

RAI Volume 3, Chapter 2.2.1.3.9, First Set, Number 1:

Justify the assumption that setting the activity of non-simulated radionuclide decay products in the aqueous phase to be equal to the activity of aqueous-phase parent radionuclides does not underestimate aqueous-phase activity.

Basis: In its saturated zone radionuclide transport model, the applicant simulates the transport of various radionuclide species in the aqueous phase. The radionuclides are subject to various processes, such as sorption, filtration, and matrix diffusion that attenuate their activity in groundwater. During transport simulations, these radionuclides are allowed to radiodecay and produce daughter products. Only a subset of the daughter products is directly simulated, however. The activities of the non-simulated daughter products are assumed, in the model, to be equivalent to the activity of the parent radionuclide in the aqueous phase.

Sorption, along with filtration and matrix diffusion can result in accumulation of radionuclides in the alluvial aquifer. These accumulated radionuclides will also decay, in addition to those radionuclides in the aqueous phase, to produce daughter radionuclides. The radioactive decay of parents accumulated in the alluvium can produce daughters whose activity in the ground water can exceed those of the dissolved parents, depending upon their sorption coefficients relative to that of the parent. Furthermore, the applicant has acknowledged that sorption coefficient distributions have been biased to lesser values than suggested from experimental evidence. An underestimation of sorption coefficients could lead to smaller quantities of accumulated radionuclides and lower doses.

The staff needs this information to determine whether radionuclide accumulation and decay products could result in an underprediction of radionuclide concentrations in groundwater. The staff needs information on the contribution to dose from accumulated radionuclides in the alluvial aquifer system to evaluate compliance with 10 CFR Part 63.114 (a)(3) and 63.311.

1. RESPONSE

This response clarifies the use of secular equilibrium in the total system performance assessment (TSPA) model, the saturated zone transport model, and the biosphere model. Secular equilibrium means that the activities of a parent radionuclide and its decay product are equal. In the TSPA model, aqueous activity concentrations of certain short-lived decay products in the saturated zone were assumed to be equal to the aqueous activity concentrations of their parent radionuclides (SAR Section 2.3.9.3.4.2.1 and SNL 2008a, Section 6.5.1.2). However, because a fraction of the mass of parent radionuclides may be sorbed to solids in the saturated zone alluvium, the sorbed masses of the parent radionuclides must also be considered when determining aqueous concentrations of the decay products. This effect has not been included in the TSPA model, but is discussed in this response and is shown in an impact assessment to have only a minor effect on estimates of repository performance.

1.1 INTRODUCTION

This response specifically addresses the effect of sorption on secular equilibrium for the following four decay chain pairs: $^{231}\text{Pa} \rightarrow ^{227}\text{Ac}$, $^{232}\text{Th} \rightarrow ^{228}\text{Ra}$, $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$, and $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$. The TSPA-LA model explicitly assumes secular equilibrium for the decay chain pairs except for $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$ (SAR Section 2.3.9.3.4.2.1 and SNL 2008a, Section 6.5.1.2). The $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$ pair must be considered in the biosphere model because radon is non-sorbing and has important inhalation pathways. The response first derives the sorption enhancement factor (Section 1.2) for the activity of a decay product in the groundwater relative to that of its parent radionuclide, including the effect of sorption, for use in this impact assessment. This sorption enhancement factor is equal to the ratio of the retardation factor of the parent radionuclide to the retardation factor of the decay product. The response next describes a radon emanation factor that is used in this impact assessment (Section 1.3). This radon emanation factor is the fraction of radon produced from radium sorbed onto the solid phase that enters the aqueous phase. The response justifies the sorption distribution coefficients and retardation factors (Section 1.4) used for the radionuclides in the decay chains considered in this response, and provides the means of the distributions of the sorption enhancement factors. The effect of enhanced radon activity in the groundwater on the biosphere dose conversion factor (BDCF) for ^{226}Ra is analyzed in Section 1.5 by examining the environmental transport and exposure pathways that account for exposure to radon, and evaluating where the effect may be important. A modified BDCF for the combined ^{226}Ra and ^{210}Pb activity is discussed in Section 1.6 and implemented in this impact assessment. Finally, the response describes the potential effect of the sorption enhancement factor on individual protection and groundwater protection results and concludes that the assumption of secular equilibrium in a rock–water system, as implemented in the TSPA model, has only a minor effect on the performance assessment results.

1.2 DERIVATION OF SORPTION ENHANCEMENT FACTOR

Secular equilibrium in radioactive decay chains occurs when the activity of the decay product becomes equal to that of the parent. Mathematically, it is written as

$$\lambda_i N_i = \lambda_{i-1} N_{i-1} \quad (\text{Eq. 1})$$

where

λ = radioactive decay constant [1/time]

N = number of moles present.

and the subscript i represents the decay product and $i-1$ represents the parent radionuclide. The time required for radioactive equilibrium to occur depends on the half-life of the decay product. When more than one phase is present, such as an aqueous phase and a solid sorbed phase, as occurs in porous medium transport, secular equilibrium occurs between the total amount of matter present in both phases. In this case, Equation 1 is written as

$$\lambda_i (N_{i,aq} + N_{i,s}) = \lambda_{i-1} (N_{i-1,aq} + N_{i-1,s}) \quad (\text{Eq. 2})$$

where the subscript aq refers to the dissolved aqueous phase and the subscript s refers to the sorbed solid phase. In a saturated environment, $N_{i,aq}$ can be written in terms of aqueous concentration by

$$N_{i,aq} = \frac{C_{i,aq} V_T \varepsilon}{MW_i} \quad (\text{Eq. 3})$$

where

- $C_{i,aq}$ = aqueous concentration [mass/water volume]
- V_T = arbitrary control volume [bulk volume]
- ε = porosity [pore volume/bulk volume]
- MW_i = molecular weight [mass/mole].

Also, the distribution coefficient K_d is defined as the ratio of the sorbed concentration (i.e., ratio of mass of a radionuclide sorbed to the mass of solid) divided by the aqueous concentration

$$K_{d,i} = \frac{N_{i,s} MW_i / [\rho_s (1 - \varepsilon) V_T]}{C_{i,aq}} = \frac{N_{i,s} MW_i / [\rho_b V_T]}{C_{i,aq}} \quad (\text{Eq. 4})$$

where

- ρ_s = solid density [mass per solid (matrix) volume]
- ρ_b = bulk density [mass per bulk volume].

Similar equations can be written for the parent radionuclide ($i-1$). Rearranging and substituting these equations into the total balance equation yields

$$\frac{\lambda_i C_{i,aq} R_{f,i}}{MW_i} = \frac{\lambda_{i-1} C_{i-1,aq} R_{f,i-1}}{MW_{i-1}} \quad (\text{Eq. 5})$$

where $R_f = 1 + \frac{\rho_b K_d}{\varepsilon}$ is the retardation factor. The term $\lambda_{i-1} C_{i-1,aq} / MW_{i-1}$ multiplied by the Avogadro constant is the activity concentration [activity/volume] of the parent radionuclide in groundwater and is calculated by the GoldSim component of the TSPA model. When secular equilibrium is assumed, Equation 5 indicates that the activity concentration of the decay product is equal to the activity concentration of the parent multiplied by $R_{f,i-1} / R_{f,i}$, the ratio of the parent retardation factor to the decay product retardation factor. The sorption enhancement factor, SEF , can thus be defined as

$$SEF_i = \frac{R_{f,i-1}}{R_{f,i}} \quad (\text{Eq. 6})$$

where

SEF_i = sorption enhancement factor for the decay product i

$R_{f,i-1}$ = retardation factor for the parent (predecessor) (dimensionless)

$R_{f,i}$ = retardation factor for the decay product (dimensionless).

Previously, this sorption enhancement factor was not accounted for in calculations of radionuclide concentrations of decay products in the saturated zone and biosphere when secular equilibrium was assumed.

As shown below, for cases in the analysis in which the retardation characteristics of the parent and decay product are similar, such as the case with ^{226}Ra and ^{210}Pb , omitting the factor has very little effect. In other cases, the sorption enhancement factor may be approximately 10, such as the case with ^{228}Ra and ^{227}Ac . In these cases, the omission has little consequence as well, as shown below. However, in the case of ^{222}Rn , the effect on concentration of radon in groundwater can be significant because radon does not sorb and thus its concentration can be significantly enhanced. Consequently, the effect of the enhanced aqueous radon concentration is explicitly considered in the context of the biosphere model, but is shown in this response to have a minor effect on estimates of repository performance.

1.3 THE RADON EMANATION FACTOR

The radon emanation factor (also known as emanation coefficient, emanating power or escape-to-production ratio) is the fraction of radon atoms released into a rock or soil pore space from a radium-bearing grain. The radon emanation factor is a parameter needed to calculate the ^{222}Rn activity concentration in groundwater in the impact assessment described in this response. The groundwater in the alluvial aquifer at the location of the hypothetical future pumping well may contain dissolved ^{222}Rn originating from decay of radionuclides released from the Yucca Mountain repository. Specifically, ^{222}Rn is produced from the decay of ^{226}Ra , which may be present in groundwater or sorbed on the surfaces of clastic sediments and secondary minerals of the alluvial deposits. Empirical evidence indicates that a significant fraction of ^{222}Rn produced from decay of ^{226}Ra associated with solids is retained in the solid phase in subsurface environments. The following three processes may occur after radon is released by the decay of radium associated with solids (Schumann 1993): it may travel a short distance and remain embedded in the same grain; it can travel across a pore space and become embedded in an adjacent grain; or it may be released into a pore space.

An analysis of the literature data was conducted as part of this response to develop a distribution of the radon emanation factor applicable to the Yucca Mountain site-specific conditions. This analysis considered both the experimental data and theoretical research. Measured radon emanation factors reported in the literature concern the fraction of radon atoms released into rock or soil pore space from a radium-bearing grain regardless of the distribution of radium within the grain, i.e., whether radium is distributed throughout the grain matrix or on the grain surface. The presence of radium in increased concentrations in grain surface coatings is known to increase the emanation factor relative to that for radium distributed throughout the grain (UNSCEAR 2000, p. 97). This characteristic is important because radium originating from Yucca Mountain will be

adsorbed to the surface of the mineral grains, thus enhancing the radon emanation factor, whereas naturally occurring radium may be either within the solid grain or on the grain surface.

The results of the theoretical research (Sasaki et al. 2004, 2005, and 2008; Bossus 1984; Thamer et al. 1981) led to the conclusion that in the case when radium is uniformly distributed within the solid grain, the maximum radon emanation factor of rocks and soils is 0.25. Consequently, experimental data with emanation factors noticeably below this maximum limit are considered as representing radium found primarily within solid grains, and are not applicable to the Yucca Mountain site-specific conditions.

Observations of emanation factors greater than the theoretical maximum were explained either by the surface distribution of radium or by the presence of nanopores on the surface of the solid grains (Sasaki et al. 2005 and 2008). The surface distribution of radium may occur in both uranium ores (tailings) and soils (Michel 1987). In many ores, uranium mineralization is related mainly to the secondary minerals, which results in the distribution of radium on the surfaces of the host rock grains. In fresh rocks, particularly granites, a significant amount (up to 30%) of the uranium (and radium) content of the rock can occur as inter-granular films. In weathered rocks and soils, uranium and thorium released during disintegration of the various minerals are readily adsorbed onto the surface of clay particles that are formed during the weathering process. Co-precipitation with iron oxide and formation of secondary minerals in pore spaces or fractures remove uranium and radium from solution and result in their distribution on grain surfaces. Consequently, most of the uranium ore and mill tailings data and selected soil data are applicable to the Yucca Mountain site-specific conditions.

The moisture conditions and grain size were considered in analysis of the existing experimental data because these two factors are known to affect the radon emanation factor (Sun et al. 1995; Schumann 1993; Sasaki et al. 2004). The emanation factor of a dry sample may be significantly lower than of a moist sample (Sun et al. 1995; Michel 1987; Sasaki et al. 2004; Prutkina et al. 1967). However, the emanation rates quickly reach the emanation rate of the saturated condition with moisture contents from 10% up to 30% (Sun et al. 1995). The data for the moist samples were considered when the information concerning the saturation conditions was available. Also, it was estimated (using data in Sasaki et al. 2004) that the ratio between the dry and moist emanation factor values would be small (1.3 or less) for the alluvial deposits at the boundary of the accessible environment.

