110832

Bechtel

Oak Ridge Corporate Center
151 Lafayette Drive
P.O. Box 350
Oak Ridge, Tennessee 37831-0350

Facsimile: (615) 220-2100

Job No. 14501, FUSRAP Project
DOE Contract No. DE-AC05-91OR21949
Code: 7200 /WBS: 134

NOV 19 1993

Randy Rohrman
U.S. Environmental Protection Agency
Region VII
RCRA/Geology
726 Minnesota Avenue
Kansas City, KS 66101

Attention: Randy Rohrman

Subject: St. Louis Airport Site Fate and Transport of Radioisotopes

Dear Mr. Rohrman:

Attached are several references which review the subject of radionuclide fate and transport in the subsurface. I hope the articles will assist you in your evaluation of contaminant transport at SLAPS.

Sincerely,

Jon S. Novick

JSN/kg:LTR_003

Enclosure

Bechtel National, Inc.

ACTION REQ'D [ ] YES [ ] NO DUE DATE ______
RESPONSE TO CHRON NO. ______________________
☐ FFA ☐ Permit ☐ Milestone ☐ OcR ☐ CCN ☐ CAR ☐ Mid-Yr ☐ Yr-End ☐ Periodic Rpt

FATE AND

TRANSPORT OF

RADIOISOTOPES

TABLE OF CONTENTS

Title of Article

ORNL-TM-4481
UC-70-Waste Management

Contract No. W-7405-eng-26

CHEMICAL TECHNOLOGY DIVISION

GEOCHEMICAL BEHAVIOR OF LONG-LIVED RADIOACTIVE WASTES

Ferruccio Gera

JULY 1975

OAK RIDGE NATIONAL LABORATORY
Oak Ridge, Tennessee 37830
operated by
UNION CARBIDE CORPORATION
for the
ENERGY RESEARCH AND DEVELOPMENT ADMINISTRATION

It can be seen that a comparison based only on the number of LAI contained in waste and ores or minerals neglects some important aspects of the problem. In fact, the actual hazard is greatly dependent on a number of factors, such as transfer of the radionuclides from the solid to the liquid phase, dispersal and accumulation mechanisms, biologic availability, reconcentration in edible organisms, etc.

The following sections of the report review the available data concerned with the behavior of natural alpha-emitters and transuranium elements in geologic systems and in terrestrial and aquatic ecosystems. Resuspension mechanisms are discussed briefly in order to obtain some feeling for the probability that a significant fraction of the activity would become airborne. Hopefully, this discussion will lead to a better understanding of the similarities and differences between natural radioactive materials and radioactive wastes, and to a more realistic assessment of the possible environmental impact of the nuclear fission industry.

3. GEOCHEMICAL DATA

The long-lived parents of the three naturally occurring radioactive decay series are ^{232}Th, ^{235}U, and ^{238}U. A fourth series, the neptunium series, with mass numbers defined by the general formula (4n + 1), is not known in nature, and is shown with the other three in the Appendix.*

Uranium and thorium, as minor constituents of the lithosphere, are present in all geologic materials. Table 9 shows the average content of uranium and thorium in various rock types.

---

*^{237}Np and ^{239}Pu are formed in nature when atoms of ^{238}U capture neutrons which can be produced by the spontaneous fission of uranium, cosmic radiation, and interaction between alpha particles and light nuclei. The reactions are:

$$^{238}\mathrm{U}\ (n,\gamma)\ ^{239}\mathrm{U}\ (\beta^-)\ ^{239}\mathrm{Np}\ (\beta^-)\ ^{239}\mathrm{Pu} \text{ and } ^{238}\mathrm{U}\ (n,2n)\ ^{237}\mathrm{U}\ (\beta^-)\ ^{237}\mathrm{Np}.$$

However, the concentrations, especially for ^{237}Np, are so low that these nuclides can be measured only with great effort.

Table 9. Concentration of Uranium and Thorium in Various Rocks[a]

Type of Rock	Concentration (ppm) U	Th	Th/U	Reference No.[b]
IGNEOUS ROCKS				
Acid Rocks (mostly granites)				
North America	4	13	3.2	16
World	3.5	18	5.1	17
Intermediate Rocks				
North America	2.6	10	3.8	16
World	1.8	7	3.9	17
Basic Rocks				
Basalts	0.8	5	6.2	18
Basalts	0.8	3	3.7	17
Average for all igneous rocks	3	10	3.3	
SEDIMENTARY ROCKS				
Placers enriched in uranium- and thorium-bearing heavy minerals	2	60	30	19
Sandstones	0.5-1	2-5		
Shales, average	3.7	12	3.2	20
Grey-green shales, USA	3.2	13.1	4.1	20
Bentonites, USA	5.0	24	4.8	20
Bauxites	9.3	53.2	5.7	20
Residual clays	1.8	13	7.2	20
Shales, Russian platform	4.1	11	2.6	21
Bituminous and carbonaceous shales	50-80 up to ~1250[c]			
Limestones	1.3	1.1	0.9	18
Limestones, USA	2.2	1.1	0.5	20
Limestones, Russian platform	2.1	2.4	1.1	21
Phosphate rocks	30-100 up to ~650			
Lignites and coals	1-200 up to ~10,000[d]			

---

[a] Modification of table from Antoni Polanski, Geochemistry of Isotopes, TT61-31327 (English Transl.), published by the Scientific Publications Foreign Cooperation Center, Warsaw, Poland, 1965.

[b] Refer to references in text.

[c] Shales of St. Hippolyte, France.

[d] Kolm, Cambrian coal, Sweden.

Uranium and thorium usually begin their geochemical cycles together. They are closely associated in the same accessory minerals of igneous rocks such as zircon, sphene, allanite, monazite, or xenotime; however, the differences in geochemical behavior (particularly in the postmagmatic phases, and during weathering, transport, and sedimentation) are fairly significant.

From our point of view, the greatest interest lies in the behavior of uranium and thorium during the weathering of rocks and minerals. Most uranium- and thorium-containing minerals dispersed in igneous rocks are extremely resistant to chemical weathering.[15] Several minerals of pegmatitic and hydrothermal deposits, particularly those rich in thorium (such as thorite and thorianite), are similarly characterized by very low solubility. Most of the dispersal of the radioactive elements contained in these minerals is thus achieved through the physical processes of erosion and transport by running water. Weathering, transport, and sedimentation can lead to the accumulation of uranium- and thorium-bearing minerals and, in favorable circumstances, placer deposits can be formed. Famous examples are the monazite sands found on the Brazilian coast between Cabo Frio and Recife, and on the Travancore coast in India.

Many minerals containing radioactive elements, particularly the uranium minerals of hydrothermal deposits, eventually undergo chemical weathering. Secondary products are formed and can be found in the oxidation zones associated with uranium deposits, especially uranite and pitchblende. During weathering under oxidizing conditions, a fraction of uranium goes into solution as stable uranyl complexes, causing partial separation of the two elements.[22] Chemical weathering of geologic materials thus results in a relative enrichment of thorium in the residual fraction.

Once the two elements reach a sedimentation basin they will eventually precipitate to the bottom. Thorium, having reached the sedimentation basin primarily by association with suspended particles, is rapidly removed by sedimentation; its residence time* in sea water has been estimated

---

*The residence time, τ, of elements or radionuclides in environmental compartments can be defined as the average time spent in the compartment before removal. It is expressed as: $\tau = A/(dA/dt)$, where A is the total amount of the element in the compartment, and dA/dt is the amount removed per unit of time. A necessary assumption is that complete mixing of the element takes place in a time which is short when compared with τ.[25]

at about 300 to 350 years. Uranium, on the other hand, is characterized by a residence time in sea water on the order of 500,000 years.[23-25] Sackett et al.[26] have estimated a residence time of uranium in the world oceans of 200,000 years. Precipitation of uranium involves chemical and biological processes which are especially active in reducing environments where the soluble hexavalent form becomes unstable and is reduced to tetravalent uranium. Consequently, liquid hydrocarbons and bituminous sediments such as dark shales are usually rich in uranium.

Some Tertiary and Cretaceous lignites are also uraniferous. The most commonly accepted explanation for this is that uranium was adsorbed on the surfaces of lignite particles from uranium-rich circulating ground waters, the source of the uranium being located in nearby bodies of igneous rocks. A direct relationship has been observed between the uranium content in peat and in mosses living in the same environment, indicating that, at least in specific cases, the uranium accumulation could have taken place in the living plants.[27] Some coal deposits are significantly enriched in uranium; a famous example is kolm, a Cambrian coal from Sweden, which can contain more than 1% uranium. The alum-shales in which kolm is contained have an average uranium content between 50 and 100 ppm, but the uranium concentration is about 200 ppm in proximity to the kolm seams.[28]

Phosphate rocks of marine origin can also be markedly enriched in uranium, up to about 650 ppm in some cases. It is believed that the uranium is mainly syngenetic, although limited redistribution during diagenesis is quite possible. Uranium in phosphorites is dispersed in the phosphate phase and apparently has been enriched by coprecipitation.[29] At least two mechanisms seem to be responsible for the presence of uranium in phosphate minerals: (1) isomorphous replacement of U^{4+} for Ca^{2+} in apatite, and (2) adsorption of uranium on the surfaces of the minute phosphate crystals and the associated organic matter. According to Bliskovskiy and Smirnov,[30] the weathering of phosphorites can result in a further increase in the uranium concentration.

