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.

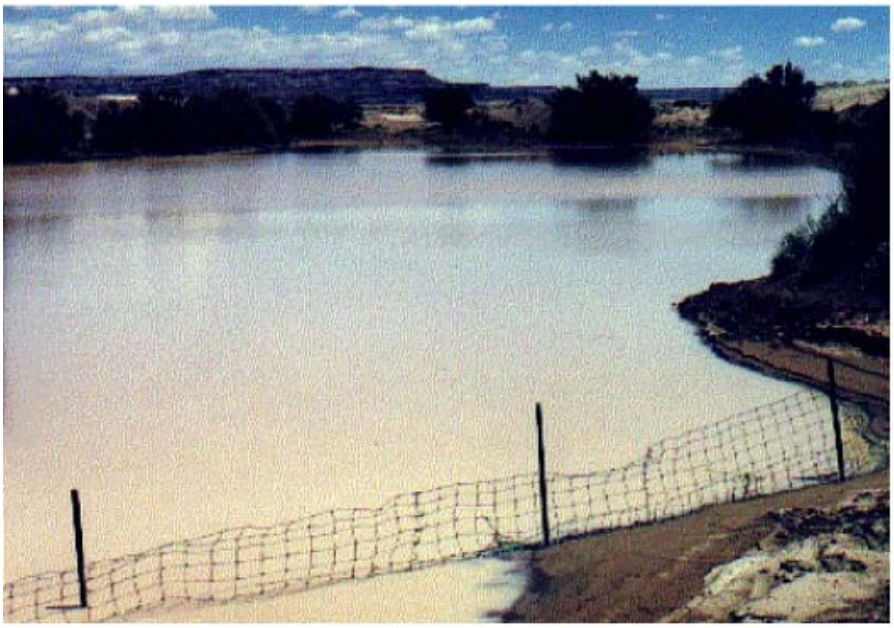
[1] Characteristics and origins of wastes mentioned in this study are more fully described in Chapter 3 of Volume I of this report (U.S. EPA 2006a).

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.

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

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}.

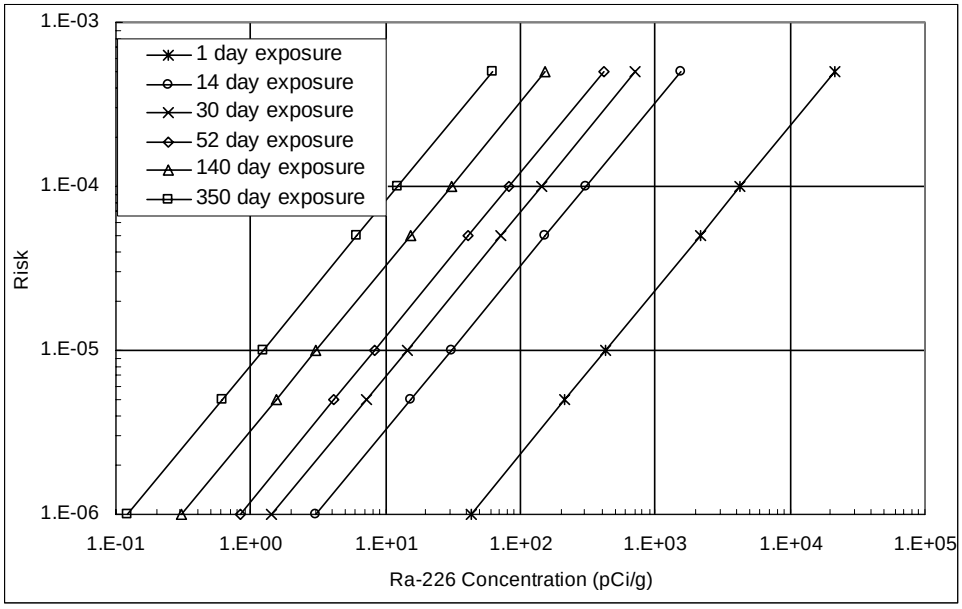


Table 3.3. Soil Screening Levels for External Exposure to Th-232

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}.