The grain size of alluvial deposits was not directly accounted for in developing the emanation factor distribution. This provides some conservative bias because the emanation factor is smaller for the larger grains, and the median grain size of the alluvial deposits (greater than 1,000 μm) is noticeably larger than the grain size of the samples used in the experimental studies (e.g., Prutkina et al. 1967).

The experimental data used in developing the Yucca Mountain site-specific emanation factor distribution are summarized in Table 1. The experimental data include uranium ore samples, mill tailings, and soils.

Table 1. Summary of the Radon Emanation Experimental Data

Source	Sample Description	Number of Samples	Saturation Conditions	Radon Emanation Factor		
				Mean	Minimum	Maximum
Prutkina et al. 1967	uranium ore	9	fully saturated	0.39	0.24	0.6
Thamer et al. 1981	uranium ore	10	moist	0.38	0.2	0.55
Strong and Levins 1982	uranium mill tailings	2	fully saturated	0.3	0.29	0.31
Rogers et al. 1984	uranium mill tailings	8	saturated	0.29	0.20	0.4
Funtua et al. 1997	granite uranium ore	1	^a	0.31	^a	^a
Przylibski 2000	crystalline rocks	3	fully saturated	0.32	0.21	0.41
Rogers et al. 1988	soils	35	^a	0.26	0.20	0.44
Damkjaer et al. 1985	soils	28	3% to 70% by weight (average 23%)	0.29	0.20	0.61

^a Not available

The emanation factors for many different soil types were reported by Damkjaer et al. (1985) (53 samples) and Rogers et al. (1988) (56 samples). The soil data summarized in Table 1 are only for the subset of these samples with the emanation factors of 0.2 or greater. The 0.2 cutoff was selected based on the cumulative probability plots of the emanation factors. Both data sets exhibit a step change around this value, which may be interpreted as a change in the radium distribution (applicable conceptual model). This is also consistent with the theoretical maximum of 0.25. One soil sample from Damkjaer et al. (1985) with the emanation factor of 0.7 was not considered. This sample was diatomite, a very fine-grained sedimentary rock. This is consistent with the theoretical results of Sasaki et al. (2005), who concluded that the emanation factors exceeding 0.6 can only be explained by the presence of very small grains.

Table 1 also includes uranium ore data for moist uncrushed samples reported by Thamer et al. (1981). These samples have, on average, higher emanation factors than moist crushed, or dry uncrushed, or dry crushed samples. Seven samples with very low emanation coefficients (below 0.2) were not considered. The uranium mill tailings data summarized in Table 1 include 8 out of 12 samples reported by Rogers et al. (1984). Four samples with the emanation coefficients ranging from 0.1 to 0.17 were not considered.

The average emanation factor for the data in Table 1 for the saturated conditions is around 0.34. The average emanation factor is 0.29 for the data sets with variable saturation or unknown saturation. Because the radon emanation factor is greater under moist or saturated conditions, it can be conservatively assumed that these two data sets apply to the dry conditions. The average emanation factor for the saturated conditions can then be calculated using the wet-to-dry emanation factor ratio estimated for the alluvial deposits at the boundary of the accessible environment, as discussed above. The resulting emanation factor is 0.38 (0.29×1.3).

Based on the analysis of the literature data applicable to the Yucca Mountain site-specific conditions, the uncertainty distribution for the ^{222}Rn emanation factor is modeled by a truncated normal distribution with a mean of 0.40 and a standard deviation of 0.07. It is reasonable to truncate the uncertainty distribution for the radon emanation factor at 0.2 for the lower limit (at the minimum of the applicable experimental data) and 0.60 for the upper limit (at the maximum of the applicable experimental data). The value of 0.4 is a conservative estimate of the mean emanation factor because it represents the maximum average emanation factor reported in the literature. This uncertainty distribution of the emanation factor is similar to the distribution developed by Rogers et al. (1988), which was a normal distribution with a mean of 0.22 and a standard deviation of 0.07, except that the mean value is higher (0.4).

This radon emanation factor is used in conjunction with the radon SEF previously defined, which is discussed in more detail in Section 1.5. The radon SEF (Equation 6) is multiplied by the radon emanation factor to calculate the actual radon enhancement in the water. That is, the activity concentration of the decay product (i.e., radon) is equal to the activity concentration of the parent (i.e., radium) multiplied by the net radon enhancement factor, $NEF_{Rn} = SEF_{Rn} \times f$, where SEF_{Rn} is the sorption enhancement factor for the decay chain pair $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$, and f is the radon emanation factor.

1.4 DISTRIBUTION COEFFICIENTS AND SORPTION ENHANCEMENT FACTORS

As derived above, the sorption enhancement factor for a decay product in secular equilibrium with its parent (where secular equilibrium applies to the system consisting of the solid and aqueous phases) is the ratio of the retardation factor of the parent to that of the decay product. For the $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$, $^{231}\text{Pa} \rightarrow ^{227}\text{Ac}$, and $^{232}\text{Th} \rightarrow ^{228}\text{Ra}$ pairs, each element is expected to sorb extensively. Because they are highly sorbing, the sorption enhancement factors for each pair can be approximated simply by the ratios of the parent K_d to the decay product K_d . However, because radon does not sorb to rock, the sorption enhancement factor for $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$ must be presented in terms of the ratio of the retardation factors as described in Equation 6.

K_d distributions for Ra, Pa, and Th were developed in support of the TSPA-LA (SNL 2008b, Table A-4). However, K_d distributions were not developed for Pb or Ac, but are considered here. Ratios of K_d s are ideally determined from adsorption experiments involving each of the paired elements in the same experiment. Alternatively, they can be determined from separate adsorption experiments conducted under identical conditions. However, such experiments are generally rare or nonexistent for the $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$, $^{231}\text{Pa} \rightarrow ^{227}\text{Ac}$, and $^{232}\text{Th} \rightarrow ^{228}\text{Ra}$ pairs for the rock type and chemical conditions expected in the alluvium in the location of the reasonably maximally exposed individual (RMEI). Where data are sparse or unavailable, the K_d ratios may instead be approximated based on the sorption data of analogous elements and rock types or on the pertinent chemical characteristics of the chemical elements.

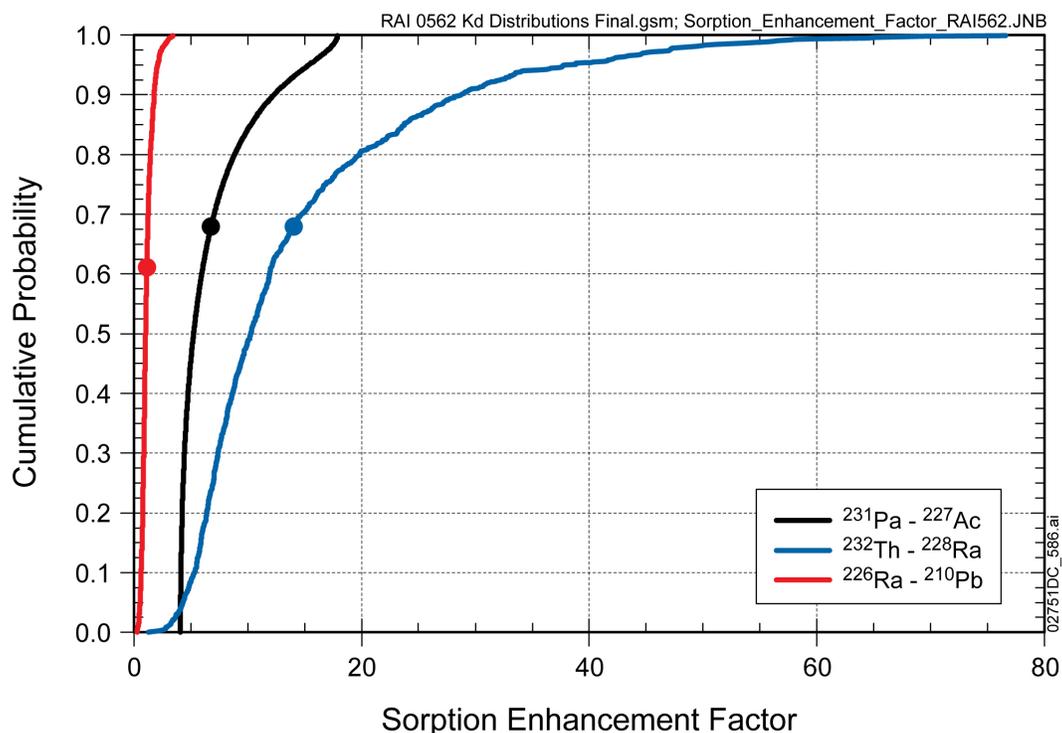
K_d values generally depend on the characteristics of the sorbent and the pH of the water. The rock type of the alluvium along the flow path is largely disaggregated tuffaceous material, and water in the alluvium is expected to have a pH range of approximately 7 to 8.5 (SNL 2008b, Section A7.5). Thus, the most relevant K_d measurements are those for tuff in the pH range of 7 to 8.5. In addition, when data for devitrified tuff are available, they are selected over other types of tuff because devitrified tuff makes up a major portion of the volcanic units exposed at the surface (SNL 2008b, Section A7.5).

K_d distributions for the parent radionuclides and decay products are listed in Table 2. A Latin hypercube sample (LHS) of size 1,000 was obtained from these distributions using GoldSim, with appropriate correlations as discussed below. Using this sample, the K_d ratio for each decay chain pair was computed, and the mean of the ratios (i.e., mean sorption enhancement factor) was determined. The resulting cumulative distributions of the sorption enhancement factors for the $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$, $^{231}\text{Pa} \rightarrow ^{227}\text{Ac}$, and $^{232}\text{Th} \rightarrow ^{228}\text{Ra}$ pairs are shown in Figure 1. The mean value of the ratio is used in this response to calculate the approximate effect that sorption enhancement has on radionuclide dose. The details of the sorption enhancement factor calculations are provided below.

Table 2. K_d Distributions for Pa, Ac, Th, Ra, and Pb on Tuff

Radioelement	K_d Distribution (mL/g)	Distribution	Rock Type	Source
Pa	1,000 to 10,000 $\mu = 5,500, \sigma = 1,500$	Normal, truncated at $\pm 3\sigma$	Devitrified tuff and alluvium	SNL 2008b (Table A-4)
	0 to 100 (but greater than 1,000 at pH>7)	Uniform	Devitrified tuff	Triay et al. 1997 (pp. 100, 137)
Ac	100 to 2,000 ^a	Uniform	Devitrified tuff	Triay et al. 1997 (p. 137)
Th	1,000 to 10,000 $\mu = 5,500, \sigma = 1,500$	Normal, truncated at $\pm 3\sigma$	Devitrified tuff and alluvium	SNL 2008b (Table A-4)
	100 to 2,000 ^a	Uniform	Devitrified tuff	Triay et al. 1997 (p. 137)
Ra	100 to 1,000	Uniform	Devitrified tuff and alluvium	SNL 2008b (Table A-4)
	100 to 500	Uniform	Devitrified tuff	Triay et al. 1997 (p. 137)
Pb	100 to 500	Uniform	Devitrified tuff	Triay et al. 1997 (p. 137)

^a Triay et al. (1997) indicate a K_d distribution similar to that of americium, which is presented as a uniform distribution from 100 to 2,000 mL/g for devitrified tuff.



NOTE: Mean values are shown by the circles.

Figure 1. Cumulative Distribution Function of the Sorption Enhancement Factor

1.4.1 Sorption Enhancement Factor for $^{231}\text{Pa} - ^{227}\text{Ac}$ Pair

To determine the K_d ratio for Ac, the K_d distribution for Pa from the TSPA is used with a K_d distribution for Ac obtained from Triay et al. (1997). For pH between 7 and 8.5, the K_d distribution for Pa used in the TSPA ranges between 1,000 and 10,000 mL/g for alluvium and devitrified tuff. This range is consistent with that reported by Triay et al. (1997) for devitrified tuff and a pH greater than 7 (Table 2). Triay et al. (1997, p. 137) equate the K_d distribution for Ac to that of americium (Am), given as a uniform distribution ranging from 100 to 2,000 mL/g (Table 2).

