface piping and subsequent runoff into containment ponds can also pose a possible hazard to workers. It is likely that the inhalation component in this scenario is negligible due to the dilution of the radon gas releases by ambient air; however, the

external component would be similar to the indoor scenario previously described. See the next section for further discussion of outdoor near-surface releases.

(iii) Groundwater Contamination Risks

Due to the nature of the ISL process (specifically the low pH and oxidation mechanisms), other heavy metals and hazardous elements are also mobilized from the ore and can contaminate the groundwater. These elements include the radioisotopes and progeny of uranium, thorium, radium, and radon, as well as the non-radioactive elements such as arsenic, vanadium, zinc, selenium, and molybdenum (for a more complete list see Table AIII-3). Because these elements become mobilized in the target aquifer by the process of uranium extraction, it is possible for them to migrate out of the ore body into surrounding aquifers which might feed the local water supply. The underground propagation of this contamination into surrounding water is known as an excursion. Horizontal excursions refer to the lateral movement of the water, while vertical excursions indicate contamination of aquifers above and below the target ore body.

In order to detect and minimize this process, ISL facilities drill monitoring wells outside of the main well-field at a distance sufficient to detect any excursion events, while minimizing any erroneous indicators as a result of normal fluctuations. Horizontal excursions are more common than vertical excursions, but do not often become problematic to the outside water supply as long as they are detected and cleaned up within a reasonable time period. Vertical excursions are generally a result of well casing failure (ineffective cementing of well casing), improper sealing of abandoned exploration wells, or discontinuous or permeable natural confinement layers. Similar to horizontal excursions, vertical excursions do not pose a significant threat unless allowed to persist over significant periods of time—this is unlikely if geological properties of the confinement layers are accurately characterized (to prevent downward vertical excursions), and the well shafts are effectively cased and proper monitoring well stations have been established. Along with well monitoring techniques, general practice at ISL facilities is to limit the injection of lixiviant so that it is always slightly less in volume than the product solution that is pumped out of the aquifer. This operating policy, known as “process bleed,” would effectively preclude excursions caused by overloading the aquifer, and the subsequent expansion and redistribution of the water.

In the United States, excursions have been frequently detected by the monitoring wells located around the well field. One of the more infamous and environmentally problematic ISL operations was located at Irigary, Wyoming. This facility was plagued by persistent environmental excursions which began in mid-March of 1979, and were not brought under control until early July of that same year. The Wyoming Department of Environmental Quality reported that these excursions were a result of the neglect of injection pressure monitoring as well as testing the integrity of the well casings (Mudd 1998). Another significant example is the Bruni mine in Texas, where there was a continued problem with both leachate spills and excursions. The Texas Department of Water Resources reported that at one point during the operational period the Bruni mine was cited for fourteen excursion incidents, while only five had originally been reported (Mudd 1998). Despite these scenarios, no significant contamination of local water supplies has been reported as a result of these excursions.

In addition to the below ground excursion incidents, the groundwater can become contaminated due to failure of the near-surface or surface piping systems which transfer the pregnant lixiviant from the well field area to the processing facility. Typical activity concentrations for the radionuclides present in lixiviant are given in Table AIII-2. In addition, Table AIII-3 displays the maximum measured concentrations of non-radioactive contaminants in pregnant lixiviant based on a survey of available licensing documents (Mackin et al. 2001). Once the pregnant lixiviant solution is released, there are three potential outcomes for the contamination; runoff into surface bodies of water, absorption into the soil and possible subsequent infiltrations of the groundwater, or runoff into a surface pond designed to prevent groundwater contamination. The first two scenarios show the possibility for contamination of drinking water sources and would have an obvious environmental impact if not dealt with in a timely fashion. The third scenario poses a possible radiological hazard for workers at the site and is discussed in Section ii of this appendix.