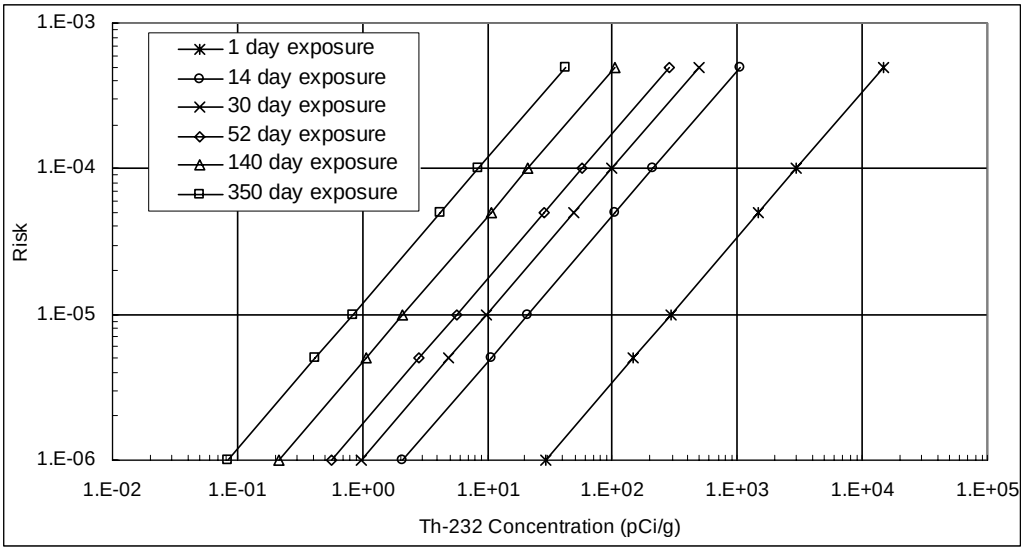


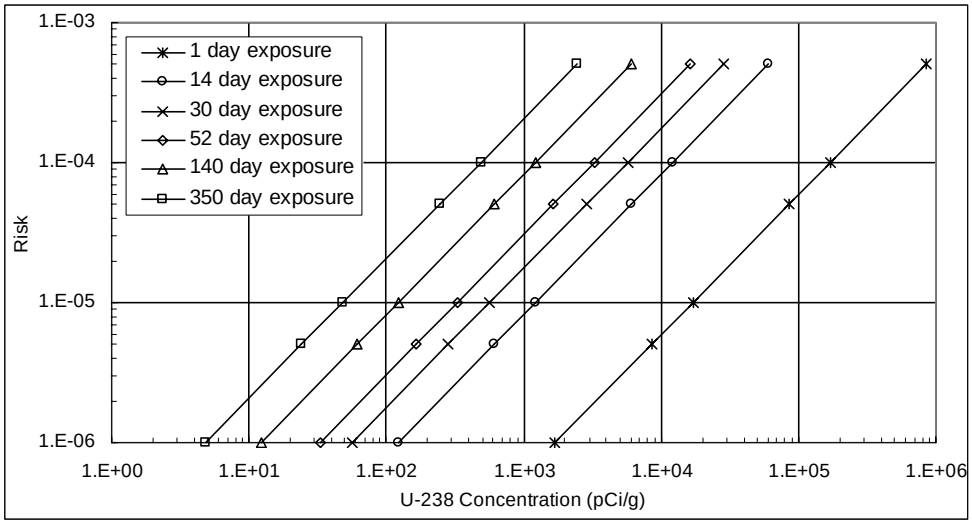
Table 3.4. Soil Screening Levels for External Exposure to Natural Uranium

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:

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

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.