GoldSim was used to generate the cumulative distribution function for the ratio of Pa K_d to Ac K_d using the TSPA K_d distribution for Pa and the Triay et al. (1997) K_d distribution for Ac. A correlation coefficient of 1 was used, consistent with the correlation between K_d values for Pa and Am used in the TSPA (SNL 2008a, Section 6.5.3[a], Table 6-9[a]). The mean of the resulting cumulative distribution function for the ratio of Pa K_d to Ac K_d is 6.8. The SEF for ^{227}Ac is approximated by the ratio of Pa K_d to Ac K_d . The cumulative distribution of the sorption enhancement factor for ^{227}Ac is shown in Figure 1.

1.4.2 Sorption Enhancement Factor for $^{232}\text{Th} - ^{228}\text{Ra}$ Pair

The distribution of the ratio of Th K_d to Ra K_d is determined directly from the Th and Ra K_d distributions used in the TSPA (Table 2). GoldSim was used to generate the cumulative

distribution function for the ratio of Th K_d to Ra K_d from these K_d distributions assuming no correlation, as previously determined (SNL 2008b, Table A-6). The mean of the resulting cumulative distribution function for the ratio of Th K_d to Ra K_d is 14. Figure 1 shows the cumulative distribution of the sorption enhancement factor for ^{228}Ra .

The mean sorption enhancement factor of 14 for ^{228}Ra may be conservative because the K_d distributions recommended by Triay et al. (1997) suggest significantly lower enhancement. There is no difference in the minimum Th and Ra K_d values of Triay et al. (1997), and only a four fold difference in the maximum values. Thus, the Th/Ra K_d ratio distribution calculated from the TSPA K_d distributions for Th and Ra and used in the enhancement calculations may overestimate ^{228}Ra aqueous concentrations.

1.4.3 Sorption Enhancement Factor for $^{226}\text{Ra} - ^{210}\text{Pb}$ Pair

For the $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$ decay chain, K_d data from identical sorption experiments on elements analogous to Ra and Pb are available for direct approximation of the Ra/Pb K_d ratio. Triay et al. (1997) recommend identical K_d ranges and distributions for devitrified tuff for both Ra and Pb (100 to 500 mL/g, uniform). These distributions are based primarily on sorption data for barium (Ba) and nickel (Ni) onto Yucca Mountain devitrified tuff. Ba is an appropriate analogue for Ra because Ra and Ba are both alkaline-earth (Group IIA) metals and are expected to have similar sorption properties. Ni is an appropriate analogue for Pb because under the conditions of the saturated zone each of these metals has a +2 valence and a dominant neutral carbonate species ($\text{NiCO}_3(\text{aq})$ and $\text{PbCO}_3(\text{aq})$).

The K_d measurements for Ba and Ni involving Yucca Mountain devitrified tuff, which were carried out on similar tuff samples for three different Yucca Mountain water compositions (J-13, P-1, and H-3), were used to calculate a correlation between Ba and Ni K_d measurements. By averaging the sorption and desorption data for each element in each water composition for the devitrified tuff samples, a correlation (R value) of approximately 0.6 was determined.

For the enhancement calculations, GoldSim was used to generate the Ra/Pb K_d ratio distribution. This distribution was based on the Triay et al. (1997) K_d distributions for Ra and Pb and the correlation coefficient (0.6) of the Ba and Ni K_d data. The mean of the resulting Ra/Pb K_d cumulative probability distribution is 1.1. Figure 1 shows the cumulative distribution of the sorption enhancement factor for ^{210}Pb .

1.4.4 Net Enhancement Factor for Radium and Radon

As described in Section 1.3, for the $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$ decay chain, the net radon enhancement factor is $NEF_{Rn} = SEF_{Rn} \times f$, where SEF_{Rn} is the sorption enhancement factor for the decay chain pair $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$, and f is the radon emanation factor. For this analysis, a distribution for the radon sorption enhancement factor is derived from the radium K_d distribution (uniform distribution between 100 and 1000 mL/g, SNL 2008b, Table A-4), the distribution of bulk density ρ_b (SNL 2008a, Table 6-8, variable bulkdensity) and the total porosity $\varepsilon = 0.30$ for the saturated zone alluvium (SNL 2008a, Section 6.5.2.3). A sample of size 1000 was generated using these distributions to generate a distribution of radon retardation factors based on the linear retardation

coefficient $R_f = 1 + \frac{\rho_b K_d}{\varepsilon}$ (see Section 1.2). A distribution of sorption enhancement factors is then derived from the ratio of the sampled retardation factors R_f for ^{226}Ra to the value for ^{222}Rn , which is 1.0.

For the radon emanation factor calculations, GoldSim was used to generate an LHS sample of 1,000 values. The sampling was based on a normal distribution with a mean of 0.4, a standard deviation of 0.07, a minimum of 0.2, and a maximum of 0.6, as discussed in Section 1.3. The net radon enhancement factor is then generated from the product of the sorption enhancement factors and radon emanation factors. The mean of the resulting net radon enhancement factor cumulative probability distribution is 1,404. Figure 2 shows the cumulative distribution of the sorption enhancement factor and the net radon enhancement factor for the $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$ decay chain.

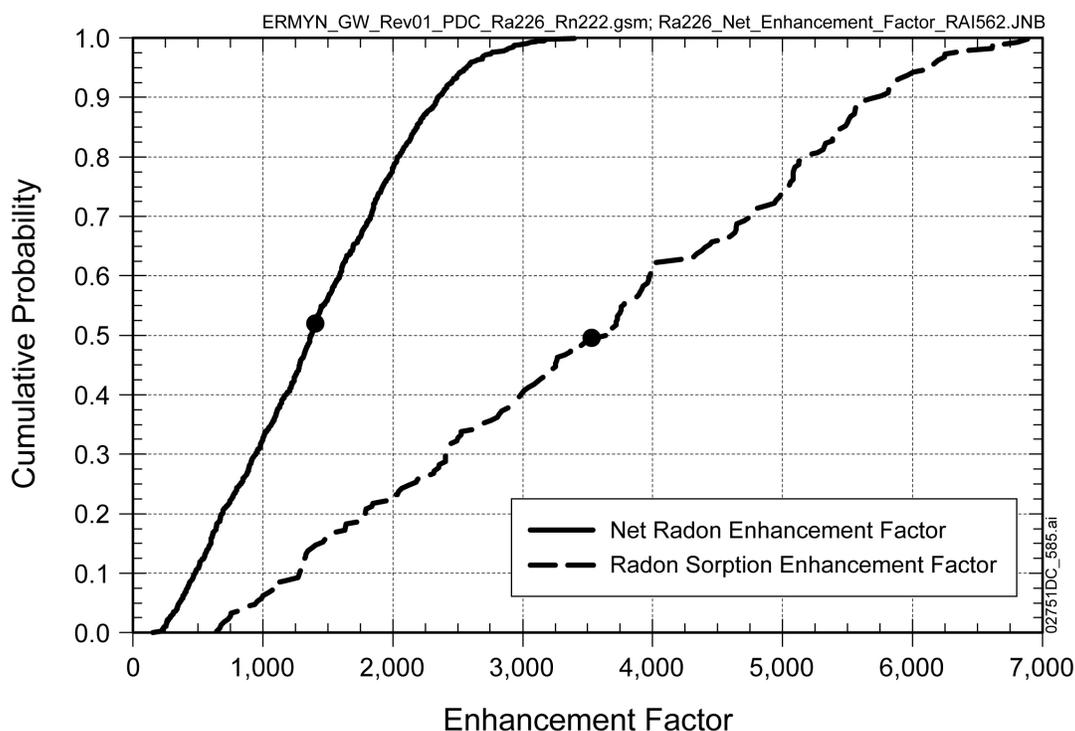


Figure 2. Cumulative Distribution Function of the Net Enhancement Factor for Radon

1.5 EVALUATION OF ENVIRONMENTAL TRANSPORT AND EXPOSURE PATHWAYS FOR THE GROUNDWATER ENRICHED IN RADON-222

This section discusses the behavior in the biosphere and radiological consequences of unsupported ^{222}Rn and other unsupported decay products in the ^{226}Ra decay chain. Unsupported means that there is an excess amount of decay products in the environmental medium, such as water or soil, that is not in radioactive equilibrium with, or not supported by, ^{226}Ra .

1.5.1 Radon Concentration in the Water in the Biosphere

When radon-enriched water reaches the biosphere, dissolved radon gas is released into the air. Radon is a noble gas that is soluble in water; however, the solubility of radon in water is relatively low. Radon's low solubility and its high vapor pressure mean that it strongly partitions into the air by diffusion (SAIC 1999, p. 1-1). Thus, when groundwater containing radon reaches the surface, the radon will inevitably be outgassed into the atmosphere until equilibrium is reached with the ambient air. This process of outgassing can be expedited by water aeration and is temperature dependent. Radon transport processes at the water-air interface, and their effect on radon and its decay product concentrations in the water, are discussed below.

1.5.1.1 Equilibrium Concentration of Radon in the Water at the Water–Atmosphere Interface

The main use of the groundwater in Amargosa Valley is for irrigation. There are three basic methods used to irrigate field crops, orchards, and gardens: surface irrigation, drip systems, and overhead sprinkler systems. The irrigation method predominantly used in Amargosa Valley is overhead sprinkler irrigation. Surface irrigation includes ditch and furrow irrigation and other flood methods that saturate part, or all, of the soil surface. Drip irrigation includes the use of bubblers, drip emitters, drip tubing, micro sprays, or other methods that deliver water to the soil surface at or near the base of plants.

During overhead sprinkler irrigation, radon dissolved in the water will escape to the atmosphere because the concentration of radon in the outdoor air is relatively low compared to that of the groundwater. During irrigation by other methods, such as drip and flood irrigation, radon dissolved in water would also escape to the atmosphere, but the transfer efficiency is likely to be less than that for the overhead sprinkler irrigation. However, following irrigation by any method, the remaining radon would quickly escape from the soil water by gas evasion across the water-air interface.

Under equilibrium conditions, the concentration of radon in water is directly proportional to the partial pressure of radon in the air above the water. This relationship is referred to as Henry's Law, which can be expressed as

$$p = \frac{k_H x}{P_T} \quad (\text{Eq. 7})$$

where

- p = mole fraction of ^{222}Rn in air (moles of ^{222}Rn per mole of air)
- x = mole fraction of ^{222}Rn in water (moles of ^{222}Rn per mole of water)
- k_H = Henry's Law constant (atm)
- P_T = total pressure (usually equal to 1 atm).

The Henry's Law constant for radon in water at 20°C is 2.26×10^3 atm (SAIC 1999, p. 2-2). Because of this large Henry's Law constant, radon easily transfers from water to air. Using Henry's Law, it can be calculated that for the typical outdoor radon concentration of 10 Bq/m^3 (SNL 2007a, p. 6-96), the concentration of radon gas in water at equilibrium would be about 5.9 Bq/m^3 . This typical outdoor radon concentration agrees well with the geometric mean for ambient radon concentration of 13.1 Bq/m^3 obtained in the regional study carried out in Nevada (NRC 1999, pp. 44 to 45). The air parameters used in this calculation are: the molecular mass of air = $28.97 \times 10^{-3} \text{ kg/mole}$ and the air density at 20°C = 1.204 kg/m^3 , which give 41.6 mole/m^3 . For ^{222}Rn , 10 Bq/m^3 equals to $7.91 \times 10^{-18} \text{ mole/m}^3$, which gives the mole fraction of ^{222}Rn in air, p , equal to $1.90 \times 10^{-19} \text{ mole } ^{222}\text{Rn per mole air}$. The mole fraction of ^{222}Rn in water, x , can thus be calculated from Equation 7 to be $8.42 \times 10^{-23} \text{ mole } ^{222}\text{Rn per mole water}$, which is equivalent to 5.9 Bq/m^3 . This equilibrium concentration of ^{222}Rn in the water is relatively low compared with the naturally occurring concentrations of radon in groundwater that have been measured in the area, which are in the range from thousands up to a hundred thousand Bq/m^3 (Holloway et al. 1989, Table 3). Thus, even though the concentration of radon can be high in the groundwater, upon entering the biosphere, most of that radon will be outgassed. Based on the contribution of ^{226}Ra to the mean annual dose (SAR Figure 2.4-20), and the mean value of the BDCF for ^{226}Ra (SNL 2007a, Table 6.11-8), the effective ^{226}Ra concentration in the groundwater can be estimated to be approximately 0.7 Bq/m^3 for the highest dose from this radionuclide for 1,000,000 years after repository closure. Assuming the radon net enhancement factor of 1,400, the corresponding concentration of ^{222}Rn would be approximately $1,000 \text{ Bq/m}^3$. Considering that the concentration of naturally occurring radon in the groundwater is likely to be greater, the radon transport processes that depend on the concentration gradient will be primarily controlled by the naturally occurring radon, rather than by the radon of Yucca Mountain repository origin.