Table AIII-3. Maximum Measured Non-radioactive Contamination in Pregnant Lixiviant

Contaminant	Concentration (mg/L)	Contaminant	Concentration (mg/L)
Arsenic	0.3	Barium	0.6
Boron	0.2	Cadmium	0.01
Chloride	1,800	Chromium	0.03
Copper	0.04	Flouride	1
Iron	0.02	Lead	0.01
Manganese	6	Mercury	<0.0001
Molybdenum	62	Nickel	0.09
Nickel	0.09	Nitrate	1
Selenium	5	Silver	<0.01
Sulfate	1,200	Total dissolved solids	5,500

(iv) Post-Operation Site Restoration and Rehabilitation

There are two main methods employed to restore the contaminated aquifer back to its preoperational conditions. In general, the first method employed is termed “groundwater sweep,” and involves pumping out the equivalent volume of groundwater from the mined aquifer and replacing it with fresh uncontaminated water. The volume of water pumped out of the mined ore zone is known as the “pore volume.” The pore volume can then be moved to an evaporation pond to remove the water and then dispose of the residual wastes. An alternate disposal of the pore volume is to inject the water into much deeper aquifers designated for waste disposal. In this case, the increased levels of contaminant should not affect neighboring aquifers or potential drinking water sources. This method has proven to be useful at the beginning stages of the restoration process. However, because of the heterogeneous properties of the ore zone aquifer, complete restoration of the mining site by this technique alone is not economical. Furthermore, many site locations do not have the resources for the large amount of clean groundwater that is required for an extensive groundwater sweep operation.

The second technique that can be employed is treating the contaminated pore volume via reverse osmosis. Here, the water is pumped out of the ore zone and passed through a reverse osmosis

membrane at high pressure. This process separates the aquifer water into a highly concentrated liquid contaminant and a clean water volume known as the reverse osmosis (RO) permeate. The RO permeate is then recirculated into the ore zone using alternating pumping wells to effectively flush the heterogeneously distributed lixiviant present in the aquifer. The benefits of reverse osmosis are that no outside source of groundwater is needed to replace the pumped pore volume, since the volume is being treated and re-injected into the depleted ore zone. In practice, this method can only be employed after groundwater sweeping, because the high concentrations of contaminants during the initial stages of the restoration process tend to disrupt the RO membranes (Davis and Curtiss 2005).

Chemicals such as hydrogen sulfide or sodium hydrosulfide may also be added to the re-injected water during the later stages of restoration to achieve a state of "chemically reducing conditions." The effect of these chemicals is to decrease the solubility of several contaminating metals that are of concern, including uranium, selenium, arsenic, and molybdenum. However, there are other contaminants, such as radium, which remain mobile under chemically reduced conditions. Barium chloride is often used to precipitate radium out of waste water and can also be used during aquifer restoration to mitigate the effect of radium contamination (Mackin et al. 2001).

Despite these efforts at returning the mining site to its original preoperational state, it is very difficult to achieve complete site rehabilitation. Not all of the contamination can be removed because lixiviant will be present in sections of the aquifer that are in areas of lower porosity. The efforts to create a chemically reduced condition to render the heavy metals insoluble do not apply to all contaminants of interest. Furthermore, achieving complete rehabilitation of the site is very time consuming and costly.

Summary

In situ leaching for uranium poses several possible environmental and health-related concerns. Through the extraction and processing of uranium ore into yellowcake, many hazardous chemicals and radionuclides are utilized or concentrated which, coupled with certain accident scenarios, can pose significant risk to workers at these facilities. From a radiological standpoint, risks are mainly significant to on-site workers, and have been shown to be minimal for the public (Mackin et al. 2001). From a hazardous chemical standpoint, the immediate concern is for on-site workers; however, the risk to the public can become significant if a prolonged release of hazardous material is allowed to contaminate nearby drinking water sources.

The leaching process poses the risk of contaminating neighboring aquifers which, in turn, might affect significant water supply sources. This can happen through horizontal and vertical excursions below the surface, or from events such as pipe failure on or near the surface. The risk of excursions is mitigated by the inclusion of vertical and horizontal monitoring wells located around the perimeter of the ore zone, as well as the operational practice of "process bleeding." The wells are designed to detect excursions in a short period of time, so that corrective actions and cleanup operations can take care of the problem before the water sources outside of the mining site are significantly degraded.

Finally, in situ leaching poses a problem from a restoration standpoint. Although there are multiple techniques to restore the mined aquifer to its preoperational state, in many cases the lixiviant can never be completely purged from the site. Attempts to bring the aquifer to a chemically reduced state cannot account for all types of contaminants, and the entire rehabilitation process is both expensive and time consuming.