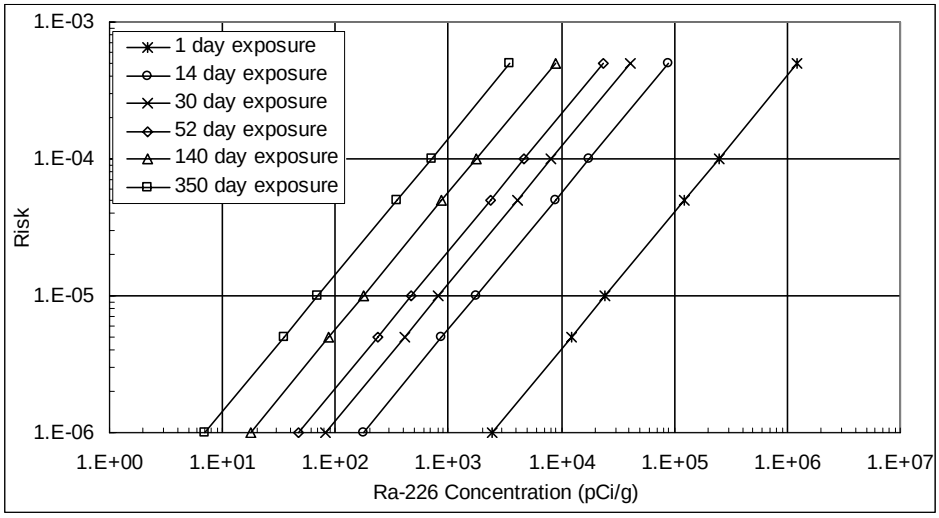


Table 3.6. Soil Screening Levels for Ingestion of Th-232 in Soil

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.

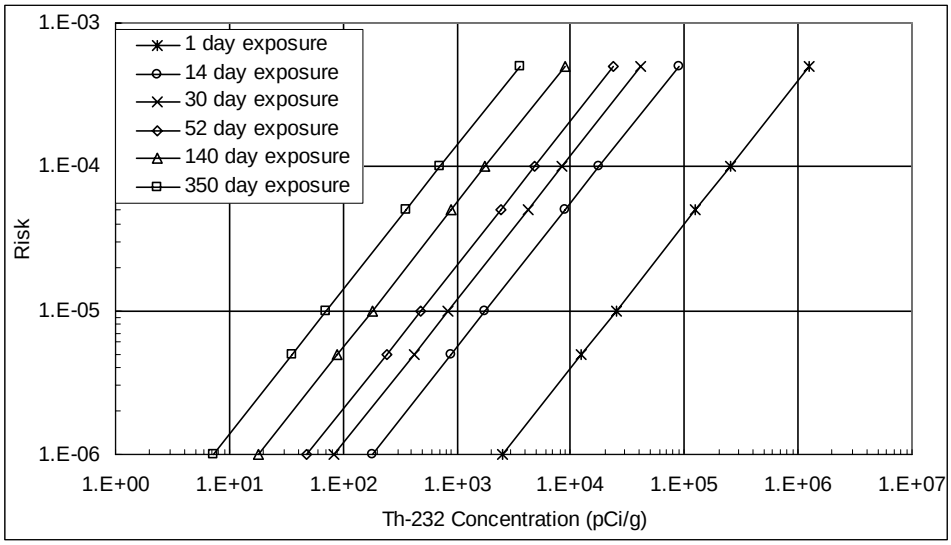
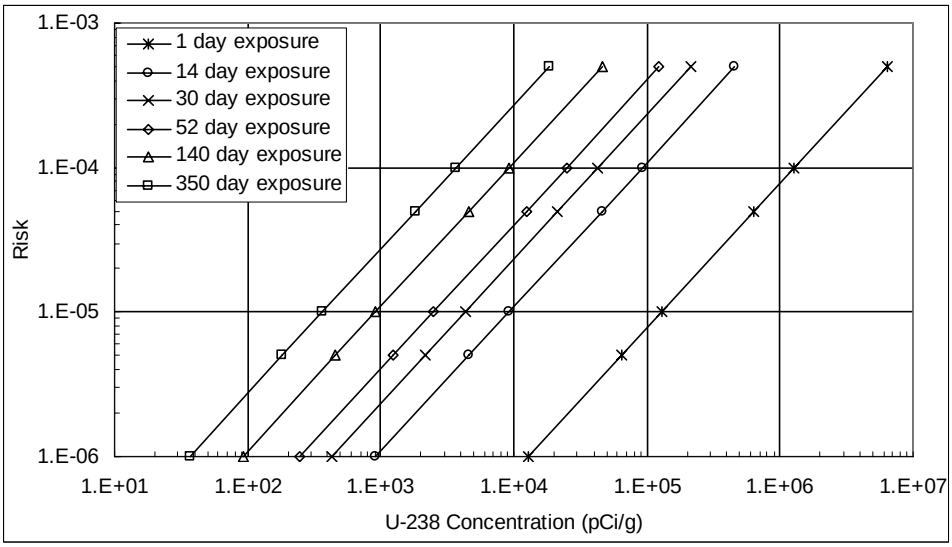


Table 3.7. Soil Screening Levels for Ingestion of Natural Uranium in Soil

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}.