1.5.1.2 Radon Flux from the Water

The rate of radon outgassing from water depends on the mechanics of the water delivery method and on the physical characteristics of radon diffusion out of the water. The rate of radon exchange across the water-air interface can be calculated by using the stagnant boundary layer model (Liss and Slater 1974). The model assumes a well-mixed atmosphere and a well-mixed water layer, where the transport is primarily controlled by turbulent diffusion, separated by the stagnant film at the water-air interface, where transport is controlled by molecular diffusion. The rate of transfer across the stagnant film at the water-air interface is due to the concentration gradient, and can be described by Fick's Law.

Using Fick's Law, the ^{222}Rn flux density, F , can be expressed as

$$F = \frac{D_w \times \frac{\Delta A}{\lambda}}{z} \quad (\text{Eq. 8})$$

where

F = radon flux density, atoms/ m^2/s

D_w = molecular diffusion coefficient for water ($1.2 \times 10^{-9} \text{ m}^2/\text{s}$)

- ΔA = ^{222}Rn activity concentration change across the stagnant film (Bq/m^3)
 λ = radioactive decay constant (2.1×10^{-6} 1/s)
 z = thickness of the stagnant film (m).

The concentration change across the stagnant film can be calculated by assuming that the water at the top of the film is in equilibrium with the atmosphere (e.g., 5.9 Bq/m^3 equilibrium concentration calculated in Section 1.5.1.1), and, initially, the bottom of the layer would have the same concentration of ^{222}Rn as the groundwater. For a unit concentration of ^{226}Ra in the groundwater (1 Bq/m^3) and the assumed net radon enhancement factor of 1,400, ^{222}Rn concentration would be $1,400 \text{ Bq/m}^3$, so the concentration change across the stagnant film would effectively be $1,400 \text{ Bq/m}^3$. For the stagnant film thickness of $50 \mu\text{m}$ (Church and Sarin 2008, p. 35), the initial radon flux density would be approximately $1.6 \times 10^4 \text{ atoms/m}^2/\text{s}$ ($3.4 \times 10^{-2} \text{ Bq/m}^2/\text{s}$). The net radon enhancement factor used in the calculation is the product of the radon sorption enhancement factor and the radon emanation factor, as described in Section 1.4.4.

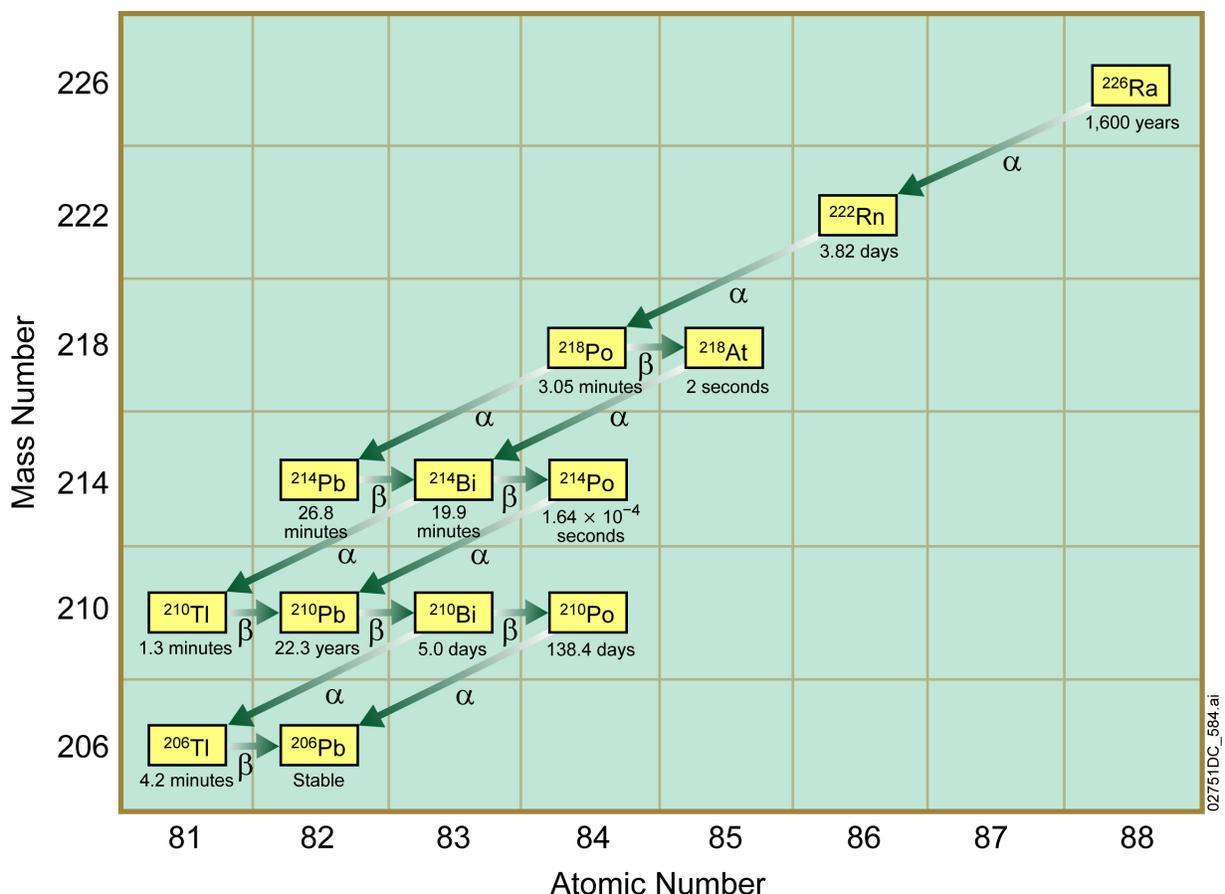
The radon flux density value calculated above can be put into perspective by considering the typical amount of daily irrigation used in the biosphere model, which is a few mm per day. Mean daily irrigation rate for the non-leafy (other) vegetables is about 8 mm/day ($0.008 \text{ m}^3/\text{m}^2/\text{day}$) (the highest of all crop types in the biosphere model). For 1 Bq/m^3 of ^{226}Ra in the groundwater and the assumed net radon enhancement factor of 1,400, there would be 11 Bq of ^{222}Rn in groundwater used per day per m^2 of soil. The corresponding rate of ^{222}Rn addition is $5.3 \times 10^6 \text{ atoms/m}^2/\text{day}$ or $8.2 \times 10^5 \text{ atoms/m}^2/\text{s}$ for an irrigation event lasting 0.179 hr/day (Table 6), the rate that is comparable with the initial radon flux density across the stagnant film. Therefore, molecular diffusion across the stagnant film would be a relatively effective mechanism of radon removal from that water.

Aeration hastens the diffusion process by providing a larger air/water surface area and a higher degree of turbulence. This technique has been reported to rapidly remove up to 99% of radon from the water (SAIC 1999, pp. 2-2 to 2-3). Spray is also very effective in removing radon from the water. For example, showering removes about 70% of radon from the water (Hopke et al. 1996, Table 2.1; Nazaroff et al. 1987, Table 1).

Radon outgassing from the irrigation water would continue on the ground. The thickness of the stagnant film of $50 \mu\text{m}$ used in the calculation above can be compared with the water film thickness around soil particles. $50 \mu\text{m}$ would be an upper bound for the water film thickness on soil particles; the more likely values would be between 1 and $10 \mu\text{m}$ (Pidwirny and Draggan 2006, Figure 81-1). Because the radon flux density is inversely proportional to the thickness of the stagnant film, the diffusion of radon from the water on the soil grains in contact with the atmosphere would be faster than that calculated above. On the other hand, the diffusion from the water film on the soil grains in contact with the soil gas would be slower because soil gas typically contains high concentrations of naturally occurring radon (NRC 1999, p. 25), which would reduce the radon concentration gradient. In summary, the outgassing of radon from the water exposed to the ambient air and to the soil gas involves many transport processes, but it is clear that most of the radon will outgas from the water.

1.5.1.3 Radon and Radon Decay Product Concentrations in the Water Used in the Biosphere

Once the radon-enriched groundwater is exposed to the atmosphere, radon will quickly diffuse out of the water across the water–air interface. However, its decay products will remain in the water and continue to decay. Figure 3 shows the decay chain of ^{226}Ra , ending with the stable isotope of lead, ^{206}Pb .



Source: Based on data by Shlein (1992, Table 8.6).

Figure 3. Radioactive Decay Chain of ^{226}Ra

Although radon gas would largely escape from the groundwater in the biosphere, its short-lived decay products that were in radioactive equilibrium with radon at the time of water withdrawal from a well would be left in the water. These decay products would continue to progress relatively quickly through the remainder of the decay chain, effectively ending with ^{210}Pb . It can be calculated that if radon did not escape, for every becquerel of ^{222}Rn and its short lived decay products, there would be 4.7×10^{-4} Bq of ^{210}Pb produced. These calculations are shown in Table 3.

Table 3. Calculation of ^{210}Pb Activity from the Decay of ^{222}Rn

Radionuclide	Half-life (s)	Radioactive Decay Constant (1/s)	Fraction of Decays	Number of Atoms per 1 Bq of ^{222}Rn and Decay Products	Activity (Bq)
^{222}Rn	3.30E+05	2.10E-06	1	4.77E+05	1
^{218}Po	183	3.79E-03	1	264.0	a
^{214}Pb	1,608	4.31E-04	0.9998	2,319.4	a
^{218}At	2	3.47E-01	0.0002	0.0005771	a
^{214}Bi	1,194	5.81E-04	1	1,722.6	a
^{214}Po	1.64E-04	4.23E+03	0.99979	0.0002366	a
^{210}Tl	78	8.89E-03	0.00021	0.0236313	a
^{210}Pb (100% ^{222}Rn decay)	7.04E+08	9.85E-10	1	4.81E+05	4.7E-04
^{210}Pb (10% ^{222}Rn decay)	7.04E+08	9.85E-10	0.1	5.20E+04	5.1E-05

^a Not calculated, but in equilibrium with ^{222}Rn .

Although evidence indicates that almost all radon will escape from the water in the biosphere, one can make a conservative assumption that 10% of radon will remain in the water and will continue through its decay chain. This will result in about 5.2×10^4 atoms or 5.1×10^{-5} Bq of ^{210}Pb per 1 Bq of ^{222}Rn initially present in the groundwater.

Considering the very short half-life of the short-lived radon decay products, relative to many processes occurring in the biosphere, it can be assumed, for the purpose of modeling biosphere transport processes, that all radon decay products remain in the water, decay instantaneously to ^{210}Pb , and contribute to the activity concentration of this radionuclide in the water.

The modeling of the radiological impact of the radon enhancement in the groundwater can be carried out by considering the effect of unsupported radon release into the air as a part of the biosphere model for ^{226}Ra and the effect of ^{210}Pb from decay of unsupported ^{222}Rn as a part of the biosphere model for ^{210}Pb . The concentration of ^{210}Pb in the water in the biosphere would then be a sum of the supported ^{210}Pb present in the water at the time of water withdrawal from a well and ^{210}Pb from the decay of excess unsupported radon (^{222}Rn). The ^{210}Pb concentration in the water can be expressed as

$$C_{w_{\text{Pb-210}}} = C_{w_{\text{Pb-210, supported}}} + C_{w_{\text{Pb-210, unsupported}}} \quad (\text{Eq. 9a})$$

$$C_{w_{\text{Pb-210, supported}}} = C_{w_{\text{Ra-226}}} SEF_{\text{Pb}} \quad (\text{Eq. 9b})$$

$$C_{w_{\text{Pb-210, unsupported}}} = C_{w_{\text{Ra-226}}} \times NEF_{\text{Rn}} \times \kappa \quad (\text{Eq. 9c})$$

$$C_{W_{Pb-210}} = C_{W_{Ra-226}} SEF_{Pb} \left(1 + \frac{1}{SEF_{Pb}} NEF_{Rn} \times \kappa \right) \quad (\text{Eq. 9d})$$

where

$C_{W_{Pb-210}}$ = concentration of ^{210}Pb in the water (Bq/m^3)

$C_{W_{Ra-226}}$ = concentration of ^{226}Ra in the water (Bq/m^3)

SEF_{Pb} = lead sorption enhancement factor (dimensionless)

$SEF_{Pb} = \frac{R_{f, Ra}}{R_{f, Pb}}$; where $R_{f, Ra}$ and $R_{f, Pb}$ are the retardation factors for radium and lead, respectively

$R_{f, Ra}$ = retardation factor for radium (dimensionless)

$R_{f, Pb}$ = retardation factor for lead (dimensionless)

NEF_{Rn} = net radon enhancement factor (dimensionless)

$NEF_{Rn} = SEF_{Rn} \times f = \frac{R_{f, Ra}}{R_{f, Rn}} \times f$; where SEF_{Rn} is the radon sorption enhancement factor; $R_{f, Ra}$ and $R_{f, Rn}$ are the retardation factors for radium and radon, respectively; and f is the radon emanation factor

κ = conversion factor – activity of ^{210}Pb resulting from a complete decay of all the short-lived decay products of ^{222}Rn and some fraction of ^{222}Rn (dimensionless; $\text{Bq}_{\text{Pb-210}}/\text{Bq}_{\text{Rn-222}}$); equal to 5.1×10^{-5} Bq of ^{210}Pb per 1 Bq of ^{222}Rn in the groundwater for 90% of ^{222}Rn release.