In summary, the geochemistry of uranium and thorium is quite similar in the magmatic phase; however, differences develop in postmagmatic stages, becoming even more significant when the rocks and minerals are exposed to exogenous processes. Consequently, high values of the Th/U ratio are observed in continental sediments, such as residual clays and bauxites. In general, any elution process, particularly if under oxidizing conditions, will lead to enrichment of thorium relative to uranium. Low values of the Th/U ratio are found in sediments accumulated by chemical processes (e.g., carbonates and evaporites). Finally, extremely low values of the Th/U ratio are found in certain sediments such as dark shales, and in some coals and phosphatic rocks.

The short-lived daughters of uranium and thorium are usually found associated with their parents in radioactive equilibrium. In specific circumstances some separation can take place, the net result is a relative geochemical individualization, especially for nuclides with fairly long half-lives or great environmental mobility. Obvious examples are ^{226}Ra and radon.

The interaction between ground water and uranium- and thorium-rich rocks is of particular interest, since this is similar to the hypothetical first step of radionuclide mobilization after waste containment failure. Radium-226 is the most hazardous nuclide, for exposure by ingestion, both in high-level radioactive waste aged more than 50,000 years and in the natural radioactive series.

In radioactive equilibrium, the Ra/U ratio is 3.6×10^{-7}; the specific activity of ^{226}Ra is $\sim$1 Ci/g. Radium is an alkaline-earth element, being located in the second group of the periodic system of elements, and barium is its closest geochemical analogue. Radium occurs in nature in a very dispersed state and no minerals are known.

The chemistry of ground waters is always controlled by the chemical composition of the enclosing rocks, and the content of radioactive substances is no exception. The enrichment of natural waters with radioactive elements, particularly uranium and radium, is dependent on the following processes:

(1) primary transfer from the rock into the water;
(2) stability in solution;
(3) separation of the element from solution.

3.1 Mobilization of Uranium and Radium

Uranium and thorium in rocks are usually bound in the crystal lattice of minerals; therefore, their transfer to ground water is controlled by the solubility of the minerals, which is influenced by water chemistry, amount of free oxygen, temperature, etc.[31] Radioactive decay transforms uranium and thorium atoms into atoms of different elements that might not be able to occupy the same position in the crystal lattice. Radium, in particular, can be leached out of the minerals in which it was generated, and thus accumulated in the interstitial water. Of course, the transfer of radium from the solid to the liquid phase is greatly dependent on the size of the rock particles since leaching is a surface-related process.

On the other hand, if the grain size is in the clay range, radium can be transferred efficiently to the surface of the particles, but release from the solid phase is impeded by the low permeability and by surface attraction phenomena.

Starik and Lazarev[32] have studied the effect of crushing monazite on the extraction of radium, thorium, and uranium. Monazite is practically insoluble, and leaching of the radioelements occurs by extraction from the pores and surfaces of the mineral grains. The extraction of radium and uranium increased with the degree of crushing up to about 3%, for an average particle size <0.07 mm. On the other hand, the extraction of thorium was not changed by crushing, despite a more than 100-fold increase of the specific surface.

In relatively permeable rocks, the transfer of radium from the rock grains to the mobile interstitial water can be imagined as divided into the following steps:

(1) Radioactive decay, through recoil and the creation of lattice instability, moves radium atoms to the surface of rock particles.

(2) Radium is leached out of the particle surfaces into the film of adsorbed water surrounding the rock grains.

(3) Radium diffuses through the adsorbed water and reaches mobile water.

The transfer from the particle surfaces to the adsorbed water is controlled by an adsorption equilibrium, and the transfer to mobile water is dependent on a diffusion equilibrium. Due to the slowness of diffusion processes, a long contact time between ground water and formation, realized in conditions of stagnant or slow-moving waters, promotes their enrichment in radium. Stagnant waters are usually highly mineralized. ~~High salinity further enhances the enrichment in radium~~, caused by the competition and exchange between the ions in solution and the ions adsorbed on the rock particle surfaces.[33,34]

The nature of the ions present in ground water is also important. The dissolution of uranium is appreciably influenced by cationic composition of the water only in carbonate rocks. On the basis of their effect on the dissolution of uranium, cations can be arranged in this order:

$$Mg^{2+} > Ca^{2+} > Na^{+} .$$

The effect of cationic composition of the water on leaching of radium is much stronger. In general, the greater the chemical affinity of an element to radium, the greater its effect on leaching. The sequence is:

$$Ba^{2+} > Pb^{2+} > Sr^{2+} > Ca^{2+} > K^{+} > Na^{+} .$$

The anionic composition of the water significantly affects the solubility of uranium, but has little effect on radium. Bicarbonate ions are the most efficient since they cause the formation of readily soluble uranyl complexes of the type $[UO_2(HCO_3)_3]^-$.

Additional factors that are important in the transfer of radioactive elements into ground water are: (1) water temperature, (2) pH, (3) gas content, (4) electrochemical properties.

The transfer of uranium and radium into solution is directly proportional to the water temperature.

The solubility of uranium and radium increases at low and high values of pH. Therefore, acidic and alkaline waters are usually enriched in radioactive elements.

The gases with most effect on the migration of radioactive elements are O_2 and CO_2. Free oxygen in ground water decreases with depth down to an "oxygen surface" that represents the lower boundary of the presence of free oxygen. Above the oxygen surface is the domain of oxidation processes, whereas reduction processes are prevalent at greater depth. The depth of the oxygen surface can vary from almost at the surface to more than 1 km.[31] Free oxygen produces a marked effect on the solubility of uranium.

Carbon dioxide increases the transfer of uranium into solution by the formation of soluble bicarbonates of the very stable uranyl ion. The enrichment with radium is also affected.

The redox potential (Eh) of ground water is an index of the intensity of oxidation processes, and is related to several of the preceding factors. Above the oxygen surface the value of Eh is directly proportional to the content of free oxygen, while for the deeper zone it is inversely proportional to the content of hydrogen sulfide. Obviously, the chemical weathering of uranium-bearing minerals and the enrichment of uranium in solution takes place to a much greater extent in waters characterized by high values of Eh. Radium, on the other hand, is affected much less by the redox potential and shows greater stability in solution in waters of low Eh.[35]

3.2 Removal of Uranium and Radium from Ground Water

Once uranium and radium have been transferred into solution they migrate with the water. The extent of migration is dependent upon their stability in solution, and on the intensity of various processes leading to their removal from solution. Any variation of the factors discussed above as affecting the mobilization of uranium and radium can induce precipitation. Decrease of temperature, loss of gas, neutralization of acidic or alkaline waters, or any other change in the chemistry of the water might cause separation of uranium and/or radium from solution.

The principal processes responsible for precipitation of uranium are:

(1) Hydrolysis and coagulation of uranium hydroxides (mostly in clays).

(2) Adsorption on the surface of clay particles and organic matter, responsible for the enrichment of uranium in clays, bituminous shales, peat, coal, etc.

(3) Breakdown of complex ions with formation of less soluble compounds. Among the various carbonates, coprecipitation with siderite is most efficient in the removal of uranium from solution.

(4) Reduction of uranium from the hexavalent to the tetravalent oxidation state.

(5) Formation of insoluble salts - vanadates, phosphates, arsenates, carbonates, and silicates.

As already mentioned, no radium minerals are known in nature; however, radium will follow its geochemical analogues. Therefore, precipitation of barium compounds from ground water will result in the removal of radium from solution. Radium follows calcium and magnesium to a lesser extent. Significant enrichment of radium also takes place in precipitates of iron and manganese, probably due to the colloidal nature of these compounds and the adsorption of radium on the surface of colloids. Adsorption and ion exchange are responsible for the enrichment of radium in peat, coal, and clays. Table 10 shows the concentration of radium in the waters of some Russian mineral springs and the enrichment in the spring deposits. Obviously, in favorable circumstances, significant accumulation and reconcentration of radium can take place.

3.3 Abundance of Natural Alpha-Emitters in the Hydrosphere

The complexity of the geochemical behavior of uranium and radium indicates that the concentrations of these elements in natural waters will vary within wide limits. In general, the content of radioactive elements in surface waters is lower than in ground waters due to the dilution by meteoric waters. A relationship exists between the climate and the uranium content in ground waters, with arid regions being characterized by significantly higher levels of uranium.[35] In specific closed basins in arid areas, the concentration of uranium can be as high

Table 10. Content of Radium in Water and Deposits of Some Russian Mineral Springs[a]

Spring	Nature of Deposit	Radium in Water (pg/liter)	Radium in Deposit (pg/g)	$\frac{\text{Radium in Deposit}}{\text{Radium in Water}}$
Malakovka	Ferruginous precipitate	7,000	153,000	22,000
Yamkun	Travertine	1,000	14,000	14,000
Zheleznovodsk	Travertine	21,000	4,600	220
	Ferruginous precipitate	15,000	60,000	4,000
Pyatigorsk	Travertine	300	1,800	6,000
	Manganic precipitate	300	410,000	1,350,000
Obershlem	Clayey precipitate	6,000	2,830,000	471,000
Vishnevogorsk	Peat	100	2,000	20,000
Baku	Ferruginous precipitate	90,000	7,800	87
Ukhta	Ferruginous precipitate	748,000	400,000	530

[a] Modification of table from A. N. Tokarev and A. V. Shcherbakov, Radiohydrogeology, Moscow (1956) AEC-tr-4100 (1960).

as ~4 X 10^{-2} g/liter.[31] The average concentrations of uranium and radium in sea water are 3 μg/liter and 0.1 pg/liter respectively. The volume of the oceans is 1.37 X 10^{9} km^{3}; therefore, it can be calculated that about 1.5 X 10^{8} g of ^{226}Ra and 4.5 X 10^{15} g of uranium are present in sea water. Table 11 shows the ranges and mean concentrations of radium and uranium in surface and various ground waters. All values of the Ra/U ratio higher than 3.6 X 10^{-7} indicate an enrichment of radium relative to uranium, while the opposite is true for values lower than equilibrium. In ground waters below the oxidation zone, the equilibrium is usually displaced toward radium.

