
Part 2: DEVELOPMENT OF PATHWAY-SPECIFIC

SOIL SCREENING LEVELS

This part of the Technical Background Document for Radionuclides describes the methods used to calculate SSLs for residential exposure pathways, along with their technical basis and limitations associated with their use. Simple, standardized equations have been developed for the more common radiological exposure pathways at Superfund sites:

- Direct ingestion of soil (Section 2.2)
- Inhalation of fugitive dust (Section 2.3)
- External radiation exposure from photon-emitting radionuclides in soil (Section 2.4)
- Ingestion of homegrown produce that has been contaminated via plant uptake (Section 2.5)
- Ingestion of contaminated ground water caused by migration of radionuclides through soil to an underlying potable aquifer (Section 2.6).

The equations were developed under the following constraints:

- They should be consistent with current Superfund risk assessment methodologies and guidance.
- To be appropriate for early-stage application, they should be simple and easy to apply.
- They should allow the use of site-specific data where they are readily available or can be easily obtained.
- The process of developing and applying SSLs should generate information that can be used and built upon as a site evaluation progresses.

The equations include easily obtained site-specific input parameters. Conservative default values have been developed for use where site-specific data are not available. Generic SSLs, calculated for 60 radionuclides using these default values, are presented in Appendix A. The generic SSLs are conservative, since the default values are designed to be protective at most sites across the country.

The migration to ground water pathway equation assumes an infinite source (e.g., source remains constant over time). As pointed out by several commenters to the December 1994 draft Soil Screening Guidance for chemicals (U.S. EPA, 1994h), SSLs developed using these models may violate mass-balance for certain contaminants and site conditions (e.g., small sources). To address this concern, EPA has incorporated a simple mass-limit model for this pathway assuming that the entire volume of contamination leaches over the duration of exposure and that the level of contaminant at the receptor does not exceed the health-based limit (Section 2.7). **Because they require a site-specific estimate of source depth, these alternative models cannot be used to calculate generic SSLs.**

The mode of exposure to radionuclides is different than that of chemicals. This renders some chemical pathways inapplicable to radionuclides (e.g., inhalation of volatiles, dermal absorption) while adding other pathways unique to radiation (e.g., external exposure to photons emitted by radionuclides). The radiological pathways listed above represent the most likely exposure mechanisms for individuals in a residential setting. For soils under the residential land use assumption, the external exposure pathway will typically be the dominant exposure pathway for most radionuclides (e.g., ^{60}Co , ^{137}Cs , ^{226}Ra , ^{238}U). For a good number of radionuclides (e.g., ^{14}C , ^{63}Ni , ^{90}Sr , ^{99}Tc), the plant ingestion pathway often dominates, although not to the extent that the external exposure pathway does. The soil ingestion pathway also plays a dominant role for some radionuclides of interest (i.e., ^{129}I , ^{232}Th , ^{239}Pu , ^{241}Am). In the majority of cases, the inhalation of fugitive dust pathway and the migration to groundwater pathway play an insignificant role.

In addition to the more common pathways of exposure in a residential setting, concerns have been raised regarding the potential for migration of radon from subsurface soils into basements. The dominant factor in indoor radon levels is home construction practices and the extent to which these practices employ radon-resistant techniques. Homes built atop soil with identical levels of radium can have orders of magnitude differences in indoor radon levels depending on the extent to which radon-resistant techniques are used. As a naturally-occurring radionuclide, radium is present in all soils. Reducing the radium content in the soil may not result in any reduction in indoor radon levels. However, taking simple and inexpensive steps in home construction will ensure that radon levels in homes are kept below Applicable or Relevant and Appropriate Requirements (ARARs) levels. For existing homes with elevated levels of radon, a variety of methods can be used to reduce radon concentrations to ARAR levels. Discussion of radon mitigation standards may be found in several EPA publications, including *Radon Mitigation Standards*, EPA 402-R-93-078 (U.S. EPA, 1994i). Also note that potential ARARs exist for indoor radon under 192.12(b)(1) and 192.41(b). For further guidance on using these ARARs, see the August 1997 memorandum from Stephen Luftig (OERR) and Larry Weinstock (ORIA) titled "Establishment of cleanup levels for CERCLA sites with radioactive contamination," OSWER Directive 9200.4-18, (U.S. EPA, 1997a).

Section 2.1 describes the human health basis of the Soil Screening Guidance for Radionuclides and provides the human toxicity and health benchmarks necessary to calculate SSLs. The selection and development of the radiological properties required to calculate SSLs are described in Part 5 of this document.

2.1 Human Health Basis

The 60 radionuclides for which generic SSLs have been calculated are listed in Table 2.1. Principal radionuclides are radionuclides with half-lives greater than six months. The decay products of any principal radionuclide down to, but not including, the next principal radionuclide in its decay chain are called associated radionuclides and consist of radionuclides with half-lives less than six months. It is assumed that a principal radionuclide is in secular equilibrium with its associated radionuclides at the point of exposure. This assumption is reasonable because it usually takes about three years or longer to clean up a site. Associated decay chains are indicated, as well as principal radionuclide half-life and the terminal nuclide or radionuclide (i.e., the principal radionuclide or stable nuclide that terminates an associated decay chain).

Tables 2.2 and 2.3 list the regulatory and human health benchmarks necessary to calculate SSLs for 60 radionuclides. For all pathways other than migration to ground water, these benchmarks are based on the latest available radionuclide cancer slope factors. For migration to ground water, drinking water MCLs are shown.

The human health benchmarks in Table 2.2 were obtained from *Health Risks from Low-Level Environmental Exposure to Radionuclides, Federal Guidance Report No. 13, Part I - Interim Version*, EPA 402-R-97-014 (U.S. EPA 1998a) (also known as FGR 13). When the slope factors are not available in FGR 13, data are taken from *Health Effects Assessment Summary Tables (HEAST): Annual Update, FY-1995*, (U.S. EPA, 1995a) (also known as the

HEAST report). Slope factors are listed for each route of intake for principal radionuclides in units of picocuries (pCi).¹ Radionuclides are presented alphabetically by element and atomic weight.

MCLs in Table 2.3 were obtained from *Drinking Water Regulations and Health Advisories* (U.S. EPA, 1995a). The current MCLs for radionuclides were enacted in 1976 and are set at 4 mrem/yr for the sum of the doses from beta particles and photon emitters, 15 pCi/L for gross alpha particle activity (including Ra-226, but excluding uranium and radon), and 5 pCi/L combined for Ra-226 and Ra-228. In July 1991, EPA proposed to revise the MCLs for Ra-226 and Ra-228 to 20 pCi/L for each, change the methodology used for determining a 4 mrem/yr dose for the sum of the doses from beta particles and photon emitters, alter the definition of alpha particle activity to exclude Ra-226, and establishing new MCLs of 300 pCi/L for Rn-222 and 20 µg/L (30 pCi/L) for uranium (56 FR 33050). EPA is under Court Order to either finalize the 1991 proposal for radionuclides (except for radon), or to ratify existing standards by November 2000. On April 21, 2000 EPA solicited comment in a Notice of Data Availability (NODA) on three options for a uranium MCL: 1) 20 ug/l **and** 20 pCi/l as a preferred option, 2) 40 ug/l **and** 40 pCi/l, and 3) 80 ug/l **and** 80 pCi/l (65 FR 21576). In this NODA, EPA indicated that changes would not be made to the existing MCLs for radium, alpha particle activity, and beta particles and photon emitters. The 1996 Amendments to the Safe Drinking Water Act (SDWA) require EPA to propose a MCL for radon by August 1999, and to finalize the MCL by August 2000. To comply with the requirements of the amended SDWA, on August 6, 1997, EPA withdrew its 1991 proposal for Rn-222 (62 FR 42221). EPA issued a new proposal for Rn-222 on November 2, 1999 (64 FR 59246). EPA proposed an MCL of 300 pCi/l with an alternative MCL of 4,000 pCi/l if a state or local indoor radon mitigation program was established.

References for each table are updated regularly. **Prior to calculating SSLs, the values in Tables 2.2 and 2.3 should be checked against the most recent version of these sources to ensure that they are up-to-date.**

Selected radionuclides and radioactive decay chain products are designated with the suffix "+D" (e.g., U-238+D, Ra-226+D, Cs-137+D) to indicate that cancer risk estimates for these radionuclides include the contributions from their short-lived decay products, assuming equal activity concentrations (i.e., secular equilibrium) with the principal or parent nuclide in the environment.² In most cases, site-specific analytical data should be used to establish the actual degree of equilibrium between each parent radionuclide and its decay products in each media sampled. However, in the absence of empirical data, the "+D" values for radionuclides should be used unless there are compelling reasons not to. Note that there may be circumstances, such as long disposal times or technologically enhanced concentrations of naturally occurring radionuclides, that may necessitate the combination of the risks of a parent radionuclide and its decay products over several contiguous subchains. For example, Ra-226 soil analyses at a site might show that all radium decay products are present in secular equilibrium down to stable Pb-206. In this case, Ra-226 risk calculations should be based on the ingestion, inhalation and external exposure slope factors for the Ra-226+D subchain, plus the ingestion, inhalation and external exposure factors for the Pb-210+D subchain. For actual sites, users should consult with a health physicist or radiochemist (1) to evaluate the site-specific analytical data to determine the degree of equilibrium between parent radionuclides and decay members of contiguous decay chains and (2) to assist in the combination of appropriate slope factor values.

¹ Slope factors are reported in the customary units of risk per picocurie (1 pCi = 10⁻¹² curies (Ci) = 3.7x10⁻² nuclear transformations per second) for consistency with the system used for radionuclides in the IRIS database. If required, slope factors can be converted into the International System (SI) units of becquerels (1 Bq = 1 nuclear transformation per second) by multiplying each inhalation, ingestion, or external exposure value by 27.03. Users can calculate cancer risks using slope factors expressed in either customary units or SI units with equivalent results, provided that they also use air, water and soil concentration values in the same system of units.

² There is one exception to the assumption of secular equilibrium. For the inhalation slope factor for Rn-222+D reported in the table, ORIA assumes a 50% equilibrium value for radon decay products (Po-218, Pb-214, Bi-214 and Po-214) in air.