The net radon enhancement factor represents the $^{222}\text{Rn}/^{226}\text{Ra}$ activity ratio in the water, assumed for the purpose of biosphere transport and exposure calculations in Section 1.5 to be equal to 1,400 (see Figure 2). The mean lead sorption enhancement factor is 1.1 (see Figure 1). For the conversion factor, κ , equal to 5.1×10^{-5} , the term in the parentheses in Equation 9d is then equal to 1.065. This term quantifies the increase in the concentration of ^{210}Pb from unsupported ^{222}Rn relative to ^{210}Pb concentration initially in the groundwater.

1.5.2 Surface Soil Submodel

1.5.2.1 Concentration of Lead-210 in Surface Soil

Irrigation with water containing unsupported ^{222}Rn will, in the long term, cause the buildup of ^{210}Pb in the surface soil. The concentration of ^{210}Pb per unit activity concentration of unsupported ^{222}Rn can be calculated by assuming that some fraction of ^{222}Rn , and 100% of the short-lived radon decay products, will continue to decay to ^{210}Pb . As shown in Section 1.5.1.3, for 90% of radon release, there will be about 5.1×10^{-5} Bq of ^{210}Pb produced per 1 Bq of ^{222}Rn in the groundwater.

The concentration of ^{210}Pb in soil from unsupported ^{222}Rn can be calculated as part of the BDCF calculations for ^{210}Pb following the methods used in the biosphere model (SNL 2007a, Equations 6.4.1-3 and 6.4.1-5). To calculate the BDCF for ^{210}Pb that includes the contribution from unsupported ^{222}Rn , the concentration of ^{210}Pb in the water would be adjusted to account for the additional ^{210}Pb , as shown in Section 1.5.1.3.

The concentrations of ^{226}Ra and ^{210}Pb in the soil are shown for comparison in Table 4 using the ^{226}Ra activity concentration in the water of 1 Bq/m^3 , the assumed net radon enhancement factor of 1,400, and the mean lead sorption enhancement factor of 1.1.

Table 4. Comparison of ^{226}Ra and ^{210}Pb Concentrations in the Soil per Unit Activity Concentration of ^{226}Ra in the Groundwater

Crop Type	Concentration in Soil per Unit Mass, Bq/kg			
	^{226}Ra (initially in groundwater) ^a	Supported ^{210}Pb (from decay of ^{226}Ra) ^a	^{210}Pb (initially in groundwater) ^b	Unsupported ^{210}Pb (from decay of unsupported ^{222}Rn) ^c
Garden crops	0.35	0.24	0.13	8.3×10^{-3}
Field crops	1.8	1.5	0.27	1.8×10^{-2}

^a Based on the biosphere model run for ^{226}Ra ; mean garden and field soil concentrations (SNL 2007a).

^b Based on the biosphere model run for ^{210}Pb ; mean garden and field soil concentrations (SNL 2007a) multiplied by the lead sorption enhancement factor of 1.1.

^c Calculated based on the biosphere model results for ^{210}Pb (SNL 2007a) for ^{210}Pb concentration in the water calculated using Equation 9d with a net radon enhancement factor of 1,400 and a lead sorption enhancement factor of 1.1.

Based on the soil concentrations presented in Table 4, decay of unsupported ^{222}Rn in the irrigation water contributes little ^{210}Pb to the concentrations of this radionuclide in the soil compared to the concentrations of supported ^{210}Pb .

1.5.2.2 Concentration of Lead-214 and Bismuth-214 in the Surface Soil

^{214}Pb and ^{214}Bi are the two short-lived radionuclides in the radon decay chain that are used for the assessment of doses from external exposure. ^{214}Bi is the primary contributor to the external exposure contribution to the BDCF for ^{226}Ra , and ^{214}Pb contributes almost as much (SNL 2007a, Table 6.4-4). In addition to ^{214}Pb and ^{214}Bi from the decay of ^{226}Ra , unsupported ^{222}Rn in the irrigation water will cause a short-term increase of these radionuclides in the soil, which is evaluated in this response.

Concentrations of ^{214}Pb and ^{214}Bi in the soil per unit activity concentration of ^{222}Rn are calculated differently from ^{210}Pb because the former radionuclides are short-lived ($T_{1/2} = 19.9 \text{ min}$ for ^{214}Bi ; $T_{1/2} = 26.8 \text{ min}$ for ^{214}Pb) and will not build up in the soil. The instantaneous activity concentration of these radionuclides is controlled by the decay of ^{222}Rn in the soil because of its much longer half-life (3.82 days) compared with that of ^{214}Pb and ^{214}Bi . The activity of ^{214}Pb and ^{214}Bi added to, or produced in, the soil in an irrigation event will have essentially decayed after about three weeks.

To evaluate the impact of the temporary increase of ^{214}Bi activity in the surface soil following an irrigation event, the short-term activity increase in the soil was converted to the long-term averages by combining the activity added during the individual irrigation events and averaging it over a period of one year. Only then can a comparison be made with the other quantities calculated in the biosphere model from supported ^{222}Rn , which are based on annual average conditions. Following irrigation with the water containing radon and its decay products, the radionuclides will continue to decay, producing atoms of ^{214}Pb and ^{214}Bi . The number of atoms of ^{214}Pb and ^{214}Bi produced in a year in the soil can be calculated by using the number of atoms for ^{222}Rn and its short-lived decay products in transient equilibrium with ^{222}Rn . The results are presented in Table 5 under the assumption that 10% of ^{222}Rn and 100% of its short-lived decay products remain in the irrigation water. The average activity concentration in the soil can be calculated from the crop irrigation parameters (SNL 2007a, Table 6.3-3), assuming that the net radon enhancement factor is 1,400, and the irrigation water is distributed in a 25-cm layer of surface soil. The results of these calculations are also presented in Table 5.

Table 5. Calculation of Average Concentration of ^{214}Pb and ^{214}Bi in the Soil

Crop Type	Average Annual Irrigation ^a (m/yr)	Number of Atoms Produced and Decayed in Soil ^b (atoms/yr)		Annual Average Number of Decays ^c (decays/s = Bq)		Average Activity Concentration in Soil ^d (Bq/kg)	
		^{214}Pb	^{214}Bi	^{214}Pb	^{214}Bi	^{214}Pb	^{214}Bi
Garden crops	0.91	6.6E+07	6.4E+07	2.1E+00	2.0E+00	5.6E-03	5.4E-03
Field crops	1.78	1.3E+08	1.3E+08	4.1E+00	4.0E+00	1.1E-02	1.1E-02
Garden crops ^e	0.91	f		f		0.35 ^e	
Field crops ^e	1.78	f		f		1.8 ^e	

^a Average annual irrigation rate was taken from SNL 2007a, Table 6.3-3.

^b The number of atoms of ^{214}Pb and ^{214}Bi produced and decayed in the soil was calculated by adding the decay products from 10% of the ^{222}Rn atoms added to soil with the irrigation water to 100% of the short-lived decay products in the decay chain up to ^{214}Pb and ^{214}Bi , respectively, using the values in Table 3.

^c The annual average number of decays per second, i.e., the average activity, was calculated by assuming that all the ^{214}Pb and ^{214}Bi atoms produced in the soil will decay.

^d The annual average concentration of ^{214}Pb and ^{214}Bi in the soil was calculated by distributing the average activity of these radionuclides over the 25-cm thickness of the surface soil, whose density was 1,500 kg/m³.

^e These ^{214}Pb and ^{214}Bi concentrations in the soil for garden and field crops are calculated based on equilibrium with ^{226}Ra , i.e., have the same activity concentration as ^{226}Ra – see Table 4.

^f Not calculated.

For comparison, Table 5 (rows 3 and 4) includes the activity concentration of ^{214}Pb and ^{214}Bi in the surface soil for garden and field crops, as modeled in the biosphere model (see Table 4), assuming secular equilibrium with ^{226}Ra . This average activity concentration of ^{214}Pb and ^{214}Bi from the decay of ^{226}Ra accumulated in the soil is about two orders of magnitude greater than that from the decay of unsupported ^{222}Rn . Therefore, it is appropriate to not include the temporary increase of ^{214}Pb and ^{214}Bi in the soil following irrigation with groundwater containing unsupported ^{222}Rn .

1.5.3 Air Submodel

The air submodel addresses radionuclide transport from contaminated water and soil to the indoor and outdoor air. As shown in Section 1.5.1.1, the majority of unsupported radon in the irrigation water is not expected to accumulate in the surface soil, but rather is released into the air during irrigation or shortly thereafter. Unsupported radon also outgasses from the water used indoors. The contribution from unsupported radon in groundwater to the concentration of this gas in outdoor and indoor air is discussed below.

1.5.3.1 Transfer of Radon from Groundwater to Outdoor Air

1.5.3.1.1 Release of Radon from Irrigation Water

Transfer of radon to outdoor air would occur primarily during irrigation. Because overhead irrigation is the predominant irrigation application method in Amargosa Valley, most radon in the water would be released quickly during irrigation or shortly thereafter (Section 1.5.1.1). The average length of the irrigation event can be estimated based on the daily irrigation rate and the irrigation intensity. Daily irrigation rate (mm/day) is a measure of the average amount of contaminated groundwater applied per day during the growing season (over all growing seasons for multiple season crops) for a crop type. Irrigation intensity (cm/hr) is a measure of the rate at which contaminated groundwater is applied to a crop type each time that crop is irrigated. The duration of the irrigation event calculated based on the average values of these parameters used in the biosphere model is shown in Table 6 for different crop types. The results indicate that irrigation water is applied for only a fraction of an hour per day. Radon released from the water would quickly disperse in the atmosphere, especially since the irrigated land is a small percentage of the available land in Amargosa Valley. Furthermore, the crops would be irrigated only during the plant growing season, and some of them would not be irrigated daily. Also, during the hot part of the year crops may be irrigated at night to conserve water. Therefore, the overall impact of radon release from the irrigation water would be negligible.

Table 6. Irrigation Parameters for the Crop Types Included in the Biosphere Model

Crop Type	Daily Average Irrigation Rate (mm/day) ^a	Irrigation Intensity (cm/hr) ^a	Irrigation Time (hr/day)
Leafy vegetables	5.41	4.3	0.126
Other vegetables	7.71	4.3	0.179
Fruits	7.41	4.3	0.172
Grains	4.64	4.3	0.108
Forage	6.55	4.3	0.152

^a BSC 2004a, Table 7.1-1.

The quantity of radon released from the water during irrigation can be compared with the quantity of radon released from the surface soil as a result of ²²⁶Ra decay in the soil. For the field crops, which require about twice as much irrigation water as the garden crops, irrigation with the groundwater containing 1 Bq/m³ of ²²⁶Ra results in an average concentration of ²²⁶Ra in

the surface soil of approximately 600 Bq/m² or 1.6 Bq/kg (SNL 2007a, Table 6.10-3). The flux density at the surface of dry soil can be estimated as (UNSCEAR 2000, p. 99)

$$J_D = C_{S_{Ra-226}} \lambda_{Rn-222} f \rho_s (1 - \varepsilon) L \quad (\text{Eq. 10})$$

where

J_D	=	radon flux density at the soil surface (Bq/m ² /s)
$C_{S_{Ra-226}}$	=	activity concentration of ²²⁶ Ra in the soil (Bq/kg)
λ_{Rn-222}	=	decay constant of ²²² Rn (1/s)
f	=	emanation fraction for soil (dimensionless)
ρ_s	=	soil grain density (kg/m ³)
ε	=	porosity of dry soil (dimensionless)
L	=	radon diffusion length, equal to $\sqrt{\frac{D_e}{\lambda_{Rn-222}}}$, where D_e is the effective diffusion coefficient.