A deficiency of radium exists in the oceans and, to a lesser extent, in lakes. This is undoubtedly related to the very low solubility and removal by sedimentation of ^{230}Th, which is the immediate precursor of ^{226}Ra in the radioactive series (see the Appendix). This interpretation is supported by the excess of radium relative to uranium that has been reported for recent deep-sea sediments.[36,37] When the concentrations of ^{230}Th in sea water and deep-sea sediments are analyzed, it is found that the deficiency of this nuclide in water is much more marked than for radium; conversely, there is an excess of ^{230}Th with respect to ^{226}Ra in the sediments. This situation can be explained when it is understood that ^{230}Th, with a half-life of 80,000 years, separates from sea water and accumulates in the sediments, where its decay produces ^{226}Ra, part of which is leached out of the sediments and returned to the water.[38,39]

It has been observed that disequilibrium usually exists between ^{238}U and ^{234}U.[24,40-42] In sea water the ^{234}U/^{238}U activity ratio is 1.14, indicating an excess of ^{234}U of 14%. This is due to the fact that ^{234}U is removed from weathering rocks more readily than ^{238}U. Electrons are lost during the decay from ^{238}U to ^{234}U and change to the more soluble +6 state is facilitated.[24]

Natural waters contain significantly less thorium than uranium. In sea water, which represents about 98% of the hydrosphere, the average thorium concentration has been estimated as falling between 2 X 10^{-8} and 5 X 10^{-8} g/liter.[25,35] The U/Th ratio is thus about 100, or a variation of about 300 with respect to the relative abundance of the two elements in the lithosphere. On the other hand, the Th/U ratio in sea bottom

Table 11. Content of Uranium and Radium in Natural Waters[a]

Type of Waters	Natural Conditions	Radium (g/liter)			Uranium (g/liter)			$\frac{Ra}{U}$
		min.	max.	mean	min.	max.	mean	
Surface Waters	Oceans and seas	8.0×10^{-14}	4.5×10^{-11}	1.0×10^{-13}	3.6×10^{-8}	5.0×10^{-6}	3.0×10^{-6}	3×10^{-8}
	Lakes	1.0×10^{-13}	8.0×10^{-12}	1.0×10^{-12}	2.0×10^{-7}	4.0×10^{-2}	8.0×10^{-6}	1×10^{-7}
	Rivers	1.0×10^{-13}	4.0×10^{-12}	2.0×10^{-13}	2.0×10^{-8}	5.0×10^{-5}	6.0×10^{-7}	3×10^{-7}
Waters of Sedimentary Rocks	Zone of intensive water circulation	1.0×10^{-13}	6.0×10^{-12}	2.0×10^{-12}	2.0×10^{-7}	8.0×10^{-6}	5.0×10^{-6}	5×10^{-7}
	Zone of highly impeded water circulation	1.0×10^{-11}	1.0×10^{-8}	3.0×10^{-10}	2.0×10^{-8}	6.0×10^{-6}	2.0×10^{-7}	1×10^{-3}
Waters of Magmatic Acid Rocks	Zone of intensive water circulation (waters of the weathering shell)	1.0×10^{-12}	7.0×10^{-12}	2.0×10^{-12}	2.0×10^{-7}	3.0×10^{-5}	7.0×10^{-6}	1×10^{-6}
	Zone of impeded water circulation (waters of deep tectonic fissures)	2.0×10^{-12}	9.0×10^{-12}	4.0×10^{-12}	2.0×10^{-7}	8.0×10^{-6}	4.0×10^{-6}	2×10^{-6}
Waters of Uranium Deposits	Zone of intensive water circulation (waters of the oxidation zone)	8.0×10^{-12}	2.0×10^{-9}	8.0×10^{-11}	5.0×10^{-5}	9.0×10^{-2}	6.0×10^{-4}	1×10^{-7}
	Zone of impeded water circulation (waters of the reduction zone)	1.0×10^{-11}	8.0×10^{-10}	6.0×10^{-11}	2.0×10^{-6}	3.0×10^{-5}	8.0×10^{-6}	1×10^{-5}

[a] Modification of table from A. N. Tokarev and A. V. Shcherbakov, Radiohydrogeology, Moscow (1956) AEC-tr-4100 (1960).

sediments is roughly similar to the average value in continental rocks.[22,43] This would indicate that, overall, the two elements are transferred to the oceans with similar efficiency. However, while the geochemistry of uranium is based mainly on aqueous solutions, the geochemical cycle of thorium is almost completely independent of aqueous solutions.

Sackett et al.[26] have recently attempted a world balance of the geochemical cycle of uranium, and have found that the input of uranium to the ocean is higher than the amount removed by deposition by at least a factor of 3. This discrepancy could be explained by the presently high input of uranium to the ocean, or by an underestimate of uranium removal. The actions of man, particularly in the use of phosphate fertilizers and the enhanced leaching of soils caused by world-wide cultivation, are probably responsible for the observed imbalance.

A short-lived nuclide in the ^{235}U series, ^{227}Th, is present in sea water at a concentration of only a few percent of the equilibrium value. This indicates that one of the longer-lived precursors, most likely ^{231}Pa (half-life, 32,500 years), is efficiently removed from sea water.[37]

Of all natural radioactive elements present in natural waters, radium is usually the most significant from the radiological point of view.[44,45] The limit of ^{226}Ra in drinking water supplies has been fixed by the U. S. Public Health Service at 3 pCi/liter.[46] Many natural waters, not only highly mineralized brines of sedimentary rocks and waters of specific mineral springs, but also waters of aquifers used extensively for drinking water supplies, contain radium in excess of this limit.

High levels of radium in ground waters are known in several areas of the United States.[47-49] Three important examples of radium-rich aquifers are (1) limestones, dolomites, and sandstones of Cambrian and Ordovician age in the upper Mississippi Valley (Illinois, Iowa, Wisconsin, Michigan, and Indiana); (2) limestones and sandstones of Ordovician age in Kansas, Oklahoma, and Missouri; and (3) the Cheyenne sandstone member of the Purgatoire Formation of Early Cretaceous age in southeastern Colorado and northeastern New Mexico. Scott[47] says that in the first two aquifers most of the uranium is probably associated with shale beds and lenses that are fairly numerous throughout the formation, whereas it is likely that uraniferous minerals are uniformly dispersed in the Cheyenne sandstone.

A correlation exists between radium concentration and salinity; fresh ground waters contain ^{226}Ra at concentrations ranging from fractions of a picocurie per liter to a few tens of picocuries per liter, while brines (dissolved solids >3000 ppm) can contain as much as several hundreds of picocuries per liter.

Samuels[50] estimates that in the upper Mississippi Valley at least one million people are exposed to drinking waters containing more than 3 picocuries of ^{226}Ra per liter, and about 50,000 people have more than 10 picocuries per liter in their water supplies. This obviously affects the radium intake in the exposed populations. The average daily intake of ^{226}Ra for the U. S. population is probably in the range 1 to 2 pCi,[51-53] of which about 90% is from food and 10% is from drinking water.[54] In the high radium areas, the daily intake can be significantly higher. Samuels[50] estimates that the contributions to the intake from food and water are probably reversed, with about 90% due to ingestion of water.

3.4 Transuranium Elements

As already mentioned, minute amounts of neptunium and plutonium are formed in nature by the interaction of neutrons with ^{238}U nuclei. Unfortunately, very little is known about the geochemistry of these elements and their long-term mobility through geologic systems.

Several authors have reported the occurrence of ^{239}Pu in uranium ores with α, which is defined as the $^{239}Pu/^{238}U$ ratio, on the order of 10^{-11} to 10^{-12}.[55-57] These values are in fairly good agreement with the α obtained theoretically, assuming complete utilization of available neutrons.

Cherdyntsev et al.[58] have measured the ^{239}Pu content in some volcanic waters and found values up to 10^{-13} g/liter with an α as high as 10^{-7}. This is orders of magnitude higher than the α observed in uranium minerals. Excluding the possibility of contamination by artificial plutonium, Cherdyntsev et al. propose an additional natural source of ^{239}Pu in the form of an unknown osmium-type transuranium element with probable atomic number 108.[58,59] Among the plutonium-bearing minerals associated with the

7. REFERENCES

1. F. Gera and D. G. Jacobs, *Considerations in the Long-Term Management of High-Level Radioactive Wastes*, ORNL-4762 (1972).

2. Staff of the Oak Ridge National Laboratory, *Siting of Fuel Reprocessing Plants and Waste Management Facilities*, ORNL-4451 (1970).

3. M. J. Bell and R. S. Dillon, *The Long-Term Hazard of Radioactive Wastes Produced by the Enriched Uranium, Pu-^{238}U, and ^{233}U-Th Fuel Cycles*, ORNL-TM-3548 (1971).

4. H. C. Claiborne, *Neutron-Induced Transmutation of High-Level Radioactive Waste*, ORNL-TM-3964 (1972).

5. M. J. Bell, *ORIGEN - The ORNL Isotope Generation and Depletion Code*, ORNL-4628 (1973).

6. F. Gera and D. G. Jacobs, "Hazard Potential of Radioactive Waste," *Radioecology Applied to the Protection of Man and His Environment, Rome, Sept. 7-10, 1971*, EUR-4800, pp. 891-907 (1972).

1C9001 I5 7. International Atomic Energy Agency, *Basic Safety Standards for Radiation Protection*, Safety Series No. 9, Vienna, 1967.

8. F. Duhamel and J. M. Lavie, "Comment établir des règles pratiques pour éviter la contamination," pp. 317-36 in *Health Physics in Nuclear Installations*, ENEA, 1959.