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Appendix IV. Risks Associated with Conventional Uranium Milling Operations

Introduction

Although uranium mill tailings are considered byproduct materials under the AEA and not TENORM, EPA's Science Advisory Board (SAB) recommended that EPA present information on uranium mill operations, as well as *in situ* leaching (ISL) mining operations, to provide a more complete picture of uranium production. This appendix summarizes information on environmental and health aspects of uranium mill operations. The primary sources used for this review are "*Technologically Enhanced Naturally Occurring Radioactive Materials from Uranium Mining. Volume 1: Mining and Reclamation Background"* by U.S. EPA (2006), "*Final Generic Environmental Impact Statement on Uranium Milling Volume 1 and 2"* by U.S. NRC (1980), "*Final Environmental Impact Statement for Standards for the Control of Byproduct Materials from Uranium Ore Processing (40 CFR 192) Volume 1"* by U.S. EPA (1983), and "*Uranium Mining and Milling Wastes - An Introduction*," by Peter Diehl of the WISE Uranium Project (2004)**.**

Background

Uranium milling is the process of converting raw ore as it arrives from mining operations into a product known as uranium yellowcake. The raw uranium ore and resultant yellowcake are shown in Figure AVI-1, and a generalized schematic of a typical milling process is shown in Figure AVI-2.

The first steps in the milling process involve crushing and grinding the ore in order to obtain smaller, uniform particle sizes throughout. Often, water is added during this stage to control dust, or lixiviant may also be added to facilitate the extraction process. Screens separate fine particles, which continue to the next stage in the milling process, from coarse particles, which are recirculated in the milling circuit. Dust that is not sufficiently suppressed by the addition of water/lixiviant is generally collected by air pollution control mechanisms, which return the fugitive particles to the milling process.

Once the ore is ground into uniform small particles, the processed ore moves to the leaching stage. In the most common leaching method, known as "acid leaching", uranium is removed from the processed ore with sulfuric acid. Sodium chlorate is also added as an oxidizing agent to improve the solubility of the uranium. An alternative approach is alkaline leaching, which is preferable when the raw ore contains a significant portion of limestone (greater than 12%), because the acid leaching process then requires uneconomically large amounts of acid to be effective. Alkaline leaching, however, requires much finer grinding of the ore in comparison to acid leaching. Both methods of leaching have similar environmental and health impacts; however, the waste produced from acid leaching is generally more mobile and will be used as the bounding scenario in this treatment (U.S. EPA 1983).

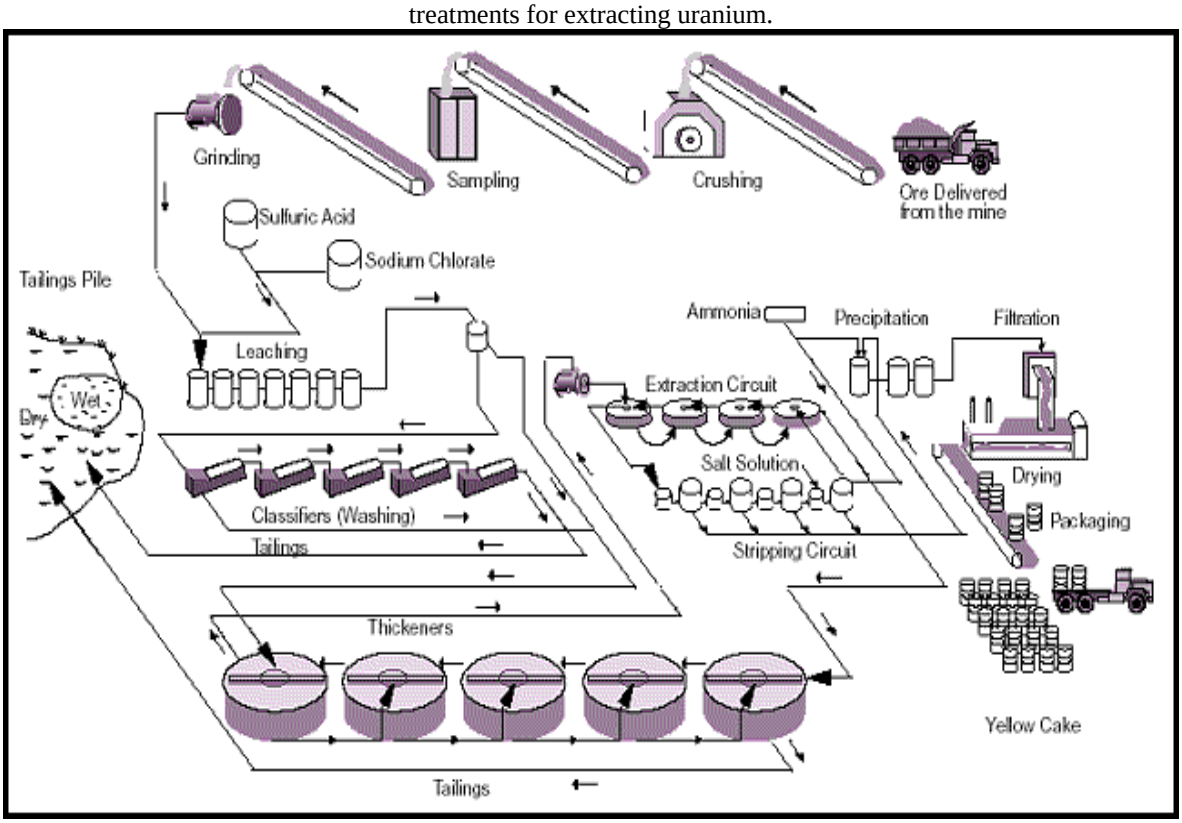
Figure AIV-1. Raw Uranium Ore and Yellowcake Product