Using $C_{S_{Ra-226}} = 1.6$ Bq/kg calculated above; $\lambda_{Rn-222} = 2.1 \times 10^{-6}$ 1/s; and all the remaining values of parameters used in Equation 10 from UNSCEAR (2000, p. 99): $f = 0.2$; $\rho_s = 2,700$ kg/m³; $D_e = 2 \times 10^{-6}$ m²/s, and $\varepsilon = 0.25$; the radon flux density is 1.3×10^{-3} Bq/m²/s. The annual flux density is 4.1×10^4 Bq/m²/yr.

The annual radon flux density from the water used to irrigate field crops can be calculated from the average irrigation rate for these crops. The average annual irrigation rate for the present day climate is 1.78 m/yr (BSC 2004a, Table 7.1-1). For the water containing 1 Bq/m³ of ²²²Rn, and assuming 100% radon release, the ²²²Rn flux density from the water used for irrigation of field crops is 1.78 Bq/m²/yr.

For the assumed net radon enhancement factor of 1,400, the radon flux density would be about 2.5×10^3 Bq/m²/yr, which is still an order of magnitude less than the ²²²Rn flux density from decay of ²²⁶Ra in the soil. Furthermore, the temporal patterns of radon release from the irrigation water and from the surface soil are quite different. Radon release from ²²⁶Ra decay in the soil is continuous. Radon release from the irrigation water would be transient and episodic, occurring mostly during irrigation, when people are unlikely to be present in the irrigated fields.

The size of the irrigated area in Amargosa Valley is small compared to the size of the occupied area. Most people in Amargosa Valley live in the triangular area approximately $17 \times 17 \times 24$ km (150 km²) in size. This area, known as the farming triangle, is also where most agriculture in the valley occurs (BSC 2006, p. 6-24). However, only less than 10% of that area was planted in agriculture (BSC 2006, p. 6-24). Considering the small size of the irrigated fields, compared to the size of the surrounding area, radon released from the irrigation water would quickly disperse due to atmospheric mixing and turbulence.

For the release of ^{222}Rn resulting from decay of ^{226}Ra in the soil, the biosphere model does not consider mixing of the contaminated air above the irrigated fields with the outside air and thus diluting ^{222}Rn concentration in the air. Rather, it treats the contaminated soil that is the source of radon as infinite in extent and depth. This results in overestimating the ^{222}Rn concentration in the air because, as noted above, only a small percentage of land would be irrigated with contaminated water. Because the model treats the entire surface soil at the RMEI location as contaminated, the RMEI receives the dose from inhalation of ^{222}Rn and its decay products at all times while in the area. This results in overestimating inhalation exposure from ^{222}Rn and its decay products and more than compensates for the outgassing of unsupported ^{222}Rn from the irrigation water. Therefore, it is reasonable to ignore the contribution of unsupported ^{222}Rn from the irrigation water to the outdoor air concentration of this radionuclide.

1.5.3.1.2 Release of Radon from Fishpond Water

Fishpond water undergoes aeration, so radon initially dissolved in the water would be very quickly released to the atmosphere. The rate of water addition to the ponds to compensate for the losses caused by evaporation is about 2 m/yr (BSC 2004b, p. 6-104). Assuming a net radon enhancement factor of 1,400, ^{222}Rn flux density is about $2.8 \times 10^3 \text{ Bq/m}^2/\text{yr}$, similar to that for the irrigated soil calculated in Section 1.5.3.1.1 (because the rate of water loss by evaporation is similar to the irrigation rate). However, the total surface area of the fishponds is only about $4,000 \text{ m}^2$ ($4 \times 10^{-3} \text{ km}^2$) (BSC 2004b, p. 6-105), which is negligible compared with the area of irrigated fields. Therefore, this environmental transport pathway can be excluded from the model.

1.5.3.2 Transfer of Radon from Groundwater to Indoor Air

1.5.3.2.1 Household Use of Water

Household water use can result in the transfer of radon from the water to indoor air. Examples of household water use include showering, doing laundry, washing dishes, cooking, and flushing toilets. All of these activities release a fraction of radon dissolved in the water into the indoor air. The quantity of radon that outgasses from the water and enters the indoor air depends on the amount of water used in such activities, activity duration and frequency, as well as the fraction of radon in the water that is liberated. The activities that typically transfer the largest proportion of the dissolved radon to the indoor air are running a washing machine and taking showers (Nazaroff et al. 1987, Table 1).

The radon concentration indoors depends on the quantity of radon released to the indoor air and the indoor air exchange rate. The latter also controls the level of potential alpha energy concentration in the indoor air, and thus the dose from short-lived radon decay products.

The increase of radon concentration in indoor air resulting from the use of water containing dissolved radon can be quantified in terms of the radon transfer coefficient (Nazaroff et al. 1987, p. 282). The transfer coefficient provides the expected increase in the concentration of ^{222}Rn in air resulting from a given concentration of ^{222}Rn in water used at home. The effective radon transfer coefficient (RTC) can be expressed as

$$RTC = \frac{\Delta C a_{Rn-222}}{C w_{Rn-222}} \quad (\text{Eq. 11})$$

where

- RTC = radon transfer coefficient (dimensionless)
- $\Delta C a_{Rn-222}$ = average incremental increase of ^{222}Rn concentration in indoor air resulting from the use of water (Bq/m^3)
- $C w_{Rn-222}$ = average concentration of ^{222}Rn in water (Bq/m^3).

The effective radon transfer coefficient is a composite parameter that accounts for the many individual water use events, the water use rates and frequencies, as well as the indoor air exchange rates. The radon transfer coefficient can be measured or calculated. The measurements of the transfer coefficients are reported in *Risk Assessment of Radon in Drinking Water* (NRC 1999, p. 52).

The transfer coefficient can also be estimated based on the rates of water use, radon transfer efficiency (a fraction of radon released from water to air), and house ventilation (SNL 2007a, Equation 7.4.3-1; Nazaroff et al. 1987, p. 282).

$$RTC = \frac{f_{wa} U_w}{(\lambda_{Rn-222} + \nu)V} \quad (\text{Eq. 12})$$

where

- f_{wa} = transfer efficiency of radon from water to air (dimensionless)
- U_w = household water use rate (L/h)
- λ_{Rn-222} = radioactive decay constant of ^{222}Rn (0.0076/h)
- ν = house ventilation rate (1/h)
- V = volume of the house (L).

The typical home ventilation rate is around 1 air exchange per hour (1/h) (SNL 2007a, Table 6.3-3), which is much greater than the radioactive decay constant for ^{222}Rn (0.0076/h). Thus, radon removal from homes is controlled primarily by the ventilation rate.

If the radon transfer coefficient is known, the increase of radon concentration in indoor air from the indoor water used can be calculated as

$$\Delta C a_{Rn-222} = RTC \times C w_{Rn-222} \quad (\text{Eq. 13})$$

Since the radon transfer coefficient depends on many factors, this RAI response uses the measured value of this parameter. The available experimental data presented in *Risk Assessment of Radon in Drinking Water* (NRC 1999, Figure 3.1) were used to construct a cumulative probability distribution of the parameter values. The mean value for this distribution is 9.2×10^{-5} . The distribution includes data points from experiments, which assumed that all the indoor radon was due to radon in the water (NRC 1999, p. 51). Since a significant fraction of indoor radon has its source in the soil gas entering a house, these results are overestimated. Furthermore, in one of the studies it was found that the transfer coefficients for mobile homes were lower than those for conventional residences (NRC 1999, p. 51). These findings are relevant because a large percentage of the houses in Amargosa Valley are mobile homes (SNL 2007a, Section 6.1.2). For 1 Bq/m^3 of ^{226}Ra in the groundwater and an assumed net radon enhancement factor of 1,400, the increase of indoor air radon concentration from unsupported radon in the water would be about $1.3 \times 10^{-1} \text{ Bq/m}^3$ for all household uses of water that result in ^{222}Rn transfer to the air. For comparison, the supported ^{222}Rn concentration indoors for normal conditions is about $1.1 \times 10^{-1} \text{ Bq/m}^3$ (based on the data in SNL 2007a, Table 6.10-3, and concepts in Section 6.4.2.3), so the contribution from the indoor use of water containing elevated unsupported radon concentration is comparable in magnitude.

The transfer coefficients illustrated in *Risk Assessment of Radon in Drinking Water* (NRC 1999, Figure 3.1) were measured at temperate climate locations, where evaporative coolers are not likely to be used. When evaporative coolers are used, the ventilation rates are much higher than the rates measured elsewhere (NRC 1999, pp. 54 to 55), including the rates used in the biosphere model for the conditions without the coolers (SNL 2007a, Table 6.6-3). Considering the negligible contribution to the radon removal from its radioactive decay, the transfer coefficient for the conditions with operating evaporative coolers can be estimated to be

$$RTC_{evap} = RTC \frac{v_n}{v_e} \quad (\text{Eq. 14})$$

where

- RTC_{evap} = radon transfer coefficient for the conditions when evaporative coolers are in operation (dimensionless)
- v_n = normal house ventilation rate (1/h)
- v_e = house ventilation rate when evaporative coolers are in operation (1/h).

Using the typical values of biosphere model parameters for ventilation rates (1.0 1/hr for the normal ventilation rate and 15.5 1/hr for the ventilation rate with the coolers), and the mean measured value of the radon transfer coefficient, the radon transfer coefficient for homes with an operating evaporative cooler is 5.9×10^{-6} . Using this transfer coefficient in Equation 14, and an assumed net radon enhancement factor of 1,400, indoor water use contributes about $8.3 \times 10^{-3} \text{ Bq/m}^3$ to the indoor radon concentration. For comparison, the supported ^{222}Rn concentration indoors for the conditions with operating evaporative coolers is about $6.1 \times 10^{-2} \text{ Bq/m}^3$ (based on the data in SNL 2007a, Table 6.10-3, and concepts in Section 6.4.2.3), which is almost one order of magnitude higher than that from the unsupported ^{222}Rn .

1.5.3.2.2 Operating Evaporative Coolers

In addition to the contribution to indoor radon from water use, evaporative cooler operation will also result in an incremental addition of radon into the indoor air. An evaporative cooler uses latent heat of water evaporation to cool the outdoor air and pumps it into the home. The water is added to the sump and gets distributed to the evaporation pads by the recirculation pump. The fan pulls outdoor air through the pads to lower its temperature; then the air is pushed through ducts into the home. Because radon would be released from the water inside the evaporative cooler, it can be assumed that 100% of that radon would enter the indoor air. During evaporative cooler operation, water is being added to the cooler to replace the water lost by evaporation.

The contribution of radon from evaporative cooler use can be calculated from the operational characteristics of the cooler (water use rate and air flow rate), the house ventilation rate, and the concentration of radon in the water, including its sorption enhancement factor. This additional radon source would only be applicable for the time when the cooler is in use.

The incremental increase in the concentration of indoor radon during the operation of evaporative coolers can be calculated as (SNL 2007a, Equation 6.4.2-3)

$$Ca_{Rn-222, \text{evap}} = \frac{M_{\text{water}}}{F_{\text{air}}} Cw_{Rn-222} \quad (\text{Eq. 15})$$

where

- $Ca_{Rn-222, \text{evap}}$ = activity concentration of ^{222}Rn in the air resulting from the operation of an evaporative cooler (Bq/m^3)
- M_{water} = water evaporation rate (water use) for an evaporative cooler (m^3/h)
- F_{air} = air flow rate for an evaporative cooler (m^3/h)
- Cw_{Rn-222} = activity concentration of ^{222}Rn in the groundwater (Bq/m^3).