Table 2.1. Radionuclides Included in Generic Soil Screening Analysis

Principal Radionuclide ^a		Associated Decay Chain ^b	Terminal Nuclide or Radionuclide ^c	
Nuclide	Half-life (yr)		Nuclide	Half-life (yr)
Ac-227+D	22	[Th-227 (98.6%, 19 d) Fr-223 (1.4%, 22 min)] Ra-223 (11 d) Rn-219 (4 s) Po-215 (2 ms) Pb-211 (36 min) Bi-211 (2 min) [Tl-207 (99.7%, 5 min) Po-211 (0.3%, 0.5 s)]	Pb-207	stable
Ag-108m	127	-	Pd-108 (91%) [Cd-108 (98%) Ag-108 (9%) Pd-108 (2%)]	stable stable 2 min stable
Ag-110m	0.7	-	Cs-110 (99%) [Cd-110 (99.7%) Ag-110 (1%) Pd-110 (0.3%)]	stable stable 25 s stable
Am-241	432	-	Np-237	2.1E+6
Am-243+D	7,400	Np-239 (2 d)	Pu-239	2.4E+4
Bi-207	38	-	Pb-207	stable
C-14	5,730	-	N-14	stable
Cd-109	1.3	-	Ag-109	stable
Ce-144+D	0.8	[Pr-244 (9%, 17 min) Pr-244m (2%, 7 min)]	Nd-144	stable
Cl-36	3.0E+5	-	S-36	stable
Cm-243	28	-	Am-243 (0.2%) ^e	7,400
Cm-244	18	-	Pu-240	6,600
Co-57	0.7	-	Fe-57	stable
Co-60	5	-	Ni-60	stable
Cs-134	2	-	Ba-134 (~100%)	stable
Cs-135	3E+6	-	Ba-135	stable
Cs-137+D	30	Ba-137m (95%, 3 min)	Ba-137	stable
Eu-152	13	-	Sm-152 (72%) Gd-152 (28%)	stable 1.1E+14
Eu-154	8	-	Gd-154 (~100%)	stable
Eu-155	5	-	Gd-155	stable
Fe-55	3	-	Mn-55	stable
Gd-153	0.7	-	Eu-153	stable
H-3	12	-	He-3	stable
I-129	1.6E+7	-	Xe-129	stable
K-40	1.3E+9	-	Ca-40 (89%) Ar-40 (11%)	stable
Mn-54	0.9	-	Cr-54	stable

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Principal Radionuclide ^a		Associated Decay Chain ^b	Terminal Nuclide or Radionuclide ^c	
Nuclide	Half-life (yr)		Nuclide	Half-life (yr)
Na-22	3	-	Ne-22	stable
Nb-94	2.0E+4	-	Mo-94	stable
Ni-59	7.5E+4	-	Co-59	stable
Ni-63	100	-	Cu-53	stable
Np-237+D	2.1E+6	Pa-233 (27 d)	U-233	1.6E+5
Pa-231	3.3E+4	-	Ac-227	22
Pb-210+D	22	Bi-210 (5 d) Po-210 (138 y)	Pb-206	stable
Pm-147	3	-	Sm-147	1.1E+11
Pu-238	88	-	U-234	2.4E+5
Pu-239	2.4E+4	-	U-235	7E+8
Pu-240	6,500	-	U-236	2.3E+6
Pu-241	14	-	Am-241	432 y
Pu-242	3.8E+5	-	U-238	4.5E+9
Pu-244+D	9.3E+7	U-240 ~100%, 14) Np-240	Pu-240	6,500
Ra-226+D	1,600	Rn-222 (4 d) Po-218 (3 min) Pb-214 (~100%, 27 min) Bi-214 (20 min) Po-214 (~100%, 1 min)	Pb-210	22
Ra-228+D	8	Ac-228 (6 h)	Th-228	2
Ru-106+D	1	Rh-106 (30 s)	Pd-106	stable
Sb-125+D	3	Te-125m (23%, 58 d)	Te-125	stable
Sm-147	1.1E+11	-	Nd-143	stable
Sm-151	90	-	Eu-151	stable
Sr-90+D	29	Y-90 (64 h)	Zr-90	stable
Tc-99	2.1E+5	-	Ru-99	stable
Th-228+D	2	Ra-224 (4 d) Rn-220 (56 s) Po-216 (0.2 s) Pb-212 (11h) Bi-212 (61 min) [Po-212 (64%, 0.3 μs) Tl-208 (36%, 3 min)]	Pb-208	stable
Th-229+D	7,300	Ra-225 (15 d) Ac-225 (10 d) Fr-221 (5 min) At-217 (32 ms) Bi-213 (46 min) [Po-213 (98%, 4 μs) Tl-209 (2%, 2 min)] Pd-209 (3 h)	Bi-209	stable
Th-230	7.7E+4	-	Ra-226	1,600
Th-232	1.4E+10	-	Ra-228	6

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Principal Radionuclide ^a		Associated Decay Chain ^b	Terminal Nuclide or Radionuclide ^c	
Nuclide	Half-life (yr)		Nuclide	Half-life (yr)
Tl-204	4	-	Pb-204 (97%) Hg-204 (3%)	stable stable
U-232	72	-	Th-228	2
U-233	1.6E+5	-	Th-229	7,300
U-234	2.4E+5	-	Th-230	8E+4
U-235+D	7.0E+8	Th-231 (26 h)	Pa-231	3.4E+4
U-236	2.3E+6	-	Th-232	1.4E+10
U-238+D	4.5E+9	Th-234 (24 d) [Pa-234m (99.8%, 1 min) Pa-234 (0.2%, 7 h)]	U-234	2.4E+5
Zn-65	0.7	-	Cu-65	stable

^a Radionuclides with half-lives greater than six months. "+D" designates principal radionuclides with associated decay chains.

^b The chain of decay products of a principal radionuclide extending to (but not including) the next principal radionuclide or a stable nuclide. Half-lives are given in parentheses. Branches are indicated by square brackets with branching ratios in parentheses.

^c The principal radionuclide or stable nuclide that terminates an associated decay chain.

^d A hyphen indicates that there are no associated decay products.

^e The branching decay for Pu-241 and Cm-243 involves multiple principal radionuclides and associated radionuclides.

Table 2.2. Radionuclide Cancer Morbidity - Slope Factors (1)

Radionuclide	Slope Factor (Morbidity Risk Coefficient) Lifetime Excess Cancer Risk per Unit Exposure					Notes
	Water Ingestion (risk/pCi)	Food Ingestion (risk/pCi)	Soil Ingestion (risk/pCi)	Inhalation (risk/pCi)	External Exposure (risk/yr per PCi/g soil)	
Ac-227+D	4.86E-10	6.53E-10	1.16E-09	2.09E-07	1.47E-06	2
Ag-108m	8.14E-12	1.12E-11	1.92E-11	2.67E-11	7.19E-06	2
Ag-110m	9.88E-12	1.37E-11	2.37E-11	2.83E-11	1.30E-05	2
Am-241	1.04E-10	1.34E-10	2.17E-10	2.81E-08	2.76E-08	
Am-243+D	1.08E-10	1.42E-10	2.32E-10	2.70E-08	6.36E-07	2
Bi-207	5.66E-12	8.14E-12	1.49E-11	2.10E-11	7.08E-06	
C-14	1.55E-12	2.00E-12	2.79E-12	7.07E-12	7.83E-12	3
Cd-109	5.00E-12	6.70E-12	1.14E-11	2.19E-11	8.73E-09	
Ce-144+D	3.53E-11	5.19E-11	1.02E-10	1.10E-10	2.44E-07	2
Cl-36	3.30E-12	4.44E-12	7.66E-12	2.50E-11	1.74E-09	
Cm-243	9.47E-11	1.23E-10	2.05E-10	2.69E-08	4.19E-07	
Cm-244	8.36E-11	1.08E-10	1.81E-10	2.53E-08	4.85E-11	
Co-57	1.04E-12	1.49E-12	2.78E-12	2.09E-12	3.55E-07	
Co-60	1.57E-11	2.23E-11	4.03E-11	3.58E-11	1.24E-05	
Cs-134	4.22E-11	5.14E-11	5.81E-11	1.65E-11	7.10E-06	
Cs-135	4.74E-12	5.88E-12	7.18E-12	1.86E-12	2.36E-11	
Cs-137+D	3.04E-11	3.74E-11	4.33E-11	1.19E-11	2.55E-06	2
Eu-152	6.07E-12	8.70E-12	1.62E-11	9.10E-11	5.30E-06	
Eu-154	1.03E-11	1.49E-11	2.85E-11	1.15E-10	5.83E-06	
Eu-155	1.90E-12	2.77E-12	5.40E-12	1.48E-11	1.24E-07	
Fe-55	8.62E-13	1.16E-12	2.09E-12	7.99E-13	0	
Gd-153	1.52E-12	2.22E-12	4.26E-12	6.55E-12	1.62E-07	
H-3	5.07E-14	6.51E-14	9.25E-14	5.62E-14	0	4
I-129	1.48E-10	3.22E-10	2.71E-10	6.07E-11	6.10E-09	5
K-40	2.47E-11	3.43E-11	6.18E-11	1.03E-11	7.97E-07	
Mn-54	2.28E-12	3.11E-12	5.14E-12	5.88E-12	3.89E-06	
Na-22	9.62E-12	1.26E-11	1.97E-11	3.89E-12	1.03E-05	
Nb-94	7.77E-12	1.11E-11	2.05E-11	3.77E-11	7.29E-06	
Ni-59	2.74E-13	3.89E-13	7.33E-13	4.66E-13	0	
Ni-63	6.70E-13	9.51E-13	1.79E-12	1.64E-12	0	
Np-237+D	6.74E-11	9.10E-11	1.62E-10	1.77E-08	7.97E-07	2
Pa-231	1.73E-10	2.26E-10	3.74E-10	4.55E-08	1.39E-07	
Pb-210+D	1.27E-09	3.44E-09	2.66E-09	1.39E-08	4.21E-09	2
Pm-147	1.69E-12	2.48E-12	4.88E-12	1.61E-11	3.21E-11	
Pu-238	1.31E-10	1.69E-10	2.72E-10	3.36E-08	7.22E-11	
Pu-239	1.35E-10	1.74E-10	2.76E-10	3.33E-08	2.00E-10	
Pu-240	1.35E-10	1.74E-10	2.77E-10	3.33E-08	6.98E-11	
Pu-241	1.76E-12	2.28E-12	3.29E-12	3.34E-10	4.11E-12	
Pu-242	1.28E-10	1.65E-10	2.63E-10	3.13E-08	6.25E-11	
Pu-244+D	1.44E-10	1.90E-10	3.14E-10	2.93E-08	1.51E-06	2
Ra-226+D	3.86E-10	5.15E-10	7.30E-10	1.16E-08	8.49E-06	2
Ra-228+D	1.04E-09	1.43E-09	2.29E-09	5.23E-09	4.53E-06	2
Ru-106+D	4.22E-11	6.11E-11	1.19E-10	1.02E-10	9.66E-07	2
Sb-125+D	5.13E-12	7.21E-12	1.32E-11	1.93E-11	1.81E-06	2
Sm-147	3.74E-11	4.77E-11	7.59E-11	6.88E-09	0	
Sm-151	5.55E-13	8.07E-13	1.59E-12	4.88E-12	3.60E-13	
Sr-90+D	7.40E-11	9.53E-11	1.44E-10	1.13E-10	1.96E-08	2
Tc-99	2.75E-12	4.00E-12	7.66E-12	1.41E-11	8.14E-11	
Th-228+D	3.00E-10	4.22E-10	8.09E-10	1.43E-07	7.76E-06	2
Th-229+D	5.28E-10	7.16E-10	1.29E-09	2.25E-07	1.17E-06	2
Th-230	9.10E-11	1.19E-10	2.02E-10	2.85E-08	8.19E-10	
Th-232	1.01E-10	1.33E-10	2.31E-10	4.33E-08	3.42E-10	
Tl-204	5.85E-12	8.25E-12	1.54E-11	2.45E-12	2.76E-09	
U-232	2.92E-10	3.85E-10	5.74E-10	1.95E-08	5.98E-10	
U-233	7.18E-11	9.69E-11	1.60E-10	1.16E-08	9.82E-10	
U-234	7.07E-11	9.55E-11	1.58E-10	1.14E-08	2.52E-10	
U-235+D	7.18E-11	9.76E-11	1.63E-11	1.01E-08	5.43E-07	2
U-236	6.70E-11	9.03E-11	1.49E-10	1.05E-08	1.25E-10	
U-238+D	8.71E-11	1.21E-10	2.10E-10	9.35E-09	1.14E-07	2