This figure shows the incoming raw uranium ore as it enters the uranium milling process (left), as well as the final product of uranium yellowcake (right)

Source: http://www.eoearth.org/upload/thumb/c/c1/Uranium_ore_square.jpg (left)
http://www.eia.doe.gov/kids/energy_fungames/energyslang/images/yellowcake1.jpg (right)

Figure AIV-2. Generalized Uranium Mill Physical Layout

This figure shows how a uranium mill is physically set up to crush raw ore into particles amenable to chemical treatments for extracting uranium.



Source: http://www.eia.doe.gov/cneaf/nuclear/page/uran_enrich_fuel/uraniummill.html

After the leaching stage, the pregnant lixiviant generally contains about 50-60% solids. These solids, called "tailings," are filtered out and sent to on-site tailings piles or impoundments in the form of sands and slimes. Once most of the solids have been removed, the filtered lixiviant is transferred to an extraction circuit where the desired uranium is stripped from the pregnant lixiviant, followed by a precipitation and drying process, which produces the desired yellowcake product.

Potential Environmental and Health Issues from Mill Tailings

The wastes produced during the milling process and stored in tailings impoundments are the principal source of milling-related health and environmental hazards. Typical properties of these mill tailings are shown in Table AVI-1. During the milling process, nearly 90% of the uranium contained in the ore is removed, and so the primary radiological concern is the remaining progeny associated with uranium such as thorium, radium, radon, and lead. The actual activity of these uranium progeny can vary depending on the specific methods employed,; however, as much as 50-86% of the original activity of the ore is retained in the mill tailings (U.S. EPA 2006). Hazardous stable elements are also extracted from the ore and transferred to the tailings piles, including arsenic, copper, selenium, vanadium, molybdenum, and other trace heavy metals.

Table AIV-1: Typical Properties of Uranium Mill Tailings

This table displays the chemical and radiological properties of the three classifications of uranium mill tailings (sand, slime, and liquid). Table was adapted from U.S. NRC 1980 and found in U.S. EPA 2006

Tailings Component	Particle Size (µm)	Chemical Composition	Radioactivity Characteristics
Sands	75 to 500	SiO_2 with <1 wt% complex silicates of Al, Fe, Mg, Ca, Na, K, Se, Mn, Ni, Mo, Zn, U, and V; also metallic oxides	0.004 to 0.01 wt % U_3O_8[a] Acid Leaching: 26-100 pCi ^{226}Ra/g; 70 to 600 pCi ^{230}Th/g
Slimes	45 to 75	Small amounts of SiO_2, but mostly very complex clay-like silicates of Na, Ca, Mn, Mg, Al, and Fe; also metallic oxides	U_3O_8 and ^{226}Ra are almost twice the concentration present in the sands Acid leaching:[b] 150 to 400 pCi ^{226}Ra/g; 70 to 600 pCi ^{230}Th/g
Liquids	[c]	Acid leaching: pH 1.2 to 2.0; Na^+,NH_4^+, SO_4^2, Cl, and PO_4^3; dissolved solids up to 1 wt % Alkaline leaching: pH 10 to 10.5; CO_3^2 and HCO_3; dissolved solids 10 wt %	Acid leaching: 0.001 to 0.01% U; 20 to 7,500 pCi ^{226}Ra/L; 2,000 to 22,000 pCi; ^{230}Th/L Alkaline leaching: 200 pCi ^{226}Ra/L; essentially no ^{230}Th (insoluble)

[a] U_3O_8 content is higher for acid leaching than for alkaline leaching

[b] Separate analyses of sands and slimes from alkaline leaching process are not available. However, total ^{226}Ra and ^{230}Th contents of up to 600 pCi/g (of each) have been reported for the combined sands and slimes.