For the typical values of the parameters used in the biosphere model ($M_{\text{water}} = 17 \text{ L/hr}$; $F_{\text{air}} = 8,300 \text{ m}^3/\text{hr}$) (SNL 2007a, Table 6.3-3), $1 \text{ Bq}/\text{m}^3$ of ^{226}Ra in the groundwater, and the assumed net radon enhancement factor of 1,400, operating an evaporative cooler contributes about $2.9 \times 10^{-3} \text{ Bq}/\text{m}^3$ to the indoor radon gas concentration. The indoor concentration from the entry of supported radon in the soil gas is $6.1 \times 10^{-2} \text{ Bq}/\text{m}^3$ when the evaporative coolers are in use, which is more than an order of magnitude higher than that from the unsupported radon from the evaporative cooler water.

1.5.3.3 Resuspension of Soil Contaminated with Radon Decay Products

The incremental increase of the activity concentration of radon decay products in the soil as a result of irrigating with water with enriched unsupported radon is very small compared to the concentration of these decay products from accumulation of ^{226}Ra in the surface soil. The short-lived decay products, other than ^{210}Pb , will quickly decay and will not accumulate in the soil. ^{210}Pb will accumulate in the soil, as discussed in Section 1.5.2.1; however, its

concentrations in the surface soil will be about two orders of magnitude lower than the concentration of ^{210}Pb from the decay of ^{226}Ra , which has accumulated in the surface soil (as shown in Table 4).

In the biosphere model, radionuclide concentrations in the air from soil resuspension are used to estimate the inhalation dose and crop contamination by the deposition of resuspended particles onto plant surfaces. Because the concentrations of radon decay products in the resuspended soil will be dominated by the supported radon decay products from the decay of ^{226}Ra in the soil, and from supported ^{210}Pb , the contribution of unsupported radon and its decay products will be negligible.

1.5.4 Plant, Animal, and Fish Submodels

If contaminated groundwater is used to irrigate crops, they become contaminated by the deposition of radionuclides onto the above-ground plant parts from interception of irrigation water and soil resuspension as well as through root uptake of radionuclides from the soil. The increased concentration of ^{222}Rn in the groundwater will affect all of these environmental transport pathways. The only radionuclide in the radon decay chain that is sufficiently long lived in comparison with the environmental transport processes and, therefore, could accumulate in the soil is ^{210}Pb . The other decay products are too short-lived; however, upon decay they contribute to the ^{210}Pb activity. As discussed in Sections 1.5.1.3 and 1.5.2.1, the contribution of ^{210}Pb from unsupported radon is proportional to the ^{226}Ra concentration in the groundwater and the sorption enhancement factor, and can be effectively evaluated by appropriately modifying the concentration of ^{210}Pb in the water (the source term) in the biosphere model (SNL 2007a, Section 6.3.1.6) by using Equations 9b and 9d in Section 1.5.1.3. As previously noted, for irrigation water containing unsupported radon, it can be assumed that 10% of ^{222}Rn and all short-lived decay products remain in the irrigation water and that there will be 5.1×10^{-5} Bq of ^{210}Pb for every becquerel of ^{222}Rn originally present in the groundwater. For an assumed net radon enhancement factor of 1,400 and a lead sorption enhancement factor of 1.1, the concentrations of ^{210}Pb in crops will increase by a factor of 1.065 relative to the concentrations from supported ^{210}Pb , i.e., ^{210}Pb which was initially in the groundwater.

The animal submodel considers the transfer of radionuclides to animal products from animal feed, water, and soil. Unsupported radon in the groundwater will affect all of these pathways, although, similarly to the plant submodel, the concentrations of ^{210}Pb in animal products will increase only by a factor of 1.065 relative to the concentrations from supported ^{210}Pb , which was initially in the groundwater. (The factor of 1.065, calculated in Section 1.5.1.3, quantifies the increase in the concentration of ^{210}Pb in the water and in the environmental media from unsupported ^{222}Rn , relative to those from ^{210}Pb concentration initially in the groundwater.)

Since radon easily transfers from water to air, it is rarely found in higher concentrations in surface waters, and is primarily of concern for ground waters. The only surface waters considered in the biosphere model are fishponds. Because fishpond water is aerated, radon will be rapidly released from the water. Although its short-lived decay products will quickly decay, ^{210}Pb would remain in the fishpond water, as discussed in Section 1.5.1.3. The resulting activity concentration in fish will increase by a factor of 1.065 relative to the concentrations from

supported ^{210}Pb , which was initially in the groundwater. This value was calculated assuming 10% radon retention in the water, which for fishpond water is an overestimate. Because of water aeration, all radon would actually escape.

1.5.5 External Exposure Submodel

In the biosphere model, external exposure to ^{226}Ra in the soil is caused primarily by the gamma radiation from ^{214}Bi and ^{214}Pb , which are short-lived decay products of ^{222}Rn (SNL 2007a, Table 6.4-4). In addition to ^{214}Bi and ^{214}Pb from the decay of ^{226}Ra , enhanced concentrations of unsupported ^{222}Rn in the irrigation water will cause a short-term increase of ^{214}Bi and ^{214}Pb in the soil. As discussed in Section 1.5.2.2, the average activity concentration of ^{214}Bi and ^{214}Pb from the decay of ^{226}Ra accumulated in the soil is much greater than that from the decay of unsupported ^{222}Rn . Therefore, the short-term incremental increase of the external exposure contribution to the BDCF for ^{226}Ra due to the decay of additional activity of ^{214}Bi and ^{214}Pb in the soil from irrigation water can be neglected.

1.5.6 Inhalation Exposure Submodel

The inhalation exposure model for ^{226}Ra includes two additional pathways of radon transfer from water to indoor air. One results from indoor water use, and the other results from evaporative cooler use. The additional dose from these two pathways can be calculated by using indoor air concentrations of ^{222}Rn calculated in Section 1.5.3. Using these concentrations of ^{222}Rn in the indoor air and the typical values of the other biosphere model parameters, it can be calculated that the inhalation dose from the indoor use of the groundwater containing 1 Bq/m^3 of ^{226}Ra , and $1,400\text{ Bq/m}^3$ of ^{222}Rn (i.e., the net radon enhancement factor is 1,400), is approximately $1.2 \times 10^{-6}\text{ Sv/yr}$, and the inhalation dose from using evaporative coolers is approximately $1.1 \times 10^{-8}\text{ Sv/yr}$. These inhalation pathways can be compared with the inhalation dose from the other inhalation exposure pathways for ^{226}Ra , which is approximately $2.0 \times 10^{-6}\text{ Sv/yr}$, and the previously calculated all-pathway dose of approximately $2.7 \times 10^{-6}\text{ Sv/yr}$ (SNL 2007a, Table 6.10-3). Since these inhalation and all-pathway doses were calculated for a unit ^{226}Ra concentration in the groundwater, they are numerically equal to the inhalation pathway and all-pathway BDCFs, respectively, for ^{226}Ra .

To calculate the dose from inhalation of radon and its short-lived decay products, an equilibrium factor is used in the biosphere model. The equilibrium factor represents the fraction of the potential alpha energy concentration (PAEC) of radon decay products in equilibrium with radon that actually exists in air. For the radon contribution from operating evaporative coolers, the PAEC, and thus the equilibrium factor, would be lower than that used in the TSPA-LA biosphere model because the radon decay products would not have time to build up in the indoor air due to the high air exchange (Hopke et al. 1996, Figures 4.11 and 4.12). Therefore, using the same equilibrium factor for radon from evaporative coolers as for radon from the soil gas and from the indoor water use is conservative.

The inhalation of ^{210}Pb in resuspended soil would be increased in proportion to the increase of this radionuclide concentration in the resuspendable soil layer, relative to the concentration from the supported ^{210}Pb (i.e., by a factor of 1.065).

1.5.7 Ingestion Submodel

Ingestion of water containing enhanced concentrations of unsupported radon will result in an additional ingestion dose pathway. The contribution from this pathway to the dose to the RMEI can be calculated as (SNL 2007a, Equation 6.4.9-2)

$$D_{ing, Rn-222} = DC_{ing, Rn-222} C_{w_{Rn-222, unsupported}} U_w \quad (\text{Eq. 16})$$

where

$$\begin{aligned} DC_{ing, Rn-222} &= \text{ingestion dose coefficient for } ^{222}\text{Rn (Sv/Bq)} \\ C_{w_{Rn-222, unsupported}} &= \text{concentration of unsupported } ^{222}\text{Rn in the groundwater (Bq/m}^3\text{)} \\ U_w &= \text{annual consumption rate of contaminated drinking water by the RMEI (m}^3\text{/yr).} \end{aligned}$$

For the ingestion dose coefficient for ^{222}Rn of 3.5 nSv/Bq (UNSCEAR 2000, p. 108), the water consumption rate of 2 L/day, ^{226}Ra concentration in the groundwater of 1 Bq/m³, and the assumed net radon enhancement factor of 1,400, the resulting annual dose to the RMEI would be about 3.6×10^{-6} Sv/yr. This is comparable with the all-pathway dose for a unit concentration of ^{226}Ra in the groundwater, when the effect of unsupported ^{222}Rn is not included, approximately equal to 2.7×10^{-6} Sv/yr (SNL 2007a, Table 6.10-3).

1.5.8 All-Pathway Dose and Biosphere Dose Conversion Factors

The annual dose to the RMEI from ^{226}Ra and ^{210}Pb in the groundwater is calculated by multiplying the concentrations of these radionuclides in the groundwater by their respective biosphere dose conversion factors.

$$D_{Ra-226 + Pb-210} = C_{w_{Ra-226}} BDCF_{Ra-226} + C_{w_{Pb-210}} BDCF_{Pb-210} \quad (\text{Eq. 17})$$

where

$$\begin{aligned} D_{Ra-226 + Pb-210} &= \text{annual dose to the RMEI from } ^{226}\text{Ra and } ^{210}\text{Pb in the groundwater (Sv/yr)} \\ C_{w_{Ra-226}} &= \text{concentration of } ^{226}\text{Ra in the groundwater (Bq/m}^3\text{)} \\ BDCF_{Ra-226} &= \text{biosphere dose conversion factor for } ^{226}\text{Ra, which includes the contribution from unsupported radon gas (} ^{222}\text{Rn) to indoor air concentrations and to the drinking water pathway (Sv/yr per Bq/m}^3\text{)} \\ C_{w_{Pb-210}} &= \text{concentration of } ^{210}\text{Pb in the groundwater, which includes the contribution from supported } ^{210}\text{Pb (originally in the groundwater), and unsupported } ^{210}\text{Pb, produced from the decay of the short-lived radon decay products (Bq/m}^3\text{).} \end{aligned}$$

$BDCF_{Pb-210}$ = biosphere dose conversion factor for ^{210}Pb (Sv/yr per Bq/m^3).

In this equation, the BDCF for ^{226}Ra includes the contribution from unsupported ^{222}Rn to the indoor air concentration and the contribution of unsupported ^{222}Rn to the drinking water pathway. The BDCF for ^{210}Pb does not include the contribution from unsupported ^{210}Pb , which results from the decay of unsupported ^{222}Rn . The dose effects of the unsupported ^{210}Pb are addressed by modifying the water source concentrations of ^{210}Pb in the biosphere model, i.e., instead of using ^{210}Pb concentration of $1 \text{ Bq}/\text{m}^3$, as it is used for calculation of the BDCF for supported ^{210}Pb , the concentration of this radionuclide in the groundwater of $1 + \frac{1}{SEF_{Pb}} NEF_{Rn} \times \kappa = 1.065$ is used.

As shown in Section 1.5.1.3, the concentration of ^{210}Pb in the groundwater can be related to the concentration of ^{226}Ra in the groundwater as

$$C_{W_{Pb-210}} = C_{W_{Ra-226}} SEF_{Pb} \left(1 + \frac{1}{SEF_{Pb}} NEF_{Rn} \times \kappa \right) \quad (\text{Eq. 18})$$

Equations 17 and 18 can be combined to give

$$D_{Ra-226 + Pb-210} = C_{W_{Ra-226}} BDCF_{Ra-226} + C_{W_{Ra-226}} SEF_{Pb} \left(1 + \frac{1}{SEF_{Pb}} NEF_{Rn} \times \kappa \right) BDCF_{Pb-210} \quad (\text{Eq. 19a})$$

$$D_{Ra-226 + Pb-210} = C_{W_{Ra-226}} [BDCF_{Ra-226} + (SEF_{Pb} + NEF_{Rn} \times \kappa) BDCF_{Pb-210}] \quad (\text{Eq. 19b})$$

The term in the square brackets in Equation 19b thus becomes an effective BDCF for the $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$ radionuclide pair.