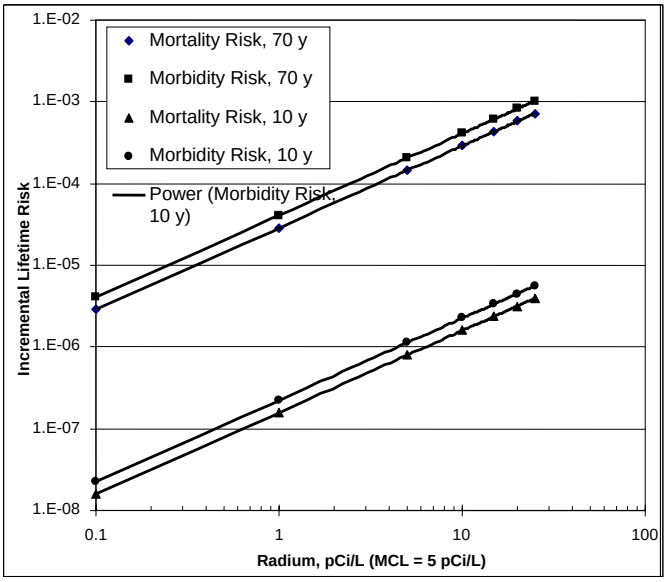


Figure 3.9. Cancer Risks from Lifetime and Recreational Exposures to Gross Alpha in Drinking Water: 70 Years, 365 Days/Year & 10 Years, 14 Days/Year Exposure

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}.

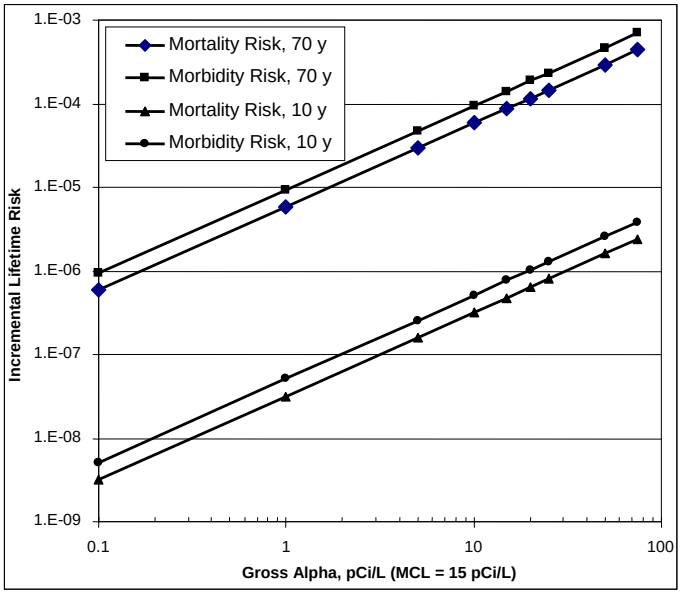


Figure 3.10. Cancer Risks from Lifetime and Recreational Exposures to Uranium in Drinking Water: 70 Years, 365 Days/Year and 10 Years 14 Days/Year Exposure

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}.

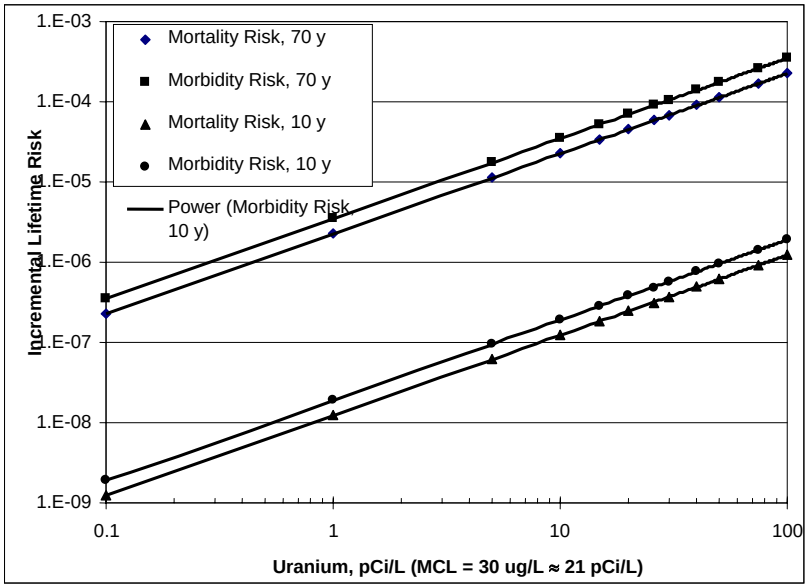


Table 3.11 estimates the potential lifetime cancer risk from radionuclides in the shallow Yazzie-312 Mine pit water (Panacea 2002), at concentrations measured before the pit was remediated. At these levels, long-term consumption of drinking water containing the radionuclides would be a significant health 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, $TR = (SF_O * C * IRW * EF * ED) / (BW * AT * 365 \text{ days/year})$