[c] Particle size does not apply. Up to 70 % vol. of the liquid may be recycled. Recycle potential is greater in the alkaline process.

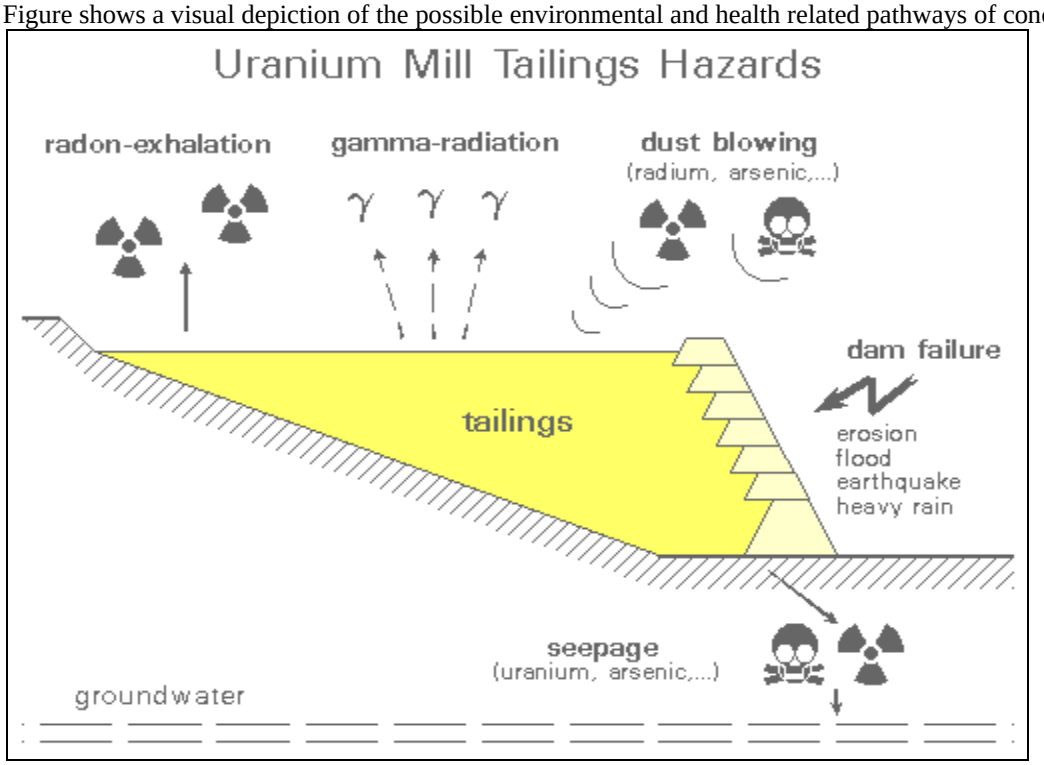
The five on-site environmental pathways through which these tailings impoundments pose a risk are represented schematically in Figure AVI-3. In addition to the on-site scenarios, tailings have also been taken off-site and used as an inexpensive building material by some local populations. Each of these hazard pathways is listed below and the associated risks are discussed later.

(i) The release of gaseous radon-222 to the atmosphere and subsequent inhalation
(ii) Possible dust loading of contaminants from the impoundment due to natural wind conditions

(iii) The localized effect of direct external gamma radiation exposure from the tailings impoundment
(iv) Ground seepage and subsequent contamination of local aquifers, which has the potential to affect the water supply
(v) Dam failure due to erosion or natural disasters (flood, earthquake, etc.)
(vi) Improper use of tailings as a building material

All six of these hazard scenarios can apply to the general public and, with the exception of building materials, to the plant workers themselves. In addition, plant workers have added risks associated with accidents that may occur within the mill. The additional issues associated with workers are discussed in a separate section.