Radionuclide	Water Ingestion (risk/pCi)	Food Ingestion (risk/pCi)	Soil Ingestion (risk/pCi)	Inhalation (risk/pCi)	External Exposure (risk/yr per PCi/g soil)	Notes
Zn-65	1.17E-11	1.54E-11	2.45E-11	5.81E-12	2.81E-06	

Notes:

1. A curie (Ci), the customary unit of activity, is equal to 3.7×10^{10} nuclear transformations per second. 1 picocurie (pCi) = 10^{-12} Ci. If required, slope factors in Table D.1 can be converted into the International System (SI) units of becquerels (1 Bq = 1 nuclear transformation per second) by multiplying each inhalation, ingestion, or external exposure value by 27.03. Users can calculate cancer risks using slope factors expressed in either customary units or SI units with equivalent results, provided that they also use air, water, food and soil concentration values in the same system of units.
2. For each radionuclide listed, slope factors correspond to the risks per unit intake or exposure for that radionuclide only, except when marked with a "+D". In these cases, the risks from associated short-lived radioactive decay products (i.e., those decay products with radioactive half-lives less than or equal to 6 months) are also included, based on an assumption of secular equilibrium. These decay chains are identified in Table C.1 of Attachment C.
3. The inhalation slope factor listed represents inhalation of C-14 as a particulate. Alternative values for inhalation of C-14 as a gas are $3.36E-15$ risk/pCi for carbon monoxide and $1.99E-14$ risk/pCi for carbon dioxide.
4. The inhalation slope factor for H-3 represents inhalation of tritiated water vapor, which is considered the most likely form in the environment. Alternative values of inhalation of H-3 include $1.99E-13$ risk/pCi for particulates, $5.62E-18$ risk/pCi for elemental hydrogen gas, and $1.28E-13$ risk/pCi for organic forms. Similarly, the ingestion slope factor values for H-3 represent ingestion of tritiated water, which is considered the most likely form in the environment. Alternative values for ingestion of organically bound forms of H-3 in water, food, and soil are $1.12E-13$ risk/pCi, $1.44E-13$ risk/pCi, and $2.02E-13$ risk/pCi, respectively.
5. The food ingestion slope factor for I-129 represents ingestion of milk. For ingestion of non-dairy foodstuffs, a lower value of $1.93E-10$ risk/pCi ingested would apply. The inhalation slope factor for I-129 represents inhalation of particulates; alternative values for inhalation of I-129 vapor are $1.24E-10$ for inhalation of methyl iodide and $1.60E-10$ for inhalation of other compounds in vapor form.

Table 2.3 Radionuclide Drinking Water MCLs

Radionuclide	Current MCL ^{a, b} (pCi/L)	Proposed MCL (pCi/L)	Risk Base Limit (RBL) ^e (pCi/L)	Mass Equiv to MCL, Proposed MCL, or RBL (mg/L)
Ac-227			0.24	3.3E-12
Ag-108m			5.8	2.2E-10
Ag-110m	90			1.9E-11
Am-241	15			4.4E-09
Am-243	15			7.5E-08
Bi-207	200			4.4E-09
C-14	2,000			4.5E-07
Cd-109	600			2.3E-10
Ce-144	30			9.1E-12
Cl-36	700			2.1E-05
Cm-243	15			2.9E-10
Cm-244	15			1.9E-10
Cm-248	15			3.5E-06
Co-57	1,000			1.2E-10
Co-60	100			8.9E-11
Cs-134	80			6.2E-11
Cs-135	900			7.8E-04
Cs-137	200			2.3E-09
Eu-152	200			1.1E-09
Eu-154	60			2.3E-10
Eu-155	600			1.3E-09
Fe-55	2,000			8.3E-10
Gd-153	600			1.7E-10
H-3	20,000			2.1E-09
I-129	1			5.7E-06
K-40			1.9	2.7E-4
Mn-54	300			3.9E-11
Na-22	400			6.4E-11
Nb-94			6.1	3.3E-8
Ni-59	300			3.7E-06
Ni-63	50			8.5E-10
Np-237	15			2.1E-05
Pa-231	15			3.2E-07
Pb-210			0.054	7.1E-13
Pm-147	587			6.3E-10
Pu-238	15			8.8E-10

Table 2.3 Radionuclide Drinking Water MCLs

Radionuclide	Current MCL ^{a, b} (pCi/L)	Proposed MCL (pCi/L)	Risk Base Limit (RBL) ^e (pCi/L)	Mass Equiv to MCL, Proposed MCL, or RBL (mg/L)
Pu-239	15			2.4E-07
Pu-240	15			6.6E-08
Pu-241			27	2.6E-10
Pu-242	15			3.8E-06
Pu-244	15			8.5E-04
Ra-226	5 ^c			5.1E-09
Ra-228	5 ^c			1.8E-11
Ru-106	30			9.0E-12
Sb-125	300			2.9E-10
Sm-147	15			6.5E-01
Sm-151	1,000			3.8E-08
Sr-90	8			5.9E-11
Tc-99	900			5.3E-05
Th-228	15			1.8E-11
Th-229	15			7.1E-08
Th-230	15			7.4E-07
Th-232	15			1.4E-01
Tl-204	300			6.5E-10
U-232		20 ^d		9.4E-10
U-232		(20 µg/l) ^d		2.0E-02
U-233		20 ^d		2.1E-06
U-233		(20 µg/l) ^d		2.0E-02
U-234		20 ^d		3.2E-06
U-234		(20 µg/l) ^d		2.0E-02
U-235		20 ^d		9.3E-03
U-235		(20 µg/l) ^d		2.0E-02
U-236		20 ^d		3.1E-04
U-236		(20 µg/l) ^d		2.0E-02
U-238		20 ^d		6.0E-02
U-238		(20 µg/l) ^d		2.0E-02
Zn-65	300			3.6E-11

Notes:

- ^a Existing MCL is 4 mrem/yr to the whole body or an organ, combined from all beta and photon emitters.
- ^b Existing MCL is 15 pCi/L, with the concentration level combined for all alpha emitters, except radon and uranium.
- ^c Existing MCL is 5 pCi/L combined for Ra-226 and Ra-228.
- ^d Preferred EPA proposed MCL standard is 20 µg/l and 20 pCi/l for uranium, with EPA soliciting comments on options of 40 µg/l **and** 40 pCi/l, and 80 µg/l **and** 80 pCi/l. The preferred proposed MCL standard for uranium of 20 µg/l and 20 pCi/l is represented in this table.
- ^e Risk Based Limits are calculated for 30-year exposure duration and 10⁻⁶ risk.

2.1.1 Additive Risk. For all pathways except migration to ground water, SSLs correspond to a 10^{-6} lifetime cancer risk level. While the pathways included in the analysis are considered to represent a residential setting, SSLs are not calculated for a specific scenario (i.e., SSLs are not summed over a set of pathways). EPA believes that setting a 10^{-6} risk level for individual radionuclides and pathways generally will lead to cumulative risks within the 10^{-4} to 10^{-6} range for the combinations of radionuclides typically found at Superfund sites.

2.1.2 Toxicokinetics of Radionuclides. The exposure assumptions used to develop SSLs are representative of a chronic exposure pathway. Exposures necessary to cause acute radiation effects are many orders of magnitude greater than those associated with SSLs. Therefore, it is extremely unlikely that even massive intakes of contaminated soil could cause such effects. It should be noted, however, that the slope factors used to calculate SSLs in this report are applicable to either chronic or acute exposure to a radionuclide. That is, the slope factor may be interpreted either as the average risk per unit exposure to members of a population exposed throughout life to a constant concentration of a radionuclide in soil, or as the average risk per unit exposure to members of a population acutely exposed to the radionuclide in soil.

2.2 Direct Ingestion of Soil

Calculation of SSLs for direct ingestion of soil is based on the methodology presented for residential land use in RAGS HHEM, Part B (U.S. EPA, 1991b). Briefly, this methodology backcalculates a soil concentration level from a target risk. A number of studies have shown that inadvertent ingestion of soil is common among children 6 years old and younger (Calabrese et al., 1989; Davis et al., 1990; Van Wijnen et al., 1990). Therefore, the approach uses an age-adjusted soil ingestion factor that takes into account the difference in daily soil ingestion rates and exposure duration for children from 1 to 6 years old and others from 7 to 31 years old. The higher intake rate of soil by children leads to a lower, or more conservative, risk-based concentration compared to an adult-only assumption. RAGS HHEM, Part B also uses this age-adjusted approach.

For radionuclides, both the magnitude and duration of exposure are important. Duration is critical because the toxicity criteria are based on "lifetime average daily dose." Therefore, the total intake, whether it be over 5 years or 50 years, is averaged over a lifetime of 70 years. To be protective of exposures to radionuclides in the residential setting, RAGS HHEM, Part B (U.S. EPA, 1991b) and EPA focus on exposures to individuals who may live in the same residence for a "high-end" period of time (e.g., 30 years). As mentioned above, exposure to soil is higher during childhood and decreases with age. Thus, Equation 1 uses the RAGS HHEM, Part B time-weighted average soil ingestion rate for children and adults; the derivation of this factor is shown in Equation 2.