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

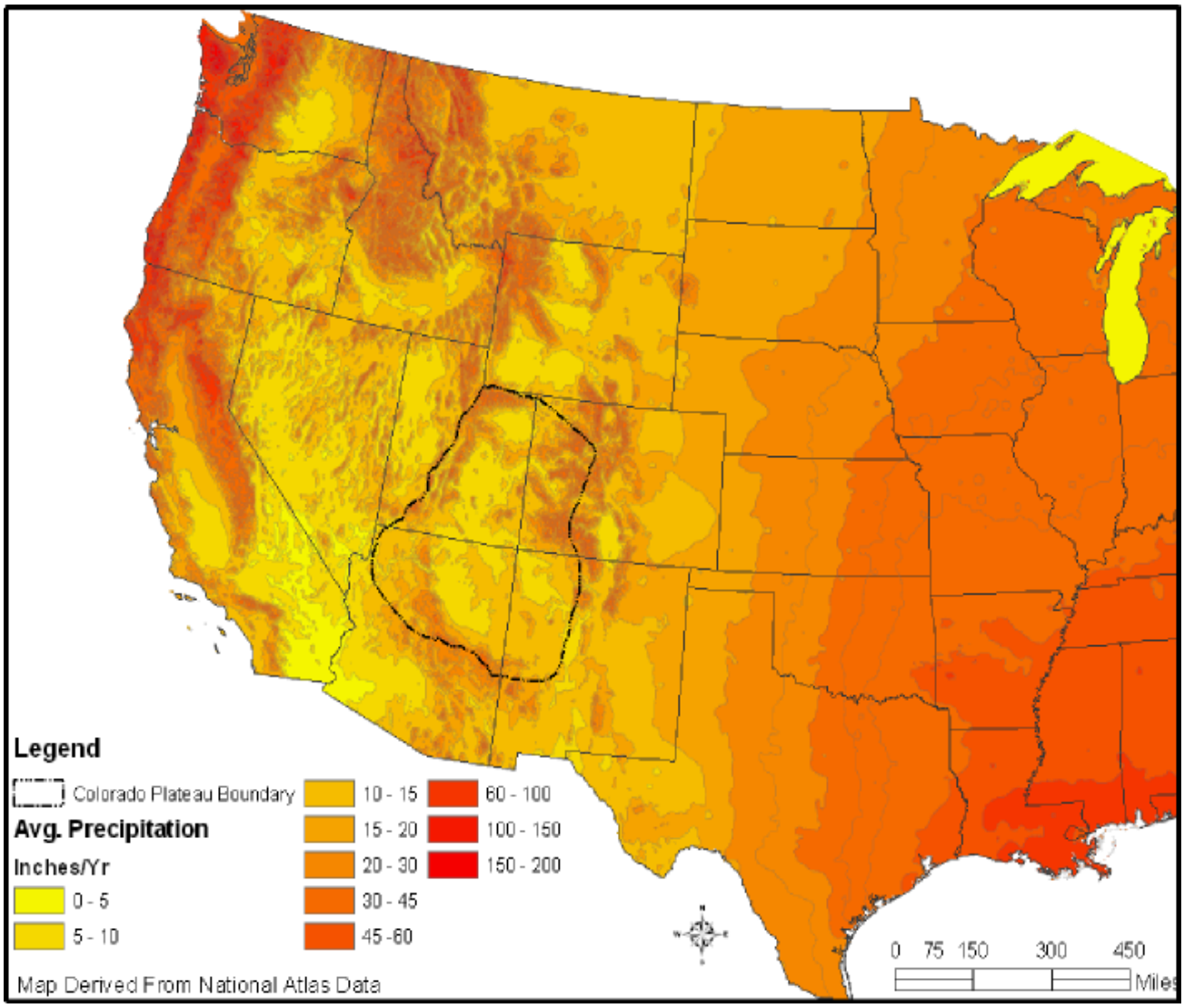
From Figure 3.11 below, the general annual precipitation range for the Colorado Plateau area is 5-15 inches (13-38 cm). This area also has high evapotranspiration rates. The 1983 EPA Report to Congress (U.S. EPA 1983a, b, and c) estimated that about 97 percent of the precipitation was lost to evapotranspiration. 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