Figure AIV-3: On-Site Accident and Risk Scenarios Associated with Uranium Mill Tailings
This Figure shows a visual depiction of the possible environmental and health related pathways of concern



Source: http://www.wise-uranium.org/uwai.html

(i) Gaseous Radon-222 Inhalation

Radon-222 is an inert radioactive gas that can readily diffuse to the surface of a tailings impoundment where it would be released to the atmosphere. The main hazard of radon inhalation is the damage to the lung from four of its shorter-lived decay products (Po-218, Pb-214, Bi-214, and Po-214). Of particular concern are the two isotopes of polonium (Po-218 and Po-214), because they produce alpha particles, which are approximately 20 times more destructive than gamma or beta radiation. Because radon-222 has a half-life of approximately 3.8 days, it has the opportunity travel a significant distance in the atmosphere before decaying. U.S. EPA 1983 states that the health of populations living at a distance greater than 80 km from a tailings pile might be affected. The radon concentration at the edge of a typical tailings pile is approximately 4 pCi/l (WISE 2004). Using the

methodology outlined in Chapter 1 of this report, a year-long exposure under these conditions would correspond to a lifetime risk of lung cancer of 1.1×10^{-2}.

(ii) Inhalation of Particles from Dust Loading

Dust loading occurs when wind blows over a dried portion of the tailings and dust containing hazardous contaminants is suspended in the air. Dust loading typically becomes a hazard in the post-operational phase of a uranium mill, as the tailings pile begins to dry, and may be exacerbated by any de-watering treatment that is performed to minimize ground seepage [see section (iv)]. The hazards associated with dust loading are dependent on the weather conditions and the amount of dried material that is available for suspension. It has been estimated that a person would have to inhale 2 grams of uranium mill tailings in a year to reach the annual dose limit for the general public (100 mrem). Assuming a continuous exposure and a breathing rate of 0.9 m^3/hr, this would correspond to a dust loading of 0.24 mg/m^3 (WISE 2004).

(iii) Direct Gamma Exposure

Uranium mill tailings pose an external exposure hazard from radioactivity that is present in the waste. Although milling operations generally remove about 90% of the uranium from the ore, the remaining waste can contain up to 86% of the original radioactivity which is mostly composed of uranium decay products such as radium and thorium. Worst-case external exposures have been estimated to be 0.41 mrem/h, if the subject were standing directly on top of the tailings; for a continuous yearly exposure, this yields a dose of 3.6 rem.

(iv) Groundwater Contamination

Groundwater contamination is so heavily dependent on site-specific parameters, such as the chemical characteristics of the waste products and soil, the location of neighboring aquifers, and the hydrology and geology of the site, that any general numerical risk assessment of groundwater contamination is of limited utility. Groundwater contamination can become a problem if liquid wastes from tailings impoundments seep into the ground and are transferred into shallow local aquifers. Mills employing acid leaching processes are of special concern, because this method renders the waste products more soluble than an alkaline leach process. The radiological contaminants would likely be pulled out of the seepage water into the immediate soil and so do not have the mobility to move offsite into neighboring aquifers. However, water-soluble non-radiological hazards may be problematic, including molybdenum, selenium, chlorine, sulfate, nitrate, arsenic, lead, and vanadium. An NRC report (1980) concluded that 95% of any possible groundwater contamination would occur while the site was in operation. Also, seepage should be expected unless the tailings pile was built on an artificial liner or impermeable natural clay formations. Besides lining tailings impoundments, milling waste is sometimes dewatered before disposal to reduce the risk of groundwater contamination. Dewatering, however, causes an increase in the rate of radon gas emissions (increase by a factor of 3.4 when comparing wet versus dry tailings) and also makes the pile more susceptible to wind-driven dust loading. An example of dewatering occurs at the White Mesa Mill, where the dry tailings are stored in an approved below-grade disposal cell. This disposal cell is covered with the excavated earth to mitigate the effects of radon emission and dust loading (Hochstein 2003).

(v) Tailings Pile Dam Failure

The least predictable risk associated with conventional uranium milling operations is the failure of a tailings dam. A dam might fail because of poor design, natural erosion of the dam, or natural disasters such as flooding, heavy snow fall, tornados, or earthquakes. In the United States, notable dam failures include the 1977 spill in Grants, New Mexico (50,000 tons of sludge and several million liters of contaminated water), and the 1979 spill in Church Rock, New Mexico (1000 tons of sludge and 400 million liters of contaminated water). The second of these noted spill events, Church Rock, is the most notorious. It heavily contaminated the Rio Puerco river and shallow aquifers located near the river, which were used by the Navajo Nation as both an agricultural and domestic water source. As of 2003, the Navajo are still unable to use this water (Ali 2003).