Screening Level Equation for Ingestion of Radionuclides in Residential Soil

(Source: RAGS HHEM, Part B; U.S. EPA, 1991b)

$$SSL = \frac{TR}{SF_s * IR_s * 1E-3 * EF * ED} \quad (1)$$

Parameter/Definition (units)	Default
TR/target cancer risk (unitless)	10 ⁻⁶
SF _s /soil ingestion slope factor (pCi) ⁻¹	See Table 2.2
IR _s /soil ingestion rate (mg/d)	120 (age-averaged)
1E-3/conversion factor (g/mg)	--
EF/exposure frequency (d/yr)	350
ED/exposure duration (yr)	30

Source: RAGS HHEM, Part B; U.S. EPA, 1991b

Equation for Age-Adjusted Soil Ingestion Factor, IR_s

$$IR_s = \frac{IR_{soil/age1-6} * ED_{age1-6} + IR_{soil/age7-31} * ED_{age7-31}}{ED} \quad (2)$$

Parameter/Definition (units)	Default
IR _s /age-adjusted soil ingestion factor (mg/d)	120
IR _{soil/age1-6} /ingestion rate of soil age 1-6 (mg/d)	200
ED _{age1-6} /exposure duration during ages 1-6 (yr)	6
IR _{soil/age7-31} /ingestion rate of soil age 7-31 (mg/d)	100
ED _{age7-31} /exposure duration during ages 7-31 (yr)	24
ED/exposure duration (yr)	30

Source: RAGS HHEM, Part B (U.S. EPA, 1991b).

Because of the impracticability of developing site-specific input parameters (e.g., soil ingestion rates) for direct soil ingestion, SSLs are calculated using the defaults listed in Equations 1 and 2. Appendix A lists these generic SSLs for direct ingestion of soil.

2.3 Inhalation of Fugitive Dusts

The models and assumptions used to calculate SSLs for the inhalation pathway are presented in Equations 3 and 4, along with the default parameter values used to calculate the generic SSLs presented in Appendix A. Particular attention is given to the emissions portion and the dispersion portion (Q/F) of the particulate emission factor (PEF) equation, which has been revised since originally presented in RAGS HHEM, Part B. The default PEF presented here is the same as that given in the SSG for chemicals, which allows for the presence of a family garden associated with the ingestion of homegrown produce pathway in a residential setting. The available radionuclide-specific human health benchmarks used in these equations are presented in Section 2.1.

2.3.1 Screening Level Equation for Inhalation of Fugitive Dusts. Equation 3 is used to calculate SSLs for the inhalation of fugitive dusts containing radionuclides.

Screening Level Equation for Inhalation of Radionuclides in Fugitive Dusts in Residential Soil

$$SSL = \frac{TR}{SF_i * IR_i * \left(\frac{1}{PEF}\right) * 1E+3 * EF * ED * [ET_o + (ET_i * DF_i)]} \quad (3)$$

Parameter/Definition (units)	Default	Source
TR/target cancer risk (unitless)	10 ⁻⁶	U.S. EPA, 1991b
SF _i /inhalation slope factor (pCi ⁻¹)	See Table 2.2	See Part 2.1
IR _i /inhalation rate (m ³ /d)	20	U.S. EPA, 1991b
PEF/particulate emission factor (m ³ /kg)	1.32E+09 (Equation 3)	U.S. EPA, 1991b
1E+3/conversion factor (g/kg)	--	--
EF/exposure frequency (d/yr)	350	U.S. EPA, 1991b
ED/exposure duration (yr)	30	U.S. EPA, 1991b
ET _o /exposure time fraction, outdoor (unitless)	0.073	U.S. EPA, 1997b
ET _i /exposure time fraction, indoor (unitless)	0.683	U.S. EPA, 1997b
DF _i /dilution factor for indoor inhalation, (unitless)	0.4	Alonza, 1979

To calculate inhalation SSLs, the particulate emission factor must be calculated. The derivation of PEF has been updated since RAGS HHEM, Part B was published and is discussed fully in Sections 2.3.3. The PEF equation can be broken into two separate models: models to estimate the emissions of dusts, and a dispersion model (reduced to the term Q/C) that simulates the dispersion of radionuclides in the atmosphere.

2.3.2 Dispersion Model. The box model in RAGS HHEM, Part B has been replaced with a Q/C term derived from a modeling exercise using meteorologic data from 29 locations across the United States.

The dispersion model used in the Part B guidance is based on the assumption that emissions into a hypothetical box will be distributed uniformly throughout the box. To arrive at the volume within the box, it is necessary to assign values to the length, width, and height of the box. The length (LS) was the length of a side of a contaminated site with a default value of 45 m; the width was based on the windspeed in the mixing zone (V) with a default value of 2.25 m (based on a windspeed of 2.25 m/s); and the height was the diffusion height (DH) with a default value of 2 m.

However, the assumptions and mathematical treatment of dispersion used in the box model may not be applicable to a broad range of site types and meteorology and do not utilize state-of-the-art techniques developed for regulatory

dispersion modeling. EPA was very concerned about the defensibility of the box model and sought a more defensible dispersion model that could be used as a replacement to the Part B guidance and had the following characteristics:

- Dispersion modeling from a ground-level area source
- Onsite receptor
- A long-term/annual average exposure point concentration
- Algorithms for calculating the exposure point concentration for area sources of different sizes and shapes.

To identify such a model, EPA held discussions with the EPA Office of Air Quality Planning and Standards (OAQPS) concerning recent efforts to develop a new algorithm for estimating ambient air concentrations from low or ground-level, nonbuoyant sources of emissions. The new algorithm is incorporated into the Industrial Source Complex Model (ISC2) platform in both a short-term mode (AREA-ST) and a long-term mode (AREA-LT). Both models employ a double numerical integration over the source in the upwind and crosswind directions. Wind tunnel tests have shown that the new algorithm performs well with onsite and near-field receptors. In addition, subdivision of the source is not required for these receptors.

Because the new algorithm provides better concentration estimates for onsite and for near-field receptors, a revised dispersion analysis was performed (Appendix D; EQ, 1994). The AREA-ST model was run for 0.5-acre and 30-acre square sources with a full year of meteorologic data for 29 U.S locations selected to be representative of the national range of meteorologic conditions (EQ, 1993). Additional modeling runs were conducted to address a range of square area sources from 0.5 to 30 acres in size (Table 2.4). The Q/C values in Table 2.4 for 0.5- and 30-acre sources differ slightly from the values in Appendix D due to differences in rounding conventions used in the final model runs.

To calculate site-specific SSLs, select a Q/C value from Table 2.4 that best represents a site's size and meteorologic condition.

To develop a reasonably conservative default Q/C for calculating generic SSLs, a default site (Minneapolis, MN) was chosen that best approximated the 90th percentile of the 29 normalized concentrations (kg/m^3 per $\text{g}/\text{m}^2\text{-s}$). The inverse of this concentration results in a default Q/C value of 90.80 $\text{g}/\text{m}^2\text{-s}$ per kg/m^3 for a 0.5-acre site.

Table 2.4 . Q/C Values by Source Area, City, and Climatic Zone

	Q/C (g/m ² -s per kg/m ³)					
	0.5 Acre	1 Acre	2 Acre	5 Acre	10 Acre	30 Acre
Zone I						
Seattle	82.72	72.62	64.38	55.66	50.09	42.86
Salem	73.44	64.42	57.09	49.33	44.37	37.94
Zone II						
Fresno	62.00	54.37	48.16	41.57	37.36	31.90
Los Angeles	68.81	60.24	53.30	45.93	41.24	35.15
San Francisco	89.51	78.51	69.55	60.03	53.95	46.03
Zone III						
Las Vegas	95.55	83.87	74.38	64.32	57.90	49.56
Phoenix	64.04	56.07	49.59	42.72	38.35	32.68
Albuquerque	84.18	73.82	65.40	56.47	50.77	43.37
Zone IV						
Boise	69.41	60.88	53.94	46.57	41.87	35.75
Winnemucca	69.23	60.67	53.72	46.35	41.65	35.55
Salt Lake City	78.09	68.47	60.66	52.37	47.08	40.20
Casper	100.13	87.87	77.91	67.34	60.59	51.80
Denver	75.59	66.27	58.68	50.64	45.52	38.87
Zone V						
Bismark	83.39	73.07	64.71	55.82	50.16	42.79
Minneapolis	90.80	79.68	70.64	61.03	54.90	46.92
Lincoln	81.64	71.47	63.22	54.47	48.89	41.65
Zone VI						
Little Rock	73.63	64.51	57.10	49.23	44.19	37.64
Houston	79.25	69.47	61.53	53.11	47.74	40.76
Atlanta	77.08	67.56	59.83	51.62	46.37	39.54
Charleston	74.89	65.65	58.13	50.17	45.08	38.48
Raleigh-Durham	77.26	67.75	60.01	51.78	46.51	39.64
Zone VII						
Chicago	97.78	85.81	76.08	65.75	59.16	50.60
Cleveland	83.22	73.06	64.78	55.99	50.38	43.08
Huntington	53.89	47.24	41.83	36.10	32.43	27.67
Harrisburg	81.90	71.87	63.72	55.07	49.56	42.40
Zone VIII						
Portland	74.23	65.01	57.52	49.57	44.49	37.88
Hartford	71.35	62.55	55.40	47.83	43.00	36.73
Philadelphia	90.24	79.14	70.14	60.59	54.50	46.59
Zone IX						
Miami	85.61	74.97	66.33	57.17	51.33	43.74

2.3.3 Particulate Emission Factor. The particulate emission factor relates the concentration of contaminant in soil with the concentration of dust particles in the air. This guidance addresses dust generated from open sources, which is termed "fugitive" because it is not discharged into the atmosphere in a confined flow stream. Other sources of fugitive dusts that may lead to higher emissions due to mechanical disturbances include unpaved roads, tilled agricultural soils, and heavy construction operations.

Both the emissions portion and the dispersion portion (Q/C) of the PEF equation have been updated since RAGS HHEM, Part B.

As in Part B, the emissions part of the PEF equation is based on the "unlimited reservoir" model from Cowherd et al. (1985) developed to estimate particulate emissions due to wind erosion. The unlimited reservoir model is most sensitive to the threshold friction velocity, which is a function of the mode of the size distribution of surface soil aggregates. This parameter has the greatest effect on the emissions and resulting concentration. For this reason, a conservative mode soil aggregate size of 500 μm was selected as the default value for calculating generic SSLs.

The mode soil aggregate size determines how much wind is needed before dust is generated at a site. A mode soil aggregate size of 500 μm yields an **uncorrected** threshold friction velocity of 0.5 m/s. This means that the windspeed must be at least 0.5 m/s before any fugitive dusts are generated. However, the threshold friction velocity **should be corrected** to account for the presence of nonerodible elements. In Cowherd et al. (1985), nonerodible elements are described as:

... clumps of grass or stones (larger than about 1 cm in diameter) on the surface (that will) consume part of the shear stress of the wind which otherwise would be transferred to erodible soil.

Cowherd et al. describe a study by Marshall (1971) that used wind tunnel studies to quantify the increase in the threshold friction velocity for different kinds of nonerodible elements. His results are presented in Cowherd et al. as a graph showing the rate of corrected to uncorrected threshold friction velocity vs. L_c , where L_c is a measure of nonerodible elements vs. bare, loose soil. Thus, the ratio of corrected to uncorrected threshold friction velocity is directly related to the amount of nonerodible elements in surface soils.