1.6 EFFECT OF SORPTION ENHANCEMENT ON DOSE, GROUNDWATER PROTECTION, AND HUMAN INTRUSION

This final section of the response describes the effect of sorption enhancement on individual protection results for both 10,000 years and 1,000,000 years, groundwater protection results for 10,000 years, and human intrusion results.

1.6.1 $^{231}\text{Pa} \rightarrow ^{227}\text{Ac}$

Accounting for sorption of ^{231}Pa increases the mean groundwater concentration of ^{227}Ac by a factor of 6.8 (Section 1.4.1). For the 10,000-year analysis, ^{227}Ac is screened out of calculations of annual dose to the RMEI (SNL 2007b, Table 7-1). The screening analysis is independent of

the calculation of groundwater concentrations in the TSPA model, and thus is not affected by omission of the sorption enhancement factor for ^{231}Pa . For the 1,000,000-year analysis, the maximum total mean annual dose from ^{227}Ac is approximately 0.03 mrem (SAR Figure 2.4-20b). Accounting for sorption enhancement of ^{227}Ac would increase the mean annual dose from ^{227}Ac by a factor of 6.8 to approximately 0.20 mrem.

The contribution of ^{227}Ac to groundwater protection results is not significant and increasing its contribution by a factor of 6.8 would have negligible impact on these results.

The total mean alpha activity concentration of ^{227}Ac at 10,000 years is 1.2×10^{-7} pCi/L based on 1.1×10^{-7} pCi/L from the seismic ground motion modeling case, 1.3×10^{-8} pCi/L from the waste package early failure modeling case, and 1.4×10^{-9} pCi/L from the drip shield early failure modeling case. This ^{227}Ac alpha concentration is the actual ^{227}Ac concentration multiplied by 4 to account for very short-lived alpha emitters that are in equilibrium with ^{227}Ac . These alpha concentrations are insignificant compared to the total mean alpha concentration of approximately 7×10^{-5} pCi/L and the background alpha concentration of 0.5 pCi/L (SAR Figure 2.4-13). An increase to the ^{227}Ac alpha concentration by a factor of 6.8 would be insignificant.

For the organ dose and whole body dose from ^{227}Ac , the actual ^{227}Ac activity concentration is used (i.e., not increased by a factor of 4). Therefore, the mean activity of ^{227}Ac is approximately 3×10^{-8} pCi/L. The organ doses and whole body dose from this concentration of ^{227}Ac are insignificant. The whole body conversion factor for ^{227}Ac is 0.8732 mrem/yr per pCi/L (based on SNL 2007a, Table 6.15-6), which gives a whole body dose of 2.6×10^{-8} mrem/yr from ^{227}Ac compared to 0.06 mrem/yr mean whole body dose from all radionuclides (SAR Figure 2.4-180). The maximum individual organ conversion factor for ^{227}Ac is 25.308 mrem/yr per pCi/L for bone surface (based on SNL 2007a, Table 6.15-6). This ^{227}Ac concentration gives a bone surface dose of 7.5×10^{-7} mrem/yr compared to 0.0123 mrem/yr for bone surface from all radionuclides (SAR Figure 2.4-180), which is insignificant. Increasing the ^{227}Ac concentration by a factor of 6.8 would still result in insignificant whole body and organ doses at 10,000 years.

1.6.2 $^{232}\text{Th} \rightarrow ^{228}\text{Ra}$

Accounting for sorption of ^{232}Th increases the mean groundwater concentration of ^{228}Ra by a factor of 14 (Section 1.4.2). However, neglecting sorption of ^{232}Th has negligible effect on both individual and groundwater protection results.

For the 10,000-year analysis, ^{232}Th , ^{226}Ra , and ^{228}Ra have been screened from calculations of annual dose to the RMEI (SNL 2007b, Table 7-1), and thus individual protection results for the 10,000-year period are not affected by omission of the sorption enhancement factor for ^{228}Ra . For the 1,000,000-year analysis, the maximum total mean annual dose is approximately 2 mrem, to which ^{228}Ra is a negligible contributor. The mean annual dose from ^{228}Ra is 2.9×10^{-5} mrem in the seismic ground motion modeling case, is 2.8×10^{-5} mrem in the igneous intrusion modeling case, is 4.9×10^{-7} mrem in the seismic fault displacement modeling case, is 2.6×10^{-8} mrem in the waste package early failure modeling case, and is 2.8×10^{-10} mrem in the drip shield early failure modeling case. Increasing these doses by a factor of 14 to account for sorption of ^{232}Th would not affect the contribution of ^{228}Ra to total mean annual dose.

Likewise, the contribution of ^{228}Ra to the quantities compared to the groundwater protection standards is always negligible, even after accounting for sorption of ^{232}Th . The three modeling cases which comprise the groundwater protection calculations are the seismic ground motion, waste package early failure, and drip shield early failure modeling cases. For the seismic ground motion modeling case, the groundwater concentrations of ^{226}Ra and ^{228}Ra are 1.2×10^{-7} pCi/L and 2.9×10^{-15} pCi/L, respectively. For the waste package early failure modeling case, the groundwater concentrations of ^{226}Ra and ^{228}Ra are 8.8×10^{-9} pCi/L and 2.8×10^{-15} pCi/L, respectively, and are 1.2×10^{-10} pCi/L and 1.0×10^{-13} pCi/L, respectively, for the drip shield early failure modeling case. The mean total radium concentration is the sum of all these concentrations, which is 1.3×10^{-7} pCi/L, essentially all of which is ^{226}Ra . Thus, increasing the ^{228}Ra concentration by a factor of 14 would have a negligible effect on the total radium concentration in groundwater.

Increasing the ^{228}Ra concentration would also not affect organ or whole body doses from combined beta and photon emitting radionuclides. Neither ^{226}Ra nor ^{228}Ra is a major contributor to organ or whole body doses (SAR Section 2.4.4.1.1.4). The primary contributors are ^{99}Tc and ^{129}I . Although ^{226}Ra is not a beta-photon emitter, it has an organ dose conversion factor that accounts for its decay products, including ^{210}Pb . The conversion factors for ^{226}Ra ($^{226}\text{Ra}+^{210}\text{Pb}$) and ^{228}Ra are comparable, with the ^{228}Ra factor being at most 2.4 times greater (some are smaller) (based on SNL 2007a, Table 6.15-5). Therefore, an increase in the ^{228}Ra concentration by a factor of 14 would have a negligible effect on organ or whole body dose because it would remain small compared to the organ or whole body dose from ^{226}Ra , which is in turn negligible compared to those from ^{99}Tc and ^{129}I .

1.6.3 $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$ and $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$

The $^{226}\text{Ra} \rightarrow ^{222}\text{Rn}$ and $^{226}\text{Ra} \rightarrow ^{210}\text{Pb}$ chains are discussed together because their combined doses are computed in the TSPA model through a single total BDCF.

Accounting for sorption enhancement in these chains increases the mean groundwater concentration of ^{222}Rn by a factor of about 1,400 and increases the mean groundwater concentration of ^{210}Pb by a factor of 1.1 (Sections 1.4.3 and 1.4.4). Radium and lead are screened out from calculations of annual dose to the RMEI for 10,000 years (SNL 2007b, Table 7-1), and thus individual protection results are not affected by omission of the enhancement factors for ^{222}Rn and ^{210}Pb for 10,000 years.

Radon is specifically excluded from the groundwater protection standard for gross alpha activity concentration for 10,000 years. The short-lived decay products of ^{222}Rn should not be included either, for consistency with the methods used to demonstrate compliance with the maximum contaminant levels of the National Primary Drinking Water Regulations at 40 CFR 141.66, which were the source of the groundwater protection standards in 10 CFR 63.331. If the long-lived decay products of radon, such as ^{210}Pb and its decay product ^{210}Po (which is an alpha emitter), are included, they would make little contribution to the estimates of gross alpha activity concentrations (based on activity comparison in Table 3). ^{210}Pb and its decay products are minor contributors to gross alpha activity concentrations and to whole body and organ doses. Therefore, increasing the groundwater concentrations of ^{222}Rn by factor of 1,400 and of ^{210}Pb by

a factor of 1.1 to account for sorption enhancement would have a negligible effect on the estimates, which are compared to the groundwater protection standards.

For the 1,000,000-year analysis, ^{226}Ra is the third highest contributor to total mean annual dose (SAR Figure 2.4-20b). The original mean BDCFs for ^{226}Ra and ^{210}Pb are 3.8×10^{-6} Sv/yr per Bq/m^3 and 2.7×10^{-6} Sv/yr per Bq/m^3 for a combined mean BDCF of 6.5×10^{-6} Sv/yr per Bq/m^3 , neglecting the effect of sorption of ^{226}Ra . The mean value for the net radon enhancement factor is 1,404. The updated mean ^{226}Ra BDCF accounting for enhanced ^{222}Rn groundwater concentrations is 8.7×10^{-6} Sv/yr per Bq/m^3 . Equation 19b is used to calculate the updated combined mean BDCF for ^{226}Ra (which includes the dose effect of ^{210}Pb and all intermediate decay products):

$$\begin{aligned} BDCF_{Ra-226} &= \left[8.7 \times 10^{-6} + (1.1 + 1404 \times 5.1 \times 10^{-5}) \times 2.7 \times 10^{-6} \right] \\ &= 8.7 \times 10^{-6} + 3.2 \times 10^{-6} = 1.2 \times 10^{-5} \frac{\text{Sv/yr}}{\text{Bq/m}^3} \end{aligned}$$

which is 1.8 times higher than the original combined mean BDCF.

The contribution to total mean annual dose from ^{226}Ra is 0.28 mrem (SAR Figure 2.4-20). Using the updated combined mean BDCF for ^{226}Ra developed for this impact assessment would increase the contribution to total mean annual dose from ^{226}Ra to 0.5 mrem.

1.6.4 Net Effect on Results for Groundwater Protection

As indicated in the previous sections, inclusion of the sorption enhancement factor for ^{227}Ac , ^{228}Ra , and ^{210}Pb in this impact assessment does not affect the groundwater protection results. These radionuclides make negligible contributions to the gross alpha activity concentration. ^{228}Ra is negligible compared to ^{226}Ra in the radium activity concentration. Finally, the organ and whole body doses are not changed.

1.6.5 Net Effect on Results for Individual Protection Standard

For the 1,000,000-year analysis, the maximum total mean annual dose is approximately 2 mrem (SAR Figure 2.4-20). Accounting for sorption of parent radionuclides in the calculation of groundwater concentrations of decay products would increase the contribution to total mean annual dose from ^{227}Ac to 0.20 mrem (from 0.03 mrem) and from ^{226}Ra to 0.5 mrem (from 0.28 mrem). The contribution to total mean annual dose from ^{228}Ra would remain negligible. If sorption of parent radionuclides was accounted for, the maximum total mean annual dose would increase from 2.0 mrem to 2.4 mrem, a minor amount of increase when compared to the individual protection standard of 100 mrem for the period beyond 10,000 years postclosure to 1,000,000 years postclosure. Thus, the assumption of secular equilibrium in a rock-water system as implemented in the TSPA model has only a minor effect on the performance assessment results.

1.6.6 Effect on Results for Human Intrusion

The mean annual dose at 1,000,000 years for the human intrusion modeling case is approximately 1×10^{-4} mrem (SAR Figure 2.4-159). The maximum mean annual dose is approximately 1×10^{-2} mrem at the time of the intrusion at 200,000 years, and is mainly from ^{99}Tc and ^{129}I . These radionuclides are not affected by sorption enhancement so the maximum dose is not changed in this impact assessment. The contributions to the mean annual dose at 1,000,000 years are 8.4×10^{-13} mrem, 4.5×10^{-9} mrem, and 4.6×10^{-7} mrem from ^{228}Ra , ^{227}Ac , and ^{226}Ra , respectively. Accounting for the sorption enhancement factors would change these contributions to 5.7×10^{-12} mrem, 6.3×10^{-8} mrem, and 8.3×10^{-7} mrem, respectively. The contribution of these radionuclides to the mean annual dose from human intrusion would still be minor and would not affect the representation of the results shown in SAR Figure 2.4-159.

2. COMMITMENTS TO NRC

None.

3. DESCRIPTION OF PROPOSED LA CHANGE

None.

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