9. K. Z. Morgan, W. S. Snyder, and M. R. Ford, "Relative Hazard of the Various Radioactive Materials," Health Phys. *10*, 151-69 (1964).

10. R. A. Hilliar, *Relative Inhalation Hazards from Nuclides Present in Fresh Irradiated Fuels*, RD/B/N - 1143 (1968).

11. S. V. Kaye, R. S. Booth, P. S. Rohwer, and E. G. Struxness, "Ecological Model Development for a Methodology to Estimate Doses to Human Populations," *Radioecology Applied to the Protection of Man and His Environment, Rome, Sept. 7-10, 1971*, EUR-4800, pp. 909-20 (1972).

12. A. S. Kubo and D. J. Rose, "Disposal of Nuclear Wastes," Science *182*, 1205-11 (1973).

13. F. X. Roser and T. L. Cullen, "External Radiation Levels in High-Background Regions of Brazil," pp. 825-36 in *The Natural Radiation Environment*, J. A. S. Adams and W. M. Lowder, Eds., Univ. of Chicago Press, Chicago, 1964.

14. P. J. Magno, P. E. Kauffman, and B. Shleien, "Plutonium in Environmental and Biological Media," Health Phys. *13*, 1325-30 (1967).

15. Antoni Polański, Geochemistry of Isotopes, TT61-31327 (English Transl.), Scientific Publications Foreign Cooperation Center, Warsaw, Poland, 1965.

16. F. E. Senftle and N. B. Keevil, "Thorium-Uranium Ratios in the Theory of Genesis of Lead Ores," Trans. Am. Geophysical Union 28, 732-38 (1947).

17. A. P. Vinogradov, "Regularity of Distribution of Chemical Elements in the Earth's Crust," Geochemistry (USSR) (English Transl.) (1), 1-43 (1956).

18. R. D. Evans and C. Goodman, "Radioactivity of Rocks," Bull. Geol. Soc. Am. 52, 459-90 (1941).

19. K. G. Bell, "Uranium and Thorium in Sedimentary Rocks," pp. 98-114 in Nuclear Geology, H. Faul, Ed., Wiley, New York, 1954.

20. J. A. S. Adams and Ch. E. Weaver, "Thorium to Uranium Ratios as Indicators of Sedimentary Processes," Bull. Am. Ass. Petrol. Geologists 42, 387-430 (1958).

21. V. I. Baranov, A. B. Ronov, and K. G. Kunashova, "Geochemistry of Dispersed Thorium and Uranium in Clays and Carbonate Rocks of the Russian Platform," Geochemistry (USSR) (English Transl.) (3), 227-35 (1956).

22. J. A. S. Adams, J. K. Osmond, and J. J. W. Rogers, "The Geochemistry of Thorium and Uranium," pp. 298-348 in Physics and Chemistry of the Earth, L. H. Ahrens, F. Press, K. Rankama, and S. K. Runcorn, Eds., Vol. 3, Pergamon, New York, 1959.

23. F. F. Koczy, "Remarks on Age Determination in Deep-Sea Sediments," pp. 155-71 in Progress in Oceanography, M. Sears, Ed., Pergamon, Oxford, 1965.

24. M. Koide and E. D. Goldberg, "Uranium-234/Uranium-238 Ratios in Sea Water," pp. 173-77 in Progress in Oceanography, M. Sears, Ed., Pergamon, Oxford, 1965.

25. E. D. Goldberg, "Minor Elements in Sea Water," pp. 163-96 in Chemical Oceanography, J. P. Riley and G. Skirrow, Eds., Vol. 1, Academic, New York, 1965.

26. W. M. Sackett, T. Mo, R. F. Spalding, and M. E. Exner, "A Revaluation of the Marine Geochemistry of Uranium," Symposium on Radioactive Contamination of the Marine Environment, Seattle, July 10-14, 1972, IAEA, pp. 757-69, 1973.

27. A. P. Lopatkina, V. S. Komarov, A. N. Sergeyev, and A. G. Andreyev, "On Concentration of Uranium by Living and Dead Peat-Forming Plants," Geochem. Int. 7 (2), 277-82 (1970).

28. K. Rankama and Th. G. Sahama, Geochemistry, Univ. of Chicago Press, Chicago, 1950.

29. E. Wm. Heinrich, Mineralogy and Geology of Radioactive Raw Materials, McGraw-Hill, New York, 1958.

30. V. Z. Bliskovskiy and A. I. Smirnov, "Radioactivity of Phosphorites," Geochem. Int. 3, 563-67 (1966).

31. A. N. Tokarev and A. V. Shcherbakov, Radiohydrogeology, Moscow, 1956; AEC-tr-4100 (1960).

32. I. E. Starik and K. F. Lazarev, "Effect of Crushing of Minerals on the Extraction of Radioactive Elements," Radiokhimiya 2 (6), 749-52 (1960).

33. E. Mazor, "Radon and Radium Content of Some Israeli Water Sources and a Hypothesis on Underground Reservoirs of Brines, Oils and Gases in the Rift Valley," Geochim. et Cosmochim. Acta 26, 765-86 (1962).

34. A. B. Tanner, "Physical and Chemical Controls on Distribution of Radium-226 and Radon-222 in Ground Water Near Great Salt Lake, Utah," pp. 253-76 in The Natural Radiation Environment, J. A. S. Adams and W. M. Lowder, Eds., Univ. of Chicago Press, Chicago, 1964.

35. A. I. Germanov, S. G. Batulin, G. A. Volkov, A. K. Lisitsin, and V. S. Serebrennikov, "Some Regularities of Uranium Distribution in Underground Waters," Proc. U. N. Intern. Conf. Peaceful Uses At. Energy, 2nd, Geneva, 1958, 2, 161-77 (1958).

36. F. F. Koczy, E. Picciotto, G. Poulaert, and S. Wilgain, "Mesure des isotopes du thorium dans l'eau de mer," Geochim. et Cosmochim. Acta 11, 103-29 (1957).

37. J. D. Burton, "Radioactive Nuclides in Sea Water, Marine Sediments, and Marine Organisms," pp. 425-75 in Chemical Oceanography, J. P. Riley and G. Skirrow, Eds., Vol. 2, Academic, New York, 1965.

38. F. F. Koczy, "Geochemical Balance in the Hydrosphere," pp. 120-27 in Nuclear Geology, H. Faul, Ed., Wiley, New York, 1954.

39. F. F. Koczy, "Natural Radium as a Tracer in the Ocean," Proc. U. N. Intern. Conf. Peaceful Uses At. Energy, 2nd, Geneva, 1958, 18, 351-57 (1958).

40. V. V. Cherdyntsev and N. I. Senina, "Radioactivity of Young Volcanic Rocks of Pacific Islands," Geochem. Int. 5, 355-61 (1968).

41. D. L. Thurber, "Anomalous $^{234}U/^{238}U$ in Nature," J. Geophys. Res. 67 (11), 4518-20 (1962).

42. D. L. Thurber, "Natural Variations in the Ratio of ^{234}U to ^{238}U," in Symposium on Radioactive Dating, Athens, 19-23 Nov. 1962, IAEA, 1963.

43. E. Picciotto and S. Wilgain, "Thorium Determination in Deep-Sea Sediments," Nature 173, 632-33 (1954).

44. M. Eisenbud, H. Petrow, R. T. Drew, F. X. Roser, G. Kegel and T. L. Cullen, "Naturally Occurring Radionuclides in Foods and Waters from the Brazilian Areas of High Radioactivity, pp. 837-54 in The Natural Radiation Environment, J. A. S. Adams and W. M. Lowder, Eds., Univ. of Chicago Press, Chicago, 1964.

45. Ernest Marsden, "Radioactivity of Some Rocks, Soils, Plants, and Bones," pp. 807-24 in The Natural Radiation Environment, J. A. S. Adams and W. M. Lowder, Eds., Univ. of Chicago Press, Chicago, 1964.

46. Public Health Service, "Drinking Water Standards, 1961," J. Am. Water Works Assoc. 53, 935-45 (1961).

47. R. C. Scott, "Radium in Natural Waters in the United States," First National Symposium on Radioecology, Fort Collins, Colo. Sept. 10-15, 1961, pp. 237-49, 1963.

48. R. C. Scott and F. B. Barker, "Radium and Uranium in Ground Water of the United States," Proc. U. N. Intern. Conf. Peaceful Uses of At. Energy, 2nd, Geneva, 1958, 2, 153-57 (1958).

49. R. C. Scott and F. B. Barker, Data on Uranium and Radium in Ground Water, United States, 1954-1957, U. S. Geological Survey Prof. Paper 426, 1962.

50. L. D. Samuels, "A Study of Environmental Exposure to Radium in Drinking Water," pp. 239-51, in The Natural Radiation Environment, J. A. S. Adams and W. M. Lowder, Eds., Univ. of Chicago Press, Chicago, 1964.

51. I. Michelson, J. C. Thompson, Jr., B. W. Hess, and C. L. Comar, "Radioactivity in Total Diet," J. Nutri. 78, 371-83 (1962).

52. I. M. Fisenne and H. W. Keller, Radium-226 in the Diet of Three United States Cities, HASL-207, pp. 1.2-4 (1969).

53. I. M. Fisenne and H. W. Keller, Radium-226 in the Diet of Two U. S. Cities, HASL-224, pp. 1.2-8 (1970).

54. H. Muth, B. Rajewsky, H. J. Hantke and K. Aurand, "The Normal Radium Content and the ^{226}Ra/Ca Ratio of Various Foods, Drinking Water and Different Organs and Tissues of the Human Body," Health Phys. 2, 239-45 (1960).