Using a ratio of corrected to uncorrected threshold friction velocity of 1, or no correction, is roughly equivalent to modeling "coal dust on a concrete pad," whereas using a correction factor of 2 corresponds to a windspeed of 19 m/s at a height of 10 m. This means that about a 43-mph wind would be required to produce any particulate emissions. Given that the 29 meteorologic data sets used in this modeling effort showed few windspeeds at, or greater than, 19 m/s, EPA felt that it was necessary to choose a default correction ratio between 1 and 2. A value of 1.25 was selected as a reasonable number that would be at the more conservative end of the range. This equates to a corrected threshold friction velocity of 0.625 m/s and an equivalent windspeed of 11.3 m/s at a height of 7 meters. The default PEF is the same as for chemicals. Another, key assumption in the derivation of the PEF is that the $\frac{1}{2}$ acre lot has only 50% vegetative cover. Although the ingestion of homegrown produce is not quantitatively evaluated in the SSG for chemicals, the assumption of 50% vegetative cover allows for the presence of a family garden.

Q/C values are needed to calculate the PEF (Equation 4); use the Q/C value in Table 2.4 that best represents a site's size and meteorologic conditions. Cowherd et al. (1985) describe how to obtain site-specific estimates of V , U_m , U_t , and $F(x)$.

Meteorologic conditions (i.e., the intensity and frequency of wind) affect both the dispersion and emissions of particulate matter. For this reason, a separate default Q/C value was derived for particulate matter [nominally 10 μm and less (PM_{10})] emissions for the generic SSLs. The PEF equation was used to calculate annual average concentrations for each of 29 sites across the country. To develop a reasonably conservative default Q/C for calculating generic SSLs, a default site (Minneapolis, MN) was selected that best approximated the 90th percentile concentration.

The results produced a revised default PEF Q/C value of 90.80 $\text{g}/\text{m}^2\text{-s}$ per kg/m^3 for a 0.5-acre site (see Appendix D; EQ, 1994). The generic PEF derived using the default values in Equation 4 is $1.32 \times 10^9 \text{ m}^3/\text{kg}$, which corresponds to a receptor point concentration of approximately $0.76 \mu\text{g}/\text{m}^3$. This represents an annual average emission rate based on wind erosion that should be compared with chronic health criteria; it is **not** appropriate for evaluating the potential for more acute exposures.

Derivation of the Particulate Emission Factor

$$PEF = Q/C * \frac{3,600}{0.036 * (1-V) * (U_m/U_t)^3 * F(x)} \quad (4)$$

Parameter/Definition (units)	Default	Source
PEF/particulate emission factor (m ³ /kg)	1.32 x 10 ⁹	- -
Q/C/inverse of mean conc. at center of square source (g/m ² -s per kg/m ³)	90.80	Table 3 (for 0.5-acre source in Minneapolis, MN)
V/fraction of vegetative cover (unitless)	0.5 (50%)	U.S. EPA, 1991b
U _m /mean annual windspeed (m/s)	4.69	EQ, 1994
U _t /equivalent threshold value of windspeed at 7 m (m/s)	11.32	U.S. EPA, 1991b
F(x)/function dependent on U _m /U _t derived using Cowherd et al. (1985) (unitless)	0.194	U.S. EPA, 1991b

2.4 External Exposure from Radionuclides in Soil

Individuals residing on a contaminated site will be exposed to photons emitted by those radionuclides present in the soil. In modeling external exposure to contaminated soil, the RAGS/HHEM Part B model (EPA91a) does not account for the following processes:

- radioactive decay and progeny ingrowth (i.e., radioactive daughters);
- correction factors for the geometry of the contaminated soil;
- depletion of the contaminated soil horizon by environmental processes, such as leaching, erosion, or plant uptake; and
- corrections for shielding by clean cover material.

The RAGS/HHEM Part B model effectively assumes that an individual is continually exposed to a non-depleting source term with a geometry that is effectively an infinite slab. The concept of an “infinite slab” means that the thickness of the contaminated zone and its aerial extent are so large that it behaves as if it were infinite in its physical dimensions. In practice, soil contaminated to a depth greater than about 15 cm and with an aerial extent greater than about 1,000 m² will create a radiation field comparable to that of an infinite slab. For calculation of SSLs for a residential setting, an adjustment for small areas is considered to be an important modification to the RAGS/HHEM Part B model, since in most residential settings the assumption of an infinite slab source will result in overly conservative SSLs. Thus, an area correction factor, ACF, has been added to the model for the calculation of SSLs. For the purposes of this report, adjustments for clean cover are not needed since, in all cases, it is assumed that the contaminated soil extends to the surface. The model provides adjustments for indoor occupancy and associated shielding effects by the simple application of a shielding factor and indoor occupancy time adjustment.

Screening Level Equation for External Exposure to Radionuclides in Soil

$$SSL = \frac{TR}{SF_e * \frac{EF}{365} * ED * ACF * [ET_o * (ET_i * GSF)]} \quad (5)$$

Parameter/Definition (units)	Default	Source
TR/target cancer risk (unitless)	10 ⁻⁶	U.S. EPA, 1991b
SF _e /external exposure slope factor (g/pCi/yr)	See Table 2.2	See Part 2.1
EF/exposure frequency (d/yr)	350	U.S. EPA, 1991b
ED/exposure duration (yr)	30	U.S. EPA, 1991b
ACF/area correction factor	0.9	ANL, 1993b
ET _o /exposure time fraction, outdoor (unitless)	0.073	U.S. EPA, 1997b
ET _i /exposure time fraction, indoor (unitless)	0.683	U.S. EPA, 1997b
GSF/gamma shielding factor	0.4	U.S. EPA, 1996d

With the exception of the area correction factor, default values are used for all input parameters in Equation 5 to calculate generic external exposure SSLs. The amount of data required to derive site-specific values for these parameters makes their collection and use impracticable for calculation of simple site-specific SSLs. Therefore, site-specific data are generally not available for this exposure pathway. An area correction factor less than 0.9 will only apply to very small sites (i.e., those with an area less than 1,000 m²). Alternative area correction factors are discussed in Section 5.1.

2.4.1. Gamma Shielding Factor - GSF (unitless)

The gamma shielding factor is the ratio of the external gamma radiation level indoors on site to the radiation level outdoors on-site. It is based on the fact that a building provides shielding against penetration of gamma radiation. Therefore, the calculation of the risk posed by gamma radiation from radionuclides in the soil should take into account this shielding effect.

EPA's previous gamma shielding factor default value, taken from RAGS/HHEM Part B (U.S. EPA, 1991b), is 0.8, which assumes that the external gamma radiation level indoors is 20% lower than the outdoor gamma radiation level. This value is based on information presented in two EPA reports, *Natural Radiation in the United States* (U.S. EPA, 1972) and *Population Exposure to External Natural Radiation Background in the United States* (U.S. EPA, 1981). Based on a review of the literature provided in these reports, EPA concluded that external background exposures in frame dwellings are 70 to 80% of outdoor values. This conclusion is based on empirical data for natural background radiation. As such, it includes the contribution from both terrestrial and cosmic radiation and from the radionuclides present in structure material.

A further review of the literature performed in the EPA report, *Reassessment of Radium and Thorium Soil Concentrations and Annual Dose Rates* (U.S. EPA, 1996d) reveals numerous publications that address indoor/outdoor gamma ray shielding factors as applied to radioactive fallout from nuclear weapons and reactor accidents. In U.S. EPA, 1981, the authors performed a review of experimentally measured reduction factors from fallout. The authors concluded that "reduction factors of 0.4 to 0.2 are recommended as representative values for above-ground lightly constructed (wood frame) and heavily constructed (block and brick) homes, respectively." On the basis of this review, U.S. EPA 1996d, suggests that a default gamma shielding factor of 0.4 based solely on the contribution of terrestrial radiation might be a more appropriate value to use at sites with soil contaminated with radionuclides than the previous EPA default of 0.8 which also included the effects of cosmic radiation and the inherent radioactivity in structure materials. Based on this rationale, the value of 0.4 is adopted in this guidance as EPA's new default gamma shielding factor.

2.5 Ingestion of Homegrown Produce.

Persons living on a contaminated site may ingest radioactive material by consumption of plants grown in a family garden. In this model, the fruits and vegetables primarily become contaminated by root uptake of radionuclides contained in the pore water of the soil in which the plants are growing.

Residential gardens that provide a significant fraction of the family diet are not uncommon, particularly in more rural areas. However, it is quite uncommon that such a family garden would be totally self-sufficient. To be self-sufficient, the family would have to raise all of its vegetables onsite—that is, grow its own potatoes, get all fruit from an onsite orchard, can or freeze enough of the summer crop to provide all food during the winter, and raise all grain used for bread and cereal. The model uses available consumption data for home grown produce from USDA surveys.

The model accounts for root uptake with a simple soil-to-plant transfer factor. These soil-to-plant transfer factors have been developed based upon the assumption that the entire plant root system is wholly exposed to contaminated soil. If the plant roots extend to a depth of 100 cm but the radionuclide contaminants are confined to the upper 15 cm, an initial assumption may be that only 15% of the root system is active in accumulating contaminants and that the reported soil-to-plant transfer factors should be reduced by a correction factor of 0.15. However, the equation for calculation of SSLs for this pathway does not apply any reduction to the soil-to-plant transfer factors. The basis for this assumption is as follows.

Most plant root systems are in fact very active in the upper soil horizon, especially in the upper 15 cm of soil. This point is illustrated in a number of ways: 1) by illustrations of root morphology and growth habit, 2) positive physiological factors including the availability of water, oxygen and nutrients near the soil surface, 3) negative physiological or agronomic factors—including subsurface soil compaction, subsurface zones of acidity, perched water tables, hypoxia, etc., 4) interactions with soil microbes—with a special focus on mycorrhizal fungi, and 5) split root studies. Thus, roots commonly proliferate in the upper layers of soil. If one assumes that a plant is actively growing, then ion uptake characteristics and lateral root growth strongly suggest that simply attributing 15% of root uptake activity to the upper 15 cm of the soil is not a sound approach. Environmental forces may influence root growth to one or more meters in depth, but more so for obtaining water than nutrients. In reality, the upper 15 cm of soil may include 50% or more of the root system—and thus 50% or more of the ion uptake (SC&A, 1994).

The decision to not include air deposition or rain-splash does not affect any radionuclides because the increase in concentration from this route is not significant or is markedly reduced when peeling, washing, cooking, and other food preparation processes are taken into consideration (U.S. EPA, 1994k). The decision to not include the irrigation pathway is only an issue when there is medium to heavy irrigation using contaminated water for a radionuclide with a long half-life, and an insignificant contribution from external exposure. The model also makes a conservative assumption to ignore the decay between harvest and ingestion and any removal during food processing.