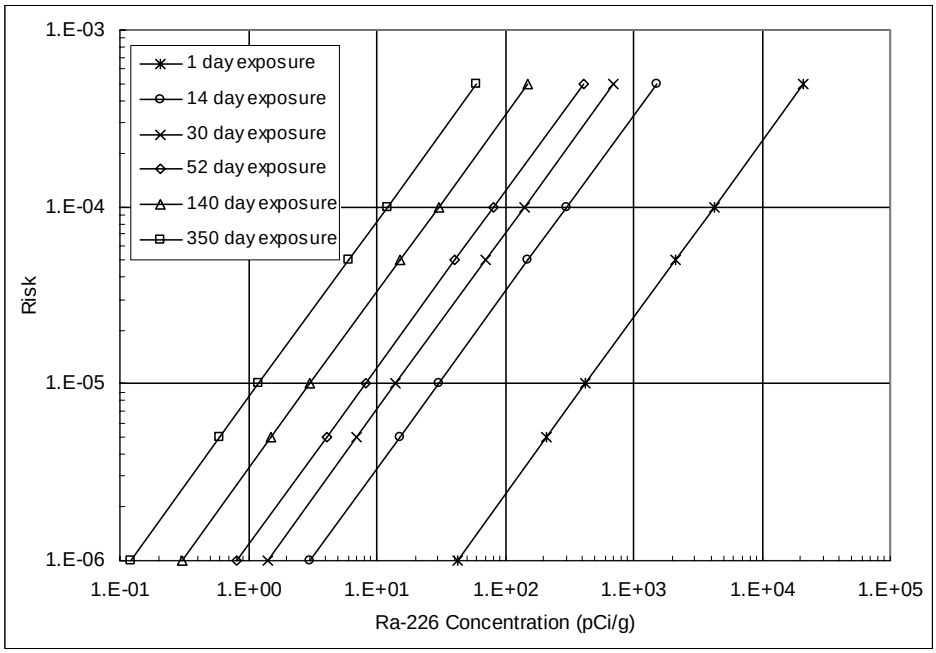


Figure 3.13. Multi-pathway Soil Screening Levels for Th-232

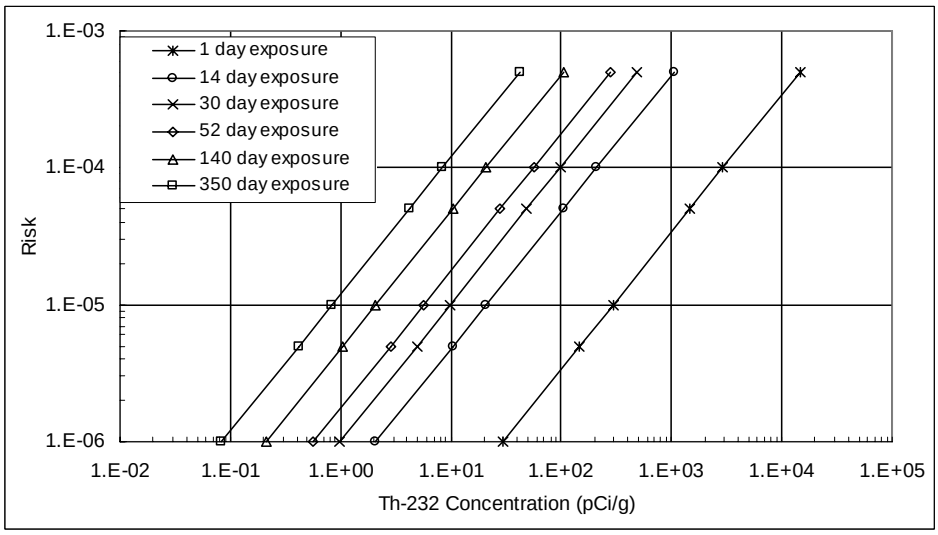


Figure 3.14. Multi-pathway Soil Screening Levels for U-238