(vi) Improper Use of Mill Tailings as a Building Material

As stated in Chapter 4 of the main report, the risk of radiological exposure to the general public is not only from the tailing piles themselves, but also the improper use of mill tailings as building materials. The sandy properties of mill tailings and their availability in certain economically depressed areas make their inclusion in concrete and use as a building material possible. This has occurred when tailings piles have been abandoned without having been properly closed, or when piles of tailings have fallen from trucks along rural highways. Though the problem has been documented on the Navajo reservation in New Mexico and cited anecdotally, its pervasiveness remains unknown. Tables 4.1 and 4.2 of the main report present annual dose values based on a few sample activity concentrations within a Navajo hogan. See Chapter 4 of the main report for more in-depth discussion and analysis of the improper use of tailings.

Summary of Modeled Risks to the Public

In a study by the Nuclear Regulatory Commission, a generalized case was modeled in which it was assumed that a "low level" of environmental controls were in place. This report concluded that if the mills in place during the time of the study (by 1980 there were 16 mills producing approximately 43,900 megatons of ore annually) were in full operation through the year 2000, it would result in approximately 610 premature deaths in North America through the year 2100 and 6,000 premature deaths through the year 3000. This model was based on a low level of environmental control, and did not take into account mitigating factors, such as covering the tailings to reduce the atmospheric release of the radon. The estimated 15-year committed dose to the public is shown in Table AVI-3, at the end of the document, which also includes an estimate of the risk as a percentage of the risk from normal background radiation exposure. For example, an individual near by a cluster of mills would accrue a 15-year committed dose of 340 mrem to the lung (an effective dose equivalent[*] of 41 mrem), and would represent an increase of 38% above the normal risk from background exposure (U.S. NRC 1980).

These risk estimates for fatal cancer have since been updated in U.S EPA 1983 and the results are shown in Table AVI-2. This study estimated the individual risk of cancer for a 15-year exposure to an individual at distances of 1,000-20,000 meters from the mill. The model also takes into account whether the mill was in an operational or post-operational phase. For each phase of operation, the individual 15-year risk is given as an average and a maximum value. The maximum value represents

[*] Effective dose equivalent based on the tissue weighting factors of ICRP-26

the individual who is downwind of the mill, while the average value represents the average of all wind directions (U.S. EPA 1983).

Table AIV-2: Results of the 1983 EPA Study[a] – Estimated 15-Year Risk of Fatal Cancer by Region and Phase of Operation

	Total Risk (Operational Phase)		Total Risk (Post-Operational Phase)	
Distance (meters)	**Average**	**Maximum**	**Average**	**Maximum**
1000	1.12E-03	1.97E-03	1.82E-03	3.18E-03
2000	3.39E-04	6.78E-04	5.51E-04	1.12E-03
3000	1.76E-04	3.60E-04	2.76E-04	5.72E-04
4000	1.17E-04	2.33E-04	1.89E-04	3.82E-04
5000	8.48E-05	1.74E-04	1.38E-04	2.76E-04
10000	3.18E-05	6.57E-05	5.09E-05	1.04E-04
20000	1.40E-05	2.76E-05	2.33E-05	4.45E-05

[a] Risk estimates are derived U.S. EPA 1983 Tables 6-1 and 6-2

Additional Risks to Workers

Mill workers, beyond the six pathways described above, experience added risks associated with accidents inside the milling facility. The hazards due to chemical spills inside the plant exist, but may be minor relative to potential radiological accident scenarios.

At acid leaching mills, sulfuric acid is present. Though the acid is corrosive to the skin and eyes, the leaching process is carried out at atmospheric pressure, and the risk of workers coming into contact with a spray during a pipe failure is not plausible. If there were a fire coupled with the release of sulfuric acid, then the inhalation of acid aerosols and sulfur dioxide could result in severe irritation of the eyes, mucous membranes, and respiratory tract. In addition to sulfuric acid, ammonia is often added to help control the pH level during the uranium precipitation phase. It is likely that this ammonia would be under significant pressure, creating the risk of a spray, in the event of a pipe failure, that poses a risk to the skin and eyes of any nearby worker. The ammonia would also quickly evaporate, adding an inhalation hazard if the accident occurred in a poorly ventilated area.