The model does not include any special calculations for estimating concentrations of ^3H and ^{14}C in plants. Such calculations assume that a state of equilibrium exists among the concentrations of ^3H and ^{14}C in all environmental media—air, water, food products, and body tissues. This assumption may be conservative for a radioactively contaminated site with a finite area, but may be appropriate for an individual pathway, such as soil-to-plant pathway (ANL93a). For these calculations, the ^3H concentration in the plant is assumed to be the same as that in the contaminated water to which the plant is exposed. Similarly, the specific activity of ^{14}C in the plant (i.e., pCi/g of ^{14}C per gram of carbon in the plant) is the same as that of the ambient CO_2 .

Other models use a factor, known as the contaminated plant fraction (CPF), to adjust for the fraction of fruits and vegetables obtained from the contaminated site relative to the total consumption rate. The ingestion rate used in these models is a total ingestion rate, which, when multiplied by the contaminated plant fraction, gives the ingestion rate of contaminated fruits and vegetables. However, the calculation of SSLs for this pathway makes use of published estimates of individual home produced fruit and vegetable intake, thus eliminating the need for a CPF.

Default values are used for all input parameters in Equation 5 to calculate SSLs for this pathway. The amount of data required to derive site-specific values for these parameters makes their collection and use impracticable for calculation of simple site-specific SSLs. Therefore, site-specific data are generally not available for this exposure pathway. The generic SSLs presented in the TBD are recommended for all sites.

Screening Level Equation for Ingestion of Radionuclides in Homegrown Produce

$$SSL = \frac{TR}{SF_p * (IR_f + IR_v) * TF_p * EF * ED} \quad (6)$$

Parameter/Definition (units)	Default	Source
TR/target cancer risk (unitless)	10 ⁻⁶	U.S. EPA, 1991b
SF _p /produce ingestion slope factor (pCi) ⁻¹	See Table 2.2	See Part 2.1
IR _f /home grown fruits ingestion rate (g/d)	189	U.S. EPA, 1997b
IR _v /home grown vegetables ingestion rate (g/d)	147	U.S. EPA, 1997b
TF _p /soil-to-plant transfer factor (pCi/g plant per pCi/g soil)	See Part 3.5.5	U.S. EPA, 1997b
EF/exposure frequency (d/yr)	350	U.S. EPA, 1991b
ED/exposure duration (yr)	30	U.S. EPA, 1991b

2.6 Migration to Ground Water

The methodology for calculating SSLs for the migration to ground water pathway was developed to identify radionuclide concentrations in soil that have the potential to contaminate ground water. Migration of radionuclides from soil to ground water can be envisioned as a two-stage process: (1) release of contaminant in soil leachate and (2) transport of the contaminant through the underlying soil and aquifer to a receptor well. The SSL methodology considers both of these fate and transport mechanisms.

The methodology incorporates a standard linear equilibrium soil/water partition equation to estimate radionuclide release in soil leachate (see Sections 2.6.1 through 2.6.4) and a simple water-balance equation that calculates a dilution factor to account for dilution of soil leachate in an aquifer (see Section 2.6.5). The dilution factor represents the reduction in soil leachate radionuclide concentrations by mixing in the aquifer, expressed as the ratio of leachate concentration to the concentration in ground water at the receptor point (i.e., drinking water well). Because the infinite source (i.e., steady-state) assumption can result in mass-balance violations for soluble contaminants and small sources, mass-limit models are provided that limit the amount of contaminant migrating from soil to ground water to the total amount of contaminant present in the source (see Section 2.7).

SSLs are backcalculated from acceptable ground water concentrations (i.e., MCLs, see Section 2.1). First, the acceptable ground water concentration is multiplied by a dilution factor to obtain a target leachate concentration. For example, if the dilution factor is 10 and the acceptable ground water concentration is 10 pCi/L, the target soil leachate concentration would be 100 pCi/L. The partition equation is then used to calculate the total soil concentration (i.e., SSL) corresponding to this soil leachate concentration.

The methodology for calculating SSLs for the migration to ground water pathway was developed under the following constraints:

- Because of the large nationwide variability in ground water vulnerability, the methodology should be flexible, allowing adjustments for site-specific conditions if adequate information is available.
- To be appropriate for early-stage application, the methodology needs to be simple, requiring a minimum of site-specific data.
- The methodology should be consistent with current understanding of subsurface processes.

- The process of developing and applying SSLs should generate information that can be used and built upon as a site evaluation progresses.

Flexibility is achieved by using readily obtainable site-specific data in standardized equations; conservative default input parameters are also provided for use when site-specific data are not available. In addition, more complex unsaturated zone fate-and-transport models have been identified that can be used to calculate SSLs when more detailed site-specific information is available or can be obtained (see Part 3). These models can extend the applicability of SSLs to subsurface conditions that are not adequately addressed by the simple equations (e.g., deep water tables; clay layers or other unsaturated zone characteristics that can attenuate contaminants before they reach ground water).

The SSL methodology was designed for use during the early stages of a site evaluation when information about subsurface conditions may be limited. Because of this constraint, the methodology is based on conservative, simplifying assumptions about the release and transport of contaminants in the subsurface (see Highlight 2).

Highlight 2: Simplifying Assumptions for the Migration to Ground Water Pathway

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

Although simplified, the SSL methodology described in this section is theoretically and operationally consistent with the more sophisticated investigation and modeling efforts that are conducted to develop soil cleanup goals and cleanup levels for protection of ground water at Superfund sites. SSLs developed using this methodology can be viewed as evolving risk-based levels that can be refined as more site information becomes available. The early use of the methodology at a site will help focus further subsurface investigations on areas of true concern with respect to ground water quality and will provide information on soil characteristics, aquifer characteristics, and radionuclide properties that can be built upon as a site evaluation progresses.

2.6.1 Development of Soil/Water Partition Equation. The methodology used to estimate radionuclide release in soil leachate is based on the Freundlich equation, which was developed to model sorption from liquids to solids. The basic Freundlich equation applied to the soil/water system is:

$$K_d = C_s / C_w^n \quad (7)$$

where

- K_d = Freundlich soil/water partition coefficient (L/kg)
- C_s = concentration sorbed on soil (mg/kg)
- C_w = soil leachate concentration (mg/L)
- n = Freundlich exponent (dimensionless).

Assuming that adsorption is linear with respect to concentration ($n=1$) and rearranging to backcalculate a sorbed concentration (C_s):

$$C_s = (K_d) C_w \quad (8)$$

For SSL calculation, C_1 is the target soil leachate concentration.

Adjusting Sorbed Soil Concentrations to Total Concentrations. To develop a screening level for comparison with contaminated soil samples, the sorbed concentration derived above (C_s) must be related to the total concentration measured in a soil sample (C_t). In a soil sample, contaminants can be associated with the solid soil materials or the soil water, and the soil air as follows (Feenstra et al., 1991):

$$M_t = M_s + M_w + M_a \quad (9)$$

where

- M_t = total contaminant mass in sample (mg)
- M_s = contaminant mass sorbed on soil materials (mg)
- M_w = contaminant mass in soil water (mg)
- M_a = contaminant mass in soil air (mg).

Furthermore,

$$M_t = C_t \rho_b V_{sp}, \quad (10)$$

$$M_s = C_s \rho_b V_{sp}, \quad (11)$$

$$M_w = C_w \theta_w V_{sp}, \quad (12)$$

and

$$M_a = C_a \theta_a V_{sp}, \quad (13)$$

where

- ρ_b = dry soil bulk density (kg/L)
- V_{sp} = sample volume (L)
- θ_w = water-filled porosity (L_{water}/L_{soil})
- C_a = concentration on soil pore air (mg/ L_{soil})
- θ_a = air-filled soil porosity (L_{air}/L_{soil}).

For contaminated soils (with concentrations below C_{sat}), C_a may be determined from C_w and the dimensionless Henry's law constant (H') using the following relationship:

$$C_a = C_w H^1 \quad (14)$$

thus

$$M_a = C_w H^1 \theta_a V_{sp} \quad (15)$$

Substituting into Equation 9:

$$C_t = \frac{C_s \rho_b + C_w \theta_w + C_w H^1 \theta_a}{\rho_b} \quad (16)$$

or

$$C_s = C_t - C_w \left(\frac{\theta_w + \theta_a H^1}{\rho_b} \right) \quad (17)$$

Substituting into Equation 8 and rearranging:

$$C_t = C_w * \left(K_d + \frac{\theta_w + \theta_a * H^1}{\rho_b} \right) \quad (18)$$

For most radionuclides there is no significant vapor pressure and the dimensionless Henry's Law constant (H^1) may be assumed to be zero, (except for radon). When H^1 is zero, Equation 18 becomes:

Soil-Water Partition Equation for Migration to Ground Water: Inorganic Contaminants:

$$SSL = C_t = C_w \left(K_d + \frac{\theta_w}{\rho_b} \right) \quad (19)$$

Parameter/Definition (units)	Default	Source
SSL /screening level in soil (mg/kg)	--	--
C_w /target soil leachate concentration (mg/L)	(MCL) × 20 DAF	Table 2.3 (MCL); Section 2.6.2 (DAF for 0.5-acre source)
K_d /soil-water partition coefficient (L/kg)	radionuclide -specific	see Section 3.3.1
θ_w /water-filled soil porosity (L_{water}/L_{soil})	0.3 (30%)	U.S. EPA/ORD
n /total soil porosity (L_{pore}/L_{soil})	0.43	$1 - \rho_b/\rho_s$
ρ_b /dry soil bulk density (kg/L)	1.5	U.S. EPA, 1991b
ρ_s /soil particle density (kg/L)	2.65	U.S. EPA, 1991b

Since MCLs for radionuclides are given in radiological activity units of picocuries per liter of water (pCi/l), equation 19 can also be written as:

$$SSL = C_t = C_w (1 * 10^{-3}) \left(K_d + \frac{\theta_w}{\rho_b} \right) \quad (20)$$

Parameter/Definition (units)	Default	Source
SSL /screening level in soil (pCi/g)	--	--
C_w /target soil leachate concentration (pCi/L)	(MCL) × 20 DAF	Table 2.3 (MCL); Section 2.6.2 (DAF for 0.5-acre source)
1×10^{-3} /conversion factor (kg/g)	--	--
K_d /soil-water partition coefficient (L/kg)	radionuclide -specific	see Section 3.3.1
θ_w /water-filled soil porosity (L_{water}/L_{soil})	0.3 (30%)	U.S. EPA/ORD
n/total soil porosity (L_{pore}/L_{soil})	0.43	$1 - \rho_b/\rho_s$
ρ_b /dry soil bulk density (kg/L)	1.5	U.S. EPA, 1991b
ρ_s /soil particle density (kg/L)	2.65	U.S. EPA, 1991b

Either Equation 19 or Equation 20 can be used to calculate SSLs (total soil concentrations, C_t) corresponding to soil leachate concentrations (C_w) equal to the target contaminant soil leachate concentration. The equation assumes that soil, water and solids are conserved during sampling.