55. D. F. Peppard, M. H. Studier, M. V. Gergel, G. W. Mason, J. C. Sullivan, and J. F. Mech, "Isolation of Microgram Quantities of Naturally-Occurring Plutonium and Examination of Its Isotopic Composition," J. Amer. Chem. Soc. 73, 2529-31 (1951).

56. C. A. Levine and G. T. Seaborg, "The Occurrence of Plutonium in Nature," J. Am. Chem. Soc. 73, 3278-83 (1951).

57. W. D. Davis and J. L. Mcwherter, "Abundance of U-236 in Nature," Geochimica et Cosmochim. Acta 26, 681-84 (1962).

58. V. V. Cherdyntsev, I. V. Kazachevskiy, L. D. Sulerzhitskiy, and Ye. A. Kuzimina, "Plutonium-239 in Nature," Geochem. Int. 2, 918-20 (1965).

59. V. V. Cherdyntsev and V. F. Mikhailov, "A Primordial Transuranium Isotope in Nature," Geochem. (USSR) (1), 1-13 (1963).

60. V. V. Cherdyntsev, V. L. Zverev, V. M. Kuptsov, and G. I. Kislitsina, "Plutonium-239 in Nature," Geochem. Int. 5, 355-61 (1968).

61. V. M. Kupstov and V. V. Cherdyntsev, "The Decay Products of Uranium and Thorium in Active Volcanism in the USSR," Geochem. Int. 6 (3), 532-45 (1969).

62. V. V. Cherdyntsev, and N. I. Senina, "Radioactivity of Young Volcanic Rocks of Pacific Islands," Geochem. Int. 7 (4), 652-63 (1970).

63. J. B. Andelman and T. C. Rozzell, "Plutonium in the Water Environment. Characteristics of Aqueous Plutonium," pp. 118-137 in Radionuclides in the Environment, S. C. Freiling, Ed., American Chemical Society, Washington, D. C., 1970.

64. G. L. Silver, Plutonium in Natural Waters, MLM-1870 (1971).

65. C. W. Christenson and R. G. Thomas, "Movement of Plutonium Through Los Alamos Tuff," Second Conference on Ground Disposal of Radioactive Wastes, Chalk River, Canada, 26-29 Sept. 1961, TID-7628, pp. 248-81 (1962).

66. M. C. De Bortoli and P. Gaglione, "Natural and Fallout Radioactivity in the Soil," Health Phys. 17, 701-10 (1969).

67. P. W. Krey and E. P. Hardy, Plutonium in Soil Around the Rock Flats Plant, HASL-235 (1970).

68. E. P. Hardy and P. W. Krey, "Determining the Accumulated Deposit of Radionuclides by Soil Sampling and Analysis," *Symposium on Environmental Plutonium*, LA-4756, pp. 37-42 (1971).

69. E. P. Hardy, Jr., P. W. Krey, and H. L. Volchok, *Global Inventory and Distribution of Pu-238 from SNAP-9A*, HASL-250 (1972).

70. E. M. Romney, H. M. Mork, and K. H. Larson, "Persistence of Plutonium in Soil, Plants and Small Mammals," Health Phys. *19*, 487-91 (1970).

71. J. H. Olafson, H. Nishita, and K. H. Larson, *The Distribution of Plutonium in the Soils of Central and Northwestern New Mexico as a Result of the Atomic Bomb Test of July 16, 1945*, UCLA-406 (1957).

72. B. F. Hajek, *Plutonium and Americium Mobility in Soils*, BNWL-CC-925 (1966).

73. W. A. Thomas and D. G. Jacobs, "Curium Behavior in Plants and Soils," Soil Sci. *108*, 305-7 (1969).

74. K. R. Price, *Uptake of ^{237}Np, ^{239}Pu, ^{241}Am, and ^{244}Cm from Soil by Tumbleweed and Cheatgrass*, BNWL-1688 (1972).

75. K. C. Pillai, R. C. Smith and T. R. Folsom, "Plutonium in the Marine Environment," Nature *203*, 568-71 (1964).

76. K. M. Wong, J. C. Burke, and V. T. Bowen, "Plutonium Concentration in Organisms of the Atlantic Ocean," *Symposium on Health Physics Aspects of Nuclear Facility Siting, Idaho Falls, Idaho, November 3-6, 1970*, pp. 529-39, 1970.

77. K. M. Wong, V. E. Noshkin, L. Surprenant, and V. T. Bowen, *Plutonium-239 in Some Marine Organisms and Sediments*, NYO-2174-115 (1970).

78. V. T. Bowen, K. M. Wong, and V. E. Noshkin, "Plutonium-239 In and Over the Atlantic Ocean," J. of Marine Res. *29* (1), 1-10 (1971).

79. V. E. Noshkin, "Ecological Aspects of Plutonium Dissemination in Aquatic Environments," Health Phys. *22*, 537-49 (1972).

80. V. E. Noshkin and V. T. Bowen, "Concentrations and Distributions of Long-Lived Fallout Radionuclides in Open Ocean Sediments," *Symposium on Radioactive Contamination of the Marine Environment, Seattle, 10-14 July 1972*, IAEA, pp. 671-86, 1973.

81. N. G. Stewart, R. N. Crooks, and E. M. R. Fisher, *The Radiological Dose to Persons in the U. K. Due to Debris from Nuclear Test Explosions Prior to January 1956*, AERE HP/R 2017 (1956).

82. Y. Miyake, Y. Katsuragi, and Y. Sugimura, "A Study on Plutonium Fallout," J. Geophys. Res. *75*, 2329-30 (1970).

83. A. Aarkrog, "Radiological Investigations of Plutonium in an Aquatic Marine Environment," Health Phys. <u>20</u>, 31-47 (1971).

84. E. P. Hardy, P. W. Krey, and H. L. Volchok, "Global Inventory and Distribution of Fallout Plutonium," Nature <u>241</u>, 444-45 (1972).

85. H. L. Volchok and P. W. Krey, "Implications of Change in the Global Strontium-90 Budget," J. Geophys. Res. <u>72</u>, 5155-56 (1967).

86. H. L. Volchok and P. W. Krey, <u>Plutonium-239 Anomaly in the Troposphere</u>, HASL-224, pp. I.14-27 (1970).

87. J. H. Harley, "Worldwide Plutonium Fallout from Weapons Tests," <u>Symposium on Environmental Plutonium</u>, LA-4756, pp. 13-19 (1971).

88. B. C. Bennett, "Global ^{90}Sr Fallout and Its Occurrences in Diet and Man," <u>Symposium on Biomedical Implications of Radiostrontium Exposure, Davis, Calif., Feb. 22-24, 1971</u>, pp. 17-30, 1972.

89. M. A. Wahlgren and D. M. Nelson, <u>Plutonium in Lake Michigan Water</u>, ANL-7960, pp. 7-14 (1972).

90. V. F. Hodge, T. R. Folsom, and D. R. Young, "Retention of Fallout Constituents in Upper Layers of the Pacific Ocean as Estimated from Studies of a Tuna Population," <u>Symposium on Radioactive Contamination of the Marine Environment, Seattle, 10-14 July 1972</u>, IAEA, pp. 263-76, 1973.

91. W. L. Templeton and A. Preston, "Transport and Distribution of Radioactive Effluents in Coastal and Estuarine Waters of the United Kingdom," <u>Symposium on the Disposal of Radioactive Wastes into Seas, Oceans and Surface Waters, Vienna, 16-20 May 1966</u>, IAEA, pp. 267-89, 1966.

92. A. D. Welander, "Distribution of Radionuclides in the Environment of Eniwetok and Bikini Atolls, August 1964," <u>Symposium on Radioecology</u>, D. J. Nelson and F. C. Evans, Eds., CONF-670503, pp. 346-54 (1969).

93. G. A. Welford and R. Baird, "Uranium Levels in Human Diet and Biological Materials," Health Phys. <u>13</u>, 1321-24 (1967).

94. R. S. Morse and G. A. Welford, "Dietary Intake of ^{210}Pb," Health Phys. <u>21</u>, 53-55 (1971).

95. M. De Bortoli and P. Gaglione, "Radium-226 in Environmental Materials and Foods," Health Phys. <u>22</u>, 43-48 (1972).

96. E. I. Hamilton, "The Concentration of Uranium in Man and His Diet," Health Phys. <u>22</u>, 149-53 (1972).

97. A. W. Klement, Jr., C. R. Miller, R. P. Minx, and B. Shleien, Estimates of Ionizing Radiation Doses in the United States 1600-2000, U.S.E.P.A., ORP-OSD 72-1 (1972).

98. K. A. Smith and P. G. Watson, Radium-226 in Diet in the United Kingdom in 1963, ARCRL 12 (1964).

99. United Nations Scientific Committee, Ionizing Radiation: Levels and Effects, Vol. 1, United Nations, New York, 1972.

100. R. C. Turner, J. M. Radley and W. V. Mayneord, "The Naturally Occurring α-Ray Activity of Foods," Health Phys. 1, 268-75 (1958).

101. L. Jacobson and R. Overstreet, "The Uptake by Plants of Plutonium and Some Products of Nuclear Fission Adsorbed on Soil Colloids," Soil Sci. 65, 129-34 (1948).

102. D. O. Wilson and J. F. Cline, "Removal of Plutonium-239, Tungsten-185 and Lead-210 from Soils," Nature 209, 941-42 (1966).

103. J. F. Cline, Uptake of ^{241}Am and ^{239}Pu by Plants, BNWL-714, pp. 8.24-25 (1968).

104. R. Scott Russell and K. A. Smith, "Naturally Occurring Radioactive Substances: The Uranium and Thorium Series," in Radioactivity and Human Diet, R. Scott Russell, Ed., Pergamon, Oxford, 1966.