The radiological hazards associated with milling work potentially involve the yellowcake product in a dangerous respirable form. The two most notable accident scenarios are a thickener tank failure where the yellowcake slurry is spilled to the floor and allowed to dry, or a yellowcake dryer accident. Inhalation of the yellowcake particulates is a significant inhalation hazard, because of the presence of U_3O_8 in the cake. The reader is referred to Appendix III: Risks Associated with *In Situ* Leaching [see section (ii) Radiological Hazards] for a more detailed description of operational accidents in the milling facility, specifically those involving yellowcake.

In the NRC report (U.S. NRC 1980), it was calculated that the committed annual dose to a worker at a conventional milling facility ranges from 2.0 rem to the bone up to 7.1 rem to the lung. These annual doses would result in an effective dose equivalent of 240 mrem to the bone marrow (red) and 60 mrem to the bone surface and lung. Any exposures accrued because of accidental exposure to yellowcake would be in addition to this. This information is summarized in Table AVI-3 found at the end of the document.

Summary

The primary hazard associated with conventional uranium milling operations is the high level of radioactive contamination contained in the mill tailings (waste products). The decay progeny of uranium are the most significant of these radioactive contaminants, including radium and radon-222, which readily moves through the interstitial spaces of the tailing pile and is released to the atmosphere. Once inhaled, radon and its decay progeny can cause significant damage to the lung via alpha radiation. Other radiological hazards include direct gamma exposure from the tailings pile and the inhalation of any dust resuspended by wind. These hazards are typically mitigated through the use of a suitable cover over the tailing to reduce the radon released to the atmosphere and attenuate direct gamma exposure. A suitable cover can also eliminate the risks associated with the suspension of dust in the air.

Ground seepage of chemically hazardous constituents of tailings piles has been known historically to contaminate nearby aquifers. Modern milling facilities often employ a liner beneath tailings piles to prevent any ground seepage and subsequent groundwater contamination. The NRC concluded that 95% of the possible contamination would happen while the mill was operating, and that the threat was mainly from toxic elements such as arsenic, not the radioactive constituents of the pile.

As with any industrial facility, safe management practices are critical to the safe operation of uranium mills. Catastrophic accidents, such as a dam failure, have the potential to release large quantities of tailings, resulting in the contamination of local water supplies and the residential population. The improper use of mill tailings as a building material can also pose a severe radiological risk to private individuals, particularly in tribal communities. Accidents occurring within the milling facility could expose workers to chemical risks, and radiological risks from contact with or inhalation of uranium yellowcake.

Table AIV-3: Results of the 1980 NRC Model Uranium Mill Study – Committed Dose Values

Receptor	Dose Commitment[a] (mrem)			Risk from Mill as Percentage of Risk Due to Background (%) [b,c]
	Whole Body	Bone	Lung	
Nearby Individual[d]				
Annual 40 CFR 190 doses (excluding radon)				
1 mill	3	45	30	--
Mill cluster	4	51	36	--
Total Dose (including radon)				
1 mill	9.7	51	220	25
Mill Cluster	13	61	340	38
Average Individual[e]				
1 mill	0.061	0.50	1.6	0.19
Mill Cluster	0.66	5.8	16	1.9
Average Worker[f]				
Annual	450	2000	7100	800
Career[g]	2.1×10^4	9.3×10^4	3.3×10^5	800
Background	143	250	704	--

[a] All doses shown are total annual 15th-year dose commitments except where noted as being those covered by 40 CFR 190 limits.

[b] The range in risks due to uncertainties in health effects models extends from about one-half to two times the central value. This range does not include uncertainties in other areas (e.g. source term estimates and dose assessment models).

[c] Risk comparisons are presented for exposure received during entire mill life; that is, 15 years of exposure during operation of the mill, and 5 years of post-operation exposure while tailings are drying out, are considered. This value is greater than that from annual exposures presented because tailings dust releases increase in the period when tailings are drying.

[d] The "nearby individual" occupies a permanent residence at a reference location about 2 km downwind of the tailings pile.

[e] The "average individual" exposure is determined by dividing the total population exposure in the model region by its population total.

[f] The "average worker" exposure is determined by averaging exposures expected at the various locations in the typical mill.

[g] The career dose is based on a person who has worked 47 years in the milling industry (that is, from ages 18 to 65).

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