Defaults. The User's Guide (U.S. EPA, 1996a) describes how to develop site-specific estimates of the soil parameters needed to calculate SSLs.

Sensitivity analyses have shown that soil bulk density (ρ_b) has too limited a range for surface soils (generally between 1.3 and 1.7 g/cm³) to affect results with nearly the significance of soil moisture conditions. Therefore, a default bulk density of 1.50 g/cm³, the mode of the range given for U.S. soils in the *Superfund Exposure Assessment Manual* (U.S. EPA, 1988), was chosen to calculate generic SSLs. This value is also consistent with the mean porosity (0.43) for loam soil presented in Carsel and Parrish (1988). The default value of ρ_s (2.65 g/cm³) was taken from U.S. EPA (1988) as the particle density for most soil mineral material.

Because migration to ground water SSLs are not particularly sensitive to soil water content (see Section 2.5.7), a value for average water-filled soil porosity that is more typical of subsurface conditions (0.30) was used. This value is between the mean field capacity (0.20) of Class B soils (Carsel et al., 1988) and the saturated volumetric water content for loam (0.43).

K_d varies by radioactive element and soil type, as discussed below.

2.6.2 Inorganics (Metals, including radionuclides)—Partition Theory. Equations 19 or 20 can be used to estimate SSLs for radionuclides for the migration to ground water pathway. The derivation of K_d values is much more complicated for metals, including radionuclides, than for organic compounds. Unlike organic compounds, for which K_d values are largely controlled by a single parameter (soil organic carbon), K_d values for metals are significantly affected by a variety of soil conditions. The most significant parameters are pH, oxidation-reduction conditions, iron oxide content, soil organic matter content, cation exchange capacity, and major ion chemistry. The number of significant influencing parameters, their variability in the field, and differences in experimental methods result in a wide range of K_d values for individual metals reported in the literature (over several orders of magnitude). Thus, it is much more difficult to derive generic K_d values for metals than for organics. For a more detailed discussion of the K_d model concept, see *Understanding Variation in Partition Coefficient, K_d Values, Volume I & Volume II* (U.S. EPA, 1999b).

2.6.3 Assumptions for Soil/Water Partition Theory. The following assumptions are implicit in the SSL partitioning methodology. *These assumptions and their implications for SSL accuracy should be read and understood before using this methodology to calculate SSLs.*

1. **There is no contaminant loss due to volatilization or degradation.** The source is considered to be infinite; i.e., these processes do not reduce soil leachate concentrations over time. This is a **conservative** assumption, especially for smaller sites.
2. **Adsorption is linear with concentration.** The methodology assumes that adsorption is independent of concentration (i.e., the Freundlich exponent = 1). This assumption is valid at low concentrations (e.g., at levels close to the MCL) for most chemicals. As concentrations increase, however, the adsorption isotherm can depart from the linear.
3. **The system is at equilibrium with respect to adsorption.** This ignores adsorption/desorption kinetics by assuming that the soil and pore water concentrations are at equilibrium levels. In other words, the pore-water residence time is assumed to be longer than the time it takes for the system to reach equilibrium conditions.

This assumption is **conservative**. If equilibrium conditions are not met, the concentration in the pore water will be less than that predicted by the methodology. The kinetics of adsorption are not adequately understood for a sufficient number of chemicals and site conditions to consider equilibrium kinetics in the methodology.

4. **Adsorption is reversible.** The methodology assumes that desorption processes operate in the same way as adsorption processes, since most of the partition coefficients are measured by adsorption experiments rather than by desorption experiments. In actuality, desorption is slower to some degree than adsorption and, in some cases, radionuclides can be irreversibly bound to the soil matrix. In general, the significance of this effect increases with increasing K_d . Note, adsorption is only one of several chemical mechanisms that may reduce the aqueous concentrations in soil or ground water. For a detailed discussion of these chemical mechanisms, see U.S. EPA, 1999b.

This assumption is **conservative**. Slower desorption rates and irreversible sorption will result in lower pore-water concentrations than that predicted by the methodology. Again, the level of knowledge on desorption processes is not sufficient to consider desorption kinetics and degree of reversibility for all of the subject radionuclides.

2.6.4 Dilution/Attenuation Factor Development. As radionuclides in soil leachate move through soil and ground water, they are subjected to physical, chemical, and biological processes that tend to reduce the eventual contaminant concentration at the receptor point (i.e., drinking water well). These processes include adsorption onto soil and aquifer media, chemical transformation (e.g., hydrolysis, precipitation), biological degradation, and dilution due to mixing of the leachate with ambient ground water. The reduction in concentration can be expressed succinctly by a DAF, which is defined as the ratio of radionuclides concentration in soil leachate to the concentration in ground water at the receptor point. When calculating SSLs, a DAF is used to backcalculate the target soil leachate concentration from an acceptable ground water concentration (e.g., MCL). For example, if the acceptable ground water concentration is 10 pCi/L and the DAF is 10, the target leachate concentration would be 100 pCi/L.

The SSL methodology addresses only one of these dilution-attenuation processes: radionuclide dilution in ground water. A simple equation derived from a geohydrologic water-balance relationship has been developed for the methodology, as described in the following subsection. The ratio factor calculated by this equation is referred to as a dilution factor rather than a DAF because it does not consider processes that attenuate radionuclides in the subsurface (i.e., adsorption and degradation processes). This simplifying assumption was necessary for several reasons.

First, the infinite source assumption (constant source concentrations maintained over period of interest) results in all subsurface adsorption sites being eventually filled and no longer available to attenuate radionuclides. Second, soil contamination extends to the water table, eliminating attenuation processes in the unsaturated zone. Additionally, the receptor well is assumed to be at the edge of the source, minimizing the opportunity for attenuation in the aquifer. Finally, chemical-specific biological and chemical degradation rates are not known for many of the SSL chemicals;

where they are available they are usually based on laboratory studies under simplified, controlled conditions. Because natural subsurface conditions such as pH, redox conditions, soil mineralogy, and available nutrients have been shown to markedly affect natural chemical and biological degradation rates, and because the national variability in these properties is significant and has not been characterized, EPA does not believe that it is possible at this time to incorporate these degradation processes into the simple site-specific methodology for national application.

If adsorption or degradation processes are expected to significantly attenuate radionuclide concentrations at a site (e.g., for sites with deep water tables or soil conditions that will attenuate radionuclides), the site manager is encouraged to consider the option of using more sophisticated fate and transport models. Many of these models can consider adsorption and degradation processes and can model transient conditions necessary to consider a finite source size. Part 3 of this document presents information on the selection and use of such models for SSL application.

The dilution factor model assumes that the aquifer is unconfined and unconsolidated and has homogeneous and isotropic properties. Unconfined (surficial) aquifers are common across the country, are vulnerable to contamination, and can be used as drinking water sources by local residents. Dilution model results may not be applicable to fractured rock or karst aquifer types. The site manager should consider use of more appropriate models to calculate a dilution factor (or DAF) for such settings.

In addition, the simple dilution model does not consider facilitated transport. This ignores processes such as colloidal transport, transport via solvents other than water (e.g., NAPLs), and transport via dissolved organic matter (DOM). However, the transport via solvents other than water is operative only if certain site-specific conditions are present. Transport by DOM and colloids has been shown to be potentially significant under certain conditions in laboratory and field studies. Although much research is in progress on these processes, the current state of knowledge is not adequate to allow for their consideration in SSL calculations.

If there is the potential for the presence of NAPLs in soils at the site or site area in question, SSLs should not be used for this area (i.e., further investigation is required). If NAPLs are suspected in site soils, refer to U.S. EPA (1992c) for additional guidance on how to estimate the potential for DNAPL occurrence in the subsurface.

Dilution Model Development. EPA evaluated four simple water balance models to adjust SSLs for dilution in the aquifer. Although written in different terms, all four options reviewed can be expressed as the same simple water balance equation to calculate a dilution factor, as follows:

Option 1 (ASTM):

$$\text{dilution factor} = (1 + U_{\text{gw}} d/IL) \tag{21}$$

where

- U_{gw} = Darcy ground water velocity (m/yr)
- d = mixing zone depth (m)
- I = infiltration rate (m/yr)
- L = length of source parallel to flow (m).

For Darcy velocity:

$$U_{\text{gw}} = Ki \tag{22}$$

where

- K = aquifer hydraulic conductivity (m/yr)
- i = hydraulic gradient (m/m).

Thus

$$\text{dilution factor} = 1 + (K_i d / I L) \quad (23)$$

Option 2 (EPA Ground Water Forum):

$$\text{dilution factor} = (Q_p + Q_A) / Q_p \quad (24)$$

where

$$\begin{aligned} Q_p &= \text{percolation flow rate (m}^3\text{/yr)} \\ Q_A &= \text{aquifer flow rate (m}^3\text{/yr)} \end{aligned}$$

For percolation flow rate:

$$Q_p = I A \quad (25)$$

where

$$A = \text{facility area (m}^2\text{)} = W L.$$

For aquifer flow rate:

$$Q_A = W d K_i \quad (26)$$

where

$$\begin{aligned} W &= \text{width of source perpendicular to flow (m)} \\ d &= \text{mixing zone depth (m)}. \end{aligned}$$

Thus

$$\text{dilution factor} = (I A + W d K_i) / I W L$$

$$\text{dilution factor} = 1 + (K_i d / I L) \quad (27)$$

Option 3 (Summers Model):

$$C_w = (Q_p C_p) / (Q_p + Q_A) \quad (28)$$

where

$$\begin{aligned} C_w &= \text{ground water contaminant concentration (mg/L)} \\ C_p &= \text{soil leachate concentration (mg/L)} \end{aligned}$$

given that

$$C_w = C_p/\text{dilution factor}$$

$$1/\text{dilution factor} = Q_p/(Q_p + Q_A)$$

or

$$\text{dilution factor} = (Q_p + Q_A)/Q_p \quad (\text{see Option 2})$$

Option 4 (EPA ORD/RSKERL):

$$\text{dilution factor} = (Q_p + Q_A)/Q_p = RX/RL \quad (29)$$

where

- R = recharge rate (m/yr) = infiltration rate (I, m/yr)
- X = distance from receptor well to ground water divide (m)

(Note that the intermediate equation is the same as **Option 2**.)

This option is a longer-term option that is not considered further in this analysis because valid X values are not currently available either nationally or for specific sites. EPA is considering developing regional estimates for these parameters.