105. A. P. Vinogradov, The Geochemistry of Rare and Dispersed Chemical Elements in Soils, Second Edition, Translated from Russian, Consultants Bureau Inc., New York, 1959.

106. A. W. Klement, Jr., "Natural Radionuclides in Food and Food Source Materials," pp. 113-55 in Radioactive Fallout, Soils, Plants, Foods, Man, E. B. Fowler, Ed., Elsevier, Amsterdam, 1965.

107. P. Newbould and E. R. Mercer, Absorption of Plutonium-239 by Plants, ARCRL-8, pp. 81-82 (1962).

108. P. Newbould, Absorption of Plutonium-239 by Plants, ARCRL-10 (1963).

109. K. R. Price, A Critical Review of Biological Accumulation, Discrimination and Uptake of Radionuclides Important to Waste Management Practices, 1943-1971, BNWL-B-148 (1971).

110. K. R. Price, Tumbleweed and Cheatgrass Uptake of Transuranium Elements Applied to Soil as Organic Acid Complexes, BNWL-1755 (1973).

111. Y. C. Ng, C. A. Burton, S. E. Thompson, R. K. Tandy, H. K. Kretner, and M. W. Pratt, Prediction of the Maximum Dosage to Man from the Fallout of Nuclear Devices. IV. Handbook for Estimating the Maximum Internal Dose from Radionuclides Released to the Biosphere, UCRL-50163, Part IV (1968).

112. R. O. McClellan, H. W. Casey, and L. K. Bustad, "Transfer of Some Transuranic Elements to Milk," Health Phys. 8, 689-94 (1962).

113. E. M. Romney and J. J. Davis, "Ecological Aspects of Plutonium Dissemination in Terrestrial Environments," Health Phys. 22, 551-57 (1972).

114. S. E. Thompson, C. A. Burton, D. J. Quinn, and Y. C. Ng, Concentration Factors of Chemical Elements in Edible Aquatic Organisms, UCRL-50564, Rev. 1 (1972).

115. R. F. Foster, "Environmental Behavior of Chromium and Neptunium," First National Symposium on Radioecology, Fort Collins, Colo. Sept. 10-15, 1961, V. Schultz and A. W. Klement, Jr., Eds., pp. 569-76, 1963.

116. J. J. Davis, R. W. Perkins, R. F. Palmer, W. C. Hanson, and J. F. Cline, "Radioactive Materials in Aquatic and Terrestrial Organisms Exposed to Reactor Effluent Water," Proc. U. N. Intern. Conf. Peaceful Uses At. Energy, 2nd, Geneva, 1958, 18, 423-28 (1958).

117. E. C. Tsivoglou, S. D. Shearer, R. M. Shaw, Jr., J. D. Jones, J. B. Anderson, C. E. Sponagle, and D. A. Clark, "Survey of Interstate Pollution of the Animas River (Colorado - New Mexico)," R. A. Taft Sanitary Engineering Center, Cincinnati, U.S.P.H.S. Report, 1959.

118. A. H. W. Aten, Jr., "Permissible Concentrations of Radionuclides in Sea Water," Health Phys. 6, 114-25 (1961).

119. V. E. Noshkin, V. T. Bowen, K. M. Wong, and J. C. Burke, "Plutonium in North Atlantic Ocean Organisms; Ecological Relationships," Third National Symposium on Radioecology, Oak Ridge, May 10-12, 1971, D. J. Nelson, Ed., CONF-710501, pp. 681-88 (1971).

120. K. M. Wong, V. F. Hodge, and T. R. Folsom, "Concentrations of Plutonium, Cobalt, and Silver Radionuclides in Selected Pacific Seaweeds," Symposium on Environmental Plutonium, LA-4756, pp. 93-98 (1971).

121. R. A. Horne, Marine Chemistry, Wiley-Interscience, New York, 1969.

122. P. E. Wilkniss and D. J. Bressan, "Use of Radionuclides to Evaluate the Transport of Pollutants from the Hydrosphere into the Atmosphere," Symposium on Nuclear Techniques in Environmental Pollution, Salzburg, 26-30 October, 1970, pp. 297-307, IAEA, 1971.

123. H. T. Ellis and R. F. Pueschel, "Solar Radiation: Absence of Air Pollution Trends at Mauna Loa," Science 172, 845-46 (1971).

124. G. M. Hidy and J. R. Brock, "An Assessment of the Global Sources of Tropospheric Aerosols," Second International Clean Air Congress, Washington, D. C., December 6-11, 1970, H. M. Englund and W. T. Beery, Eds., pp. 1088-97, Academic, New York, 1971.

125. E. Robinson and R. C. Robbins, "Emissions, Concentrations, and Fate of Gaseous Atmospheric Pollutants," pp. 1-93 in Air Pollution Control, W. Strauss, Ed., Part II, Wiley-Interscience, New York, 1972.

126. R. A. Bagnold, The Physics of Blown Sand and Desert Dunes, Methuen, London, 1960.

127. W. S. Chepil and N. P. Woodruff, "The Physics of Wind Erosion and Its Control," Advances in Agronomy, A. G. Norman, Ed., Vol. 15, pp. 211-302, Academic, New York, 1963.

128. Committee on Sedimentation, "Sediment Transportation Mechanics: Wind Erosion and Transportation," J. Hydraulics Div. Am. Soc. Civil Engrs. 91, 267-87 (1965).

129. W. S. Chepil, "Relation of Wind Erosion to Water-Stable and Dry Clod Structure of Soil," Soil Sci. 55, 275-87 (1943).

130. E. Eriksson, "The Yearly Circulation of Chloride and Sulfur in Nature: Meteorological, Geochemical, and Pedological Implications, Part I," Tellus 11, 375-403 (1959).

131. E. Eriksson, "The Yearly Circulation of Chloride and Sulfur in Nature: Meteorological, Geochemical, and Pedological Implications, Part II," Tellus 12, 63-109 (1960).

132. C. Junge, Air Chemistry and Radioactivity, Chap. 2, Academic, New York, 1963.

133. S. Judson, "Erosion of the Land - or What's Happening to Our Continents?" Am. Scientist 56, 356-74 (1968).

134. N. Bhandari, J. R. Arnold, and D. R. Parkin, "Cosmic Dust in the Stratosphere," J. Geophys. Res. 73, 1837-45 (1968).

135. F. Went, "Organic Matter in the Atmosphere and Its Possible Relation to Petroleum Formation," Proc. Natl. Acad. Sci. U. S. 46, 212-21 (1960).

136. E. Robinson and R. C. Robbins, Sources, Abundances, and Fate of Gaseous Atmospheric Pollutants, Rept. PR-6755, Stanford Research Institute, Menlo Park, Calif. (February 1968) and Suppl. (June 1969).

137. J. W. Healy and J. J. Fuquay, "Wind Pickup of Radioactive Particles from the Ground," Proc. U. N. Intern. Conf. Peaceful Uses At. Energy, 2nd, Geneva, 1958, 18, 291-95 (1958).

138. M. T. Mills, R. C. Dahlman, and J. S. Olson, personal communication, Oak Ridge National Laboratory, July 1973.

139. L. J. Hagen and N. P. Woodruff, "Air Pollution from Duststorms in The Great Plains," Atmosph. Environ. 7, 323-32 (1973).

140. D. A. Gillette, I. H. Blifford, Jr., and C. R. Fenster, "Measurements of Aerosol Size Distributions and Vertical Fluxes of Aerosols on Land Subject to Wind Erosion," J. of Appl. Meteorol. 11, 977-82 (1972).

141. R. E. Lee, Jr., "The Size of Suspended Particulate Matter in Air," Science 178, 567-75 (1972).

142. Task Group on Lung Dynamics, "Deposition and Retention Models for Internal Dosimetry of the Human Respiratory Tract," Health Phys. 12, 173-207 (1966).

143. E. I. Hamilton, "The Concentration of Uranium in Air from Contrasted Natural Environments," Health Phys. 19, 511-20 (1970).

144. D. H. Peirson, P. A. Cawse, L. Salmon, and R. S. Cambray, "Trace Elements in the Atmospheric Environment," Nature 241, 252-56 (1973).

145. H. J. M. Bowen, Trace Elements in Biochemistry, Academic, New York, 1966.

146. W. John, R. Kaifer, K. Rahn, and J. J. Wesolowski, "Trace Element Concentrations in Aerosols from the San Francisco Bay Area," Atmos. Environ. 7, 107-18 (1973).

147. B. Mason, Principles of Geochemistry, 3rd Ed., Wiley, New York, 1966.

148. K. A. Rahn and J. W. Winchester, Sources of Trace Elements in Aerosols - An Approach to Clean Air, COO-1705-9 (1971).

149. W. S. Chepil, "Properties of Soil Which Influence Wind Erosion: III. Effect of Apparent Density on Erodibility," Soil. Sci. 71, 141-53 (1951).

150. J. H. Healy, "Some Thoughts on Plutonium in Soils," Symposium on Environmental Plutonium, LA-4756, pp. 113-15 (1971).

151. W. H. Langham, "Plutonium Distribution as a Problem in Environmental Science," Symposium on Environmental Plutonium, LA-4756, pp. 3-11 (1971).

152. H. S. Jordan, "Distribution of Plutonium from Accidents and Field Experiments," Symposium on Environmental Plutonium, LA-4756, pp. 21-35 (1971).

153. E. Iranzo and S. Salvador, "Inhalation Risks to People Living Near a Contaminated Area " presented at Second International Congress of the International Radiation Protection Association, Brighton, England, 3-8 May, 1970.

154. H. L. Volchok, "Resuspension of Plutonium-239 in the Vicinity of Rocky Flats," Symposium on Environmental Plutonium, LA-4756, pp. 99-103 (1971).

155. E. Penna Franca, J. C. Almeida, J. Becker, M. Emmerich, F. X. Roser, G. Kegel, L. Hainsberger, T. L. Cullen, H. Petrow, R. T. Drew, and M. Eisenbud, "Status of Investigations in the Brazilian Areas of High Natural Radioactivity," Health Phys. 11, 699-712 (1965).

156. E. Penna Franca, C. Costa Ribeiro, M. Teitakowski, H. Londres, P. L. Santos, and H. A. Albuquerque, "Survey of Radioactive Content of Food Grown on Brazilian Areas of High Natural Radioactivity," Health Phys. 11, 1471-84 (1965).

157. E. Penna Franca, M. Fiszman, N. Lobão, H. Trindade, C. Costa Ribeiro, and P. L. Santos, "Radioactivity in the Diet in High Background Areas of Brazil," Health Phys. 19, 657-62 (1970).

158. D. S. Bharatwal and G. J. Vaze, "Radiation Dose Measurements in the Monazite Areas of Kerala State in India," Proc. U. N. Intern. Conf. Peaceful Uses At. Energy, 2nd, Geneva, 1958, 23, 563-67 (1966).

159. K. B. Mistry, A. R. Gopal-Ayengar, and K. G. Bharathan, "On the Radioactivity of Plants from the High Radiation Areas of the Kerala Coast and Adjoining Regions. II. Studies on the Uptake of Alpha and Gammas Emitters," Health Phys. 11, 1459-70 (1965).

160. K. B. Mistry, K. G. Bharathan, and A. R. Gopal-Ayengar, "Radioactivity in the Diet of Population of the Kerala Coast Including Monazite Bearing High Radiation Areas," Health Phys. 19, 535-42 (1970).

161. A. R. Gopal-Ayengar, K. Sundaram, K. B. Mistry, C. M. Sunta, K. S. V. Nambi, S. P. Kathuria, A. S. Basu, and M. David, "Evaluation of the Long-Term Effects of High Background Radiation on Selected Population Groups on the Kerala Coast," Proc. U. N. Intern. Conf. Peaceful Uses of At. Energy, 4th, Geneva, 1972, 11, 31-51 (1972).

162. Z. E. Arlin, "Deep Well Disposal of Uranium Tailing Water," Second Conference on Ground Disposal of Radioactive Wastes, Chalk River, Canada, 26-29 Sept., 1961, TID-7628, pp. 356-62 (1962).

163. E. C. Tsivoglou and R. L. O'Connell, Waste Guide for the Uranium Milling Industry, U.S.P.H.S., Technical Report W62-12 (1962).

164. J. B. Anderson, E. C. Tsivoglou, and S. D. Shearer, "Effects of Uranium Mill Wastes on Biological Fauna of the Animas River (Colorado - New Mexico)," First National Symposium on Radioecology, Fort Collins, Colorado, Sept. 10-15, 1961, V. Schultz and A. W. Klement, Jr., Eds., pp. 373-83, 1963.

165. E. C. Tsivoglou, S. D. Shearer, J. D. Jones, C. E. Sponagle, H. R. Paren, J. B. Anderson, and D. A. Clark, "Survey of Interstate Pollution of the Animas River (Colorado - New Mexico)," R. A. Taft Sanitary Engineering Center, Cincinnati, U.S.P.H.S. Report, 1960.

166. E. C. Tsivoglou, S. D. Shearer, Jr., J. D. Jones, and D. A. Clark, "Estimating Human Radiation Exposure on the Animas River," J. Am. Water Works Assoc. 52, 1271-90 (1960).

167. S. S. Martin, W. T. Helm, and W. F. Sigler, "Accumulation of ^{226}Ra in Two Aquatic Ecosystems," Second National Symposium of Radioecology, Ann Arbor, Mich., May 15-17, 1957, CONF-670503, pp. 307-18 (1969).

168. S. D. Shearer, Jr., and G. F. Lee, "Leachability of Radium-226 from Uranium Mill Solids and River Sediments," Health Phys. 10, 217-27 (1964).

169. B. Havlík, J. Gráfová, and B. Nýčová, "Radium-226 Liberation from Uranium Ore Processing Mill Waste Solids and Uranium Rocks into Surface Streams - I, The Effect of Different pH of Surface Waters," Health Phys. 14, 417-22 (1968).

170. B. Havlík, B. Nýčová, and J. Gráfová, "Radium-226 Liberation from Uranium Ore Processing Mill Waste Solids and Uranium Rocks into Surface Streams - II, The Effect of Different Chemical Composition of Surface Waters," Health Phys. 14, 423-30 (1968).

171. R. N. Snelling, and S. D. Shearer, Jr., "Environmental Survey of Uranium Mill Tailings Pile, Tuba City, Arizona," Radiol. Health Data 10, 475-87 (1969).

172. R. N. Snelling, "Environmental Survey of Uranium Mill Tailings Pile, Monument Valley, Arizona," Radiol. Health Data 11, 511-17 (1970).

173. R. N. Snelling, "Environmental Survey of Uranium Mill Tailings Pile, Mexican Hat, Utah," Radiol. Health Data 12, 17-28 (1971).

174. Atomic Energy Commission, The Nuclear Industry, 1971, WASH-1174-71, p. 20 (1971).

175. E. C. Tsivoglou and R. L. O'Connell, "Nature, Volume and Activity of Uranium Mill Wastes," Symposium on Radiological Health and Safety in Mining and Milling of Nuclear Materials, Vienna, 26-31 August 1963, IAEA-WHO, Vol. II, pp. 101-22, 1964.

APPENDIX

RADIOACTIVE DECAY SERIES[a]

---

[a] From:

Radiological Health Handbook
U. S. Department of Health, Education, and Welfare
Public Health Service, 1970

Thorium Series (4n)*

Nuclide	Historical name	Half-life	Major radiation energies (MeV) and intensities† α	β	γ
$^{232}_{90}Th$	Thorium	1.41×10^{10}y	3.95 (24%) 4.01 (76%)	---	---
$^{228}_{88}Ra$	Mesothorium I	6.7y	---	0.055 (100%)	---
$^{228}_{89}Ac$	Mesothorium II	6.13h	---	1.18 (35%) 1.75 (12%) 2.09 (12%)	0.34c‡ (15%) 0.908 (25%) 0.96c (20%)
$^{228}_{90}Th$	Radiothorium	1.910y	5.34 (28%) 5.43 (71%)	---	0.084 (1.6%) 0.214 (0.3%)
$^{224}_{88}Ra$	Thorium X	3.64d	5.45 (6%) 5.68 (94%)	---	0.241 (3.7%)
$^{220}_{86}Rn$	Emanation Thoron (Tn)	55s	6.29 (100%)	---	0.55 (0.07%)
$^{216}_{84}Po$	Thorium A	0.15s	6.78 (100%)	---	---
$^{212}_{82}Pb$	Thorium B	10.64h	---	0.346 (81%) 0.586 (14%)	0.239 (47%) 0.300 (3.2%)
$^{212}_{83}Bi$ 64.0% 36.0%	Thorium C	60.6m	6.05 (25%) 6.09 (10%)	1.55 (5%) 2.26 (55%)	0.040 (2%) 0.727 (7%) 1.620 (1.8%)
$^{212}_{84}Po$	Thorium C'	304ns	8.78 (100%)	---	---
$^{208}_{81}Tl$	Thorium C"	3.10m	---	1.28 (25%) 1.52 (21%) 1.80 (50%)	0.511 (23%) 0.583 (86%) 0.860 (12%) 2.614 (100%)
$^{208}_{82}Pb$	Thorium D	Stable	---	---	---

*This expression describes the mass number of any member in this series, where n is an integer.
Example: $^{232}_{90}Th$ (4n)......4(58) = 232

†Intensities refer to percentage of disintegrations of the nuclide itself, not to original parent of series.

‡Complex energy peak which would be incompletely resolved by instruments of moderately low resolving power such as scintillators.

Data taken from: Lederer, C. M., Hollander, J. M., and Perlman, I., Table of Isotopes (6th ed.: New York: John Wiley & Sons, Inc., 1967) and Hogan, O. H., Zigman, P. E., and Mackin, J. L., Beta Spectra (USNRDL-TR-802 [Washington, D.C.: U.S. Atomic Energy Commission, 1964]).

Neptunium Series (4n + 1)*

Nuclide	Element name	Half-life	Major radiation energies (MeV) and intensities†		
			α	β	γ
$^{241}_{94}$Pu -100% 0.0023%	Plutonium	13.2y	4.85 (0.0003%) 4.90 (0.0019%)	0.021 (-100%)	0.145 (.00016%)
$^{241}_{95}$Am	Americium	458y	5.44 (13%) 5.49 (85%)	---	0.060 (36%) 0.101c‡ (0.04%)
$^{237}_{92}$U	Uranium	6.75d	---	0.248 (96%)	0.060 (36%) 0.208 (23%)
$^{237}_{93}$Np	Neptunium	2.14×10^{6}y	4.65c (12%) 4.78c (75%)	---	0.030 (14%) 0.086 (14%) 0.145 (1%)
$^{233}_{91}$Pa	Protactinium	27.0d	---	0.145 (37%) 0.257 (58%) 0.568 (5%)	0.31c (44%)
$^{233}_{92}$U	Uranium	1.62×10^{5}y	4.78 (15%) 4.82 (83%)	---	0.042 (?) 0.097 (?)
$^{229}_{90}$Th	Thorium	7340y	4.84 (58%) 4.90 (11%) 5.05 (7%)	---	0.137c (-3%) 0.20c (-10%)
$^{225}_{88}$Ra	Radium	14.8d	---	0.32 (100%)	0.040 (33%)
$^{225}_{89}$Ac	Actinium	10.0d	5.73c (10%) 5.79 (28%) 5.83 (54%)	---	0.099 (?) 0.150 (?) 0.187 (?)
$^{221}_{87}$Fr	Francium	4.8m	6.12 (15%) 6.34 (82%)	---	0.218 (14%)
$^{217}_{85}$At	Astatine	0.032s	7.07 (-100%)	---	---
$^{213}_{83}$Bi 97.8% 2.2%	Bismuth	47m	5.87 (-2.2%)	1.39 (-97.8%)	0.437 (?)
$^{213}_{84}$Po	Polonium	4.2μs	8.38 (-100%)	---	---
$^{209}_{81}$Tl	Thallium	2.2m	---	1.99 (100%)	0.12 (50%) 0.45 (100%) 1.56 (100%)
$^{209}_{82}$Pb	Lead	3.30h	---	0.637 (100%)	---
$^{209}_{83}$Bi	Bismuth	Stable ($>2\times10^{18}$y)	---	---	---

*This expression describes the mass number of any member in this series, where n is an integer.