Dilution Model Input Parameters. As shown, all three options for calculating contaminant dilution in ground water can be expressed as the same equation (equation 30):

Ground Water Dilution Factor

$$\text{dilution factor} = 1 + (Kid/IL) \quad (30)$$

Parameter/Definition (units)
K/aquifer hydraulic conductivity (m/yr)
i/hydraulic gradient (m/m)
d/mixing zone depth (m)
I/infiltration rate (m/yr)
L/source length parallel to ground water flow (m)

Mixing Zone Depth (d). Because of its dependence on the other variables, mixing zone depth is estimated with the method used for the MULTIMED model (Sharp-Hansen et al., 1990). The MULTIMED estimation method was selected to be consistent with that used by EPA’s Office of Solid Waste for the EPA Composite Model for Landfills (EPACML). The equation for estimating mixing zone depth (d) is as follows:

$$d = (2 \alpha_v L)^{0.5} + d_a \{1 - \exp[(-L I)/(V_s n_e d_a)]\} \quad (31)$$

where

- α_v = vertical dispersivity (m/m)
- V_s = horizontal seepage velocity (m/yr)
- n_e = effective aquifer porosity ($L_{\text{pore}}/L_{\text{aquifer}}$)
- d_a = aquifer depth (m).

The first term, $(2\alpha_v L)^{0.5}$, estimates the depth of mixing due to vertical dispersivity ($d_{\alpha v}$) along the length of ground water travel. Defining the point of compliance with ground water standards at the downgradient edge of the source, this travel distance becomes the length of the source parallel to flow L . Vertical dispersivity can be estimated by the following relationship (Gelhar and Axness, 1981):

$$\alpha_v = 0.056 \alpha_L \quad (32)$$

where

- α_L = longitudinal dispersivity = $0.1 x_r$
- x_r = horizontal distance to receptor (m).

Because the potential receptor is assumed to have a well at the edge of the facility, $x_r = L$ and

$$\alpha_v = 0.0056 L \quad (33)$$

Thus

$$d_{\alpha v} = (0.0112 L^2)^{0.5} \quad (34)$$

The second term, $d_a \{1 - \exp[(-LI) / (V_s n_e d_a)]\}$, estimates the depth of mixing due to the downward velocity of infiltrating water, d_{Iv} . In this equation, the following substitution may be made:

$$V_s = Ki/n_e \quad (35)$$

so

$$d_{Iv} = d_a \{1 - \exp[(-L I)/(K I d_a)]\} \quad (36)$$

Thus, mixing zone depth is calculated as follows:

$$d = d_{\alpha v} + d_{Iv} \quad (37)$$

Estimation of Mixing Zone Depth

$$d = (0.0112 L^2)^{0.5} + d_a \{1 - \exp[-L I / (K i d_a)]\} \quad (38)$$

Parameter/Definition (units)
d/mixing zone depth (m)
L/source length parallel to ground water flow (m)
I/infiltration rate (m/yr)
K/aquifer hydraulic conductivity (m/yr)
i/hydraulic gradient (m/m)
d _a /aquifer thickness (m)

Incorporation of this equation for mixing zone depth into the SSL dilution equation results in five parameters that must be estimated to calculate dilution: source length (L), infiltration rate (I), aquifer hydraulic conductivity (K), aquifer hydraulic gradient (i), and aquifer thickness (d_a). Aquifer thickness also serves as a limit for mixing zone depth. The User's Guide for Radionuclides (U.S. EPA, 1999a) describes how to develop site-specific estimates for these parameters. Parameter definitions and defaults used to develop generic SSLs are as follows:

- Source Length (L)** is the length of the source (i.e., area of contaminated soil) parallel to ground water flow and affects the flux of contaminant released in soil leachate (IL) as well as the depth of mixing in the aquifer. The default option for this parameter assumes a square, 0.5-acre contaminant source. This default was changed from 30 acres in response to comments to be more representative of actual contaminated soil sources (see Section 1.3.4). Increasing source area (and thereby area) may result in a lower dilution factor. Appendix A includes an analysis of the conservatism associated with the 0.5-acre source size.
- Infiltration Rate (I)**. Infiltration rate times the source area determines the amount of contaminant (in soil leachate) that enters the aquifer over time. Thus, increasing infiltration decreases the dilution factor. Two options can be used to generate infiltration rate estimates for SSL calculation. The first assumes that infiltration rate is equivalent to recharge. This is generally true for uncontrolled contaminated soil sites but would be conservative for capped sites (infiltration < recharge) and nonconservative for sites with an additional source of infiltration, such as surface impoundments (infiltration > recharge). Recharge estimates for this option can be obtained from Aller et al. (1987) by hydrogeologic setting, as described in Section 2.5.6. of the 1996 TBD (EPA 1996b). Recent additional information may also be found in "Estimation of Infiltration Rate in the Vadose Zone: Compilation of Simple Mathematical Models. Volume I" (U.S. EPA 1998c)

The second option is to use the HELP model to estimate infiltration, as was done for OSW's EPACML and EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) modeling efforts. The Soil Screening Guidance (U.S. EPA, 1995c) provides information on obtaining and using the HELP model to estimate site-specific infiltration rates.

- Aquifer Parameters.** Aquifer parameters needed for the dilution factor model include hydraulic conductivity (K, m/yr), hydraulic gradient (i, m/m), and aquifer thickness (d_a, m). The User's Guide (U.S. EPA, 1996a) describes how to develop aquifer parameter estimates for calculating a site-specific dilution factor.

2.6.5 Default Dilution-Attenuation Factor. EPA has selected a default DAF of 20 to account for contaminant dilution and attenuation during transport through the saturated zone to a compliance point (i.e., receptor well). This assumption was developed in Section 2.5.6 of the 1996 TBD (EPA 1996b). At most sites, this adjustment will more accurately reflect a radionuclide's threat to ground water resources than assuming a DAF of 1 (i.e., no

dilution or attenuation). EPA selected a DAF of 20 using a "weight of evidence" approach. This approach considers results from OSW's EPACMTP model as well as results from applying the SSL dilution model described in Section 2.6.4 of this report to 300 ground water sites across the country.

The default DAF of 20 represents an adjustment from the DAF of 10 presented in the December 1994 draft Soil Screening Guidance (U.S. EPA, 1994h) to reflect a change in default source size from 30 acres to 0.5 acre. A DAF of 20 is protective for sources up to 0.5 acre in size. Analyses presented in Appendix A indicate that it can be protective of larger sources as well. However, this hypothesis should be examined on a case-by-case basis before applying a DAF of 20 to sources larger than 0.5 acre.

2.7 Mass-Limit Model Development

This section describes the development of models to solve the mass-balance violations inherent in the infinite source (i.e., steady-state) models used to calculate SSLs for the migration to ground water exposure pathway. The model developed is not a finite source model per se, but is designed for use with the current infinite source (i.e., steady-state) model to provide a lower, mass-based limit for SSLs for the migration to ground water.

The mass-limit model calculates a soil concentration that corresponds to the release of all radionuclides present within the source, at a constant health-based concentration, over the duration of exposure. These concentration limits are used as a minimum concentration for each SSL; below this concentration, a receptor point concentration time-averaged over the exposure period cannot exceed the health-based concentration on which it is based.

2.7.1 Mass Balance Issues. Infinite source (i.e., steady-state) models are subject to mass balance violations under certain conditions. Depending on a compound's solubility and the size of the source, modeled leaching rates can result in a source being depleted in a shorter time than the exposure duration (or the flux over a 30- or 70-year duration would release a greater mass of radionuclides than are present). Several commenters to the December 1994 draft Soil Screening Guidance for chemicals expressed concern that it is unrealistic for total emissions over the duration of exposure to exceed the total mass of contaminants in a source.

In summary, the mass-limit approach offers the following advantages:

- It corrects the possible mass-balance violation in the infinite-source SSLs.
- It does not require development of a finite source model to calculate SSLs.
- It is appropriate for screening, being based on the conservative assumption that all of the radionuclide present leaches over the period of exposure.
- It is easy to develop and implement, requiring only very simple algebraic equations and input parameters that are, with the exception of source depth, already used to calculate SSLs.

The derivation of this model is described below. It should be noted that the American Industrial Health Council (AIHC) independently developed identical models to solve the mass-balance violation as part of their public comments on the Soil Screening Guidance.

2.7.2 Migration to Ground Water Mass-Limit Model. For the migration to ground water pathway, the mass of radionuclide leached from a contaminant source over a fixed exposure duration (ED) period can be calculated as

$$M_l = C_w * I * A_s * ED \quad (39)$$

where

- M_l = mass of contaminant leached (g)
- C_w = leachate contaminant concentration (mg/L or g/m³)
- I = infiltration rate (m/yr)
- A_s = source area (m²)
- ED = exposure duration (yr).

The total mass of contaminants present in a source can be expressed as

$$M_T = C_t * \rho_b * A_s * d_s \quad (40)$$

where

- M_T = total mass of contaminant present (g)
- C_t = total soil contaminant concentration (mg/kg or g/Mg, dry basis)
- ρ_b = dry soil bulk density (kg/L or Mg/m³)
- A_s = source area (m²)
- d_s = source depth (m).

To avoid a mass balance violation, the mass of contaminant leached cannot exceed the total mass of contaminants present (i.e., M_l cannot exceed M_T). Therefore, the maximum possible contaminant mass that can be leached from a source (assuming no volatilization or degradation) is M_T and the upper limit for M_l is

$$M_l = M_T$$

or

$$C_w * I * A_s * ED = C_t * \rho_b * A_s * d_s \quad (41)$$

Rearranging to solve for the total soil concentration (C_t) corresponding to this situation (i.e., maximum possible leaching), as equation 41 becomes:

Mass-Limit Model for Migration to Ground Water Pathway

$$C_t = \frac{C_w * I * ED}{\rho_b * d_s} \quad (42)$$

Parameter/Definition (units)	Default
C _s /screening level in soil (mg/kg)	--
C _w /target soil leachate concentration (mg/L)	(MCL or HBL) × 20 DAF
I/infiltration rate (m/yr)	site-specific
ED/exposure duration (yr)	70
ρ _b /dry soil bulk density (kg/L)	1.5
d _s /average source depth (m)	site-specific

This soil concentration (C_s) represents a lower limit for soil screening levels calculated for the migration to ground water pathway. It represents the soil concentration corresponding to complete release of soil contaminants over the ED time period at a constant soil leachate concentration (C_w). Below this C_s, the soil leachate concentration averaged over the ED time period cannot exceed C_w.

SSLs for radionuclides may also be calculated in traditional radiological units of pCi/g, as follows:

$$SSL = \frac{C_w * I * ED * 1 \times 10^{-3}}{\rho_b * d_s} \quad (43)$$

Parameter/Definition (units)	Default
SSL/screening level in soil (pCi/g)	--
C _w /target soil leachate concentration (pCi/L)	(MCL or HBL) × 20 DAF
I/infiltration rate (m/yr)	site-specific
ED/exposure duration (yr)	70
1x10 ⁻³ /conversion factor (kg/g)	--
ρ _b /dry soil bulk density (kg/L)	1.5
d _s /average source depth (m)	site-specific