Example: $^{229}_{90}$Th (4n + 1)......4(57) + 1 = 229

The (4n + 1) series is included here for completion. It is not found as a naturally-occurring series.

†Intensities refer to percentage of disintegrations of the nuclide itself, not to original parent of series.

‡Complex energy peak which would be incompletely resolved by instruments of moderately low resolving power such as scintillators.

Data taken from: Table of Isotopes and USNRDL-TR-802.

Uranium Series (4n + 2)*

Nuclide	Historical name	Half-life	Major radiation energies (MeV) and intensities†		
			α	β	γ
$^{238}_{92}$U	Uranium I	4.51x10⁹y	4.15 (25%) 4.20 (75%)	---	---
$^{234}_{90}$Th	Uranium X_1	24.1d	---	0.103 (21%) 0.193 (79%)	0.063c‡ (3.5%) 0.093c (4%)
$^{234}_{91}$Pa^m 99.87% | 0.13%	Uranium X_2	1.17m	---	2.29 (98%)	0.765 (0.30%) 1.001 (0.60%)
$^{234}_{91}$Pa	Uranium Z	6.75h	---	0.53 (66%) 1.13 (13%)	0.100 (50%) 0.70 (24%) 0.90 (70%)
$^{234}_{92}$U	Uranium II	2.47x10⁵y	4.72 (28%) 4.77 (72%)	---	0.053 (0.2%)
$^{230}_{90}$Th	Ionium	8.0 x10⁴y	4.62 (24%) 4.68 (76%)	---	0.068 (0.6%) 0.142 (0.07%)
$^{226}_{88}$Ra	Radium	1602y	4.60 (6%) 4.78 (95%)	---	0.186 (4%)
$^{222}_{86}$Rn	Emanation Radon (Rn)	3.823d	5.49 (100%)	---	0.510 (0.07%)
$^{218}_{84}$Po 99.98% | 0.02%	Radium A	3.05m	6.00 (~100%)	0.33 (~0.019%)	---
$^{214}_{82}$Pb	Radium B	26.8m	---	0.65 (50%) 0.71 (40%) 0.98 (6%)	0.295 (19%) 0.352 (36%)
$^{218}_{85}$At	Astatine	~2s	6.65 (6%) 6.70 (94%)	? (~0.1%)	---
$^{214}_{83}$Bi 99.98% | 0.02%	Radium C	19.7m	5.45 (0.012%) 5.51 (0.008%)	1.0 (23%) 1.51 (40%) 3.26 (19%)	0.609 (47%) 1.120 (17%) 1.764 (17%)
$^{214}_{84}$Po	Radium C'	164μs	7.69 (100%)	---	0.799 (0.014%)
$^{210}_{81}$Tl	Radium C"	1.3m	---	1.3 (25%) 1.9 (56%) 2.3 (19%)	0.296 (80%) 0.795 (100%) 1.31 (21%)
$^{210}_{82}$Pb	Radium D	21y	3.72 (.0000002%)	0.016 (85%) 0.061 (15%)	0.047 (4%)
$^{210}_{83}$Bi ~100% | .00013%	Radium E	5.01d	4.65 (.00007%) 4.69 (.00005%)	1.161 (~100%)	---
$^{210}_{84}$Po	Radium F	138.4d	5.305 (100%)	---	0.803 (0.0011%)
$^{206}_{81}$Tl	Radium E"	4.19m	---	1.571 (100%)	---
$^{206}_{82}$Pb	Radium G	Stable	---	---	---

*This expression describes the mass number of any member in this series, where n is an integer.

Example: $^{206}_{82}$Pb (4n + 2)......4(51) + 2 = 206

†Intensities refer to percentage of disintegrations of the nuclide itself, not to original parent of series.

‡Complex energy peak which would be incompletely resolved by instruments of moderately low resolving power such as scintillators.

Data taken from: Table of Isotopes and USNRDL-TR-802.

Actinium Series (4n + 3)*

Nuclide	Historical name	Half-life	Major radiation energies (MeV) and intensities†		
			α	β	γ
$^{235}_{92}$U	Actinouranium	7.1 ×10^8y	4.37 (18%) 4.40 (57%) 4.58c‡ (8%)	---	0.143 (11%) 0.185 (54%) 0.204 (5%)
$^{231}_{90}$Th	Uranium Y	25.5h	---	0.140 (45%) 0.220 (15%) 0.305 (40%)	0.026 (2%) 0.084c (10%)
$^{231}_{91}$Pa	Protoactinium	3.25×10^4y	4.95 (22%) 5.01 (24%) 5.02 (23%)	---	0.027 (6%) 0.29c (6%)
$^{227}_{89}$Ac 98.6% 1.4%	Actinium	21.6y	4.86c (0.18%) 4.95c (1.2%)	0.043 (~99%)	0.070 (0.08%)
$^{227}_{90}$Th	Radioactinium	18.2d	5.76 (21%) 5.98 (24%) 6.04 (23%)	---	0.050 (8%) 0.237c (15%) 0.31c (8%)
$^{223}_{87}$Fr	Actinium K	22m	5.44 (~0.005%)	1.15 (~100%)	0.050 (40%) 0.080 (13%) 0.234 (4%)
$^{223}_{88}$Ra	Actinium X	11.43d	5.61 (26%) 5.71 (54%) 5.75 (9%)	---	0.149c (10%) 0.270 (10%) 0.33c (6%)
$^{219}_{86}$Rn	Emanation Actinon (An)	4.0s	6.42 (8%) 6.55 (11%) 6.82 (81%)	---	0.272 (9%) 0.401 (5%)
$^{215}_{84}$Po ~100% .00023%	Actinium A	1.78ms	7.38 (~100%)	0.74 (~.00023%)	---
$^{211}_{82}$Pb	Actinium B	36.1m	---	0.29 (1.4%) 0.56 (9.4%) 1.39 (87.5%)	0.405 (3.4%) 0.427 (1.8%) 0.832 (3.4%)
$^{215}_{85}$At	Astatine	~0.1ms	8.01 (~100%)	---	---
$^{211}_{83}$Bi 0.28% 99.7%	Actinium C	2.15m	6.28 (16%) 6.62 (84%)	0.60 (0.28%)	0.351 (14%)
$^{211}_{84}$Po	Actinium C'	0.52s	7.45 (99%)	---	0.570 (0.5%) 0.90 (0.5%)
$^{207}_{81}$Tl	Actinium C"	4.79m	---	1.44 (99.8%)	0.897 (0.16%)
$^{207}_{82}$Pb	Actinium D	Stable	---	---	---

*This expression describes the mass number of any member in this series, where n is an integer.
Example: $^{207}_{82}$Pb (4n + 3)......4(51) + 3 = 207

†Intensities refer to percentage of disintegrations of the nuclide itself, not to original parent of series.

‡Complex energy peak which would be incompletely resolved by instruments of moderately low resolving power such as scintillators.

Data taken from: Table of Isotopes and USNRDL-TR-802.

EPA 520/6-78-007-a
August 1978
Volume 1 of 2

RADIONUCLIDE INTERACTIONS WITH SOIL AND ROCK MEDIA

Volume 1: Processes Influencing Radionuclide Mobility and Retention Element Chemistry and Geochemistry Conclusions and Evaluation

by
L. L. Ames
Dhanpat Rai

Battelle
Pacific Northwest Laboratories
Richland, Washington 99352

Final Report for Contract 68-03-2514

Project Officer
Robert F. Kaufmann
Evaluation Branch
Office of Radiation Programs-Las Vegas Facility
U.S. Environmental Protection Agency
Las Vegas, Nevada 89114

1a

RADIUM

Natural Soil and Rock Distributions

The radium content of various igneous and sedimentary rocks is given in Table 3-60. Vinogradov (1959) reported that the radium content of soils ranged from 0.5×10^{-6} to 1.1×10^{-6} ppm. The small amount of radium present in soils has become separated from parent thorium and uranium during rock weathering processes, and most closely follows barium. The radium content of igneous rocks increases by 100 times from ultra-basic to granitic rocks. The radium content of most sedimentary rocks is about the same as that for granites.

TABLE 3-60. AVERAGE RADIUM CONTENT OF VARIOUS ROCK TYPES

Rock Type	Ra, ppm	Reference
Ultrabasic Igneous	0.009×10^{-6}	Davis, 1947
Basic Igneous	0.6×10^{-6}	Evans et al., 1942
Intermediate, Igneous	0.917×10^{-6}	Senftle and Keevil, 1947
Granitic Igneous	1.395×10^{-6}	Senftle and Keevil, 1947
Sandstones	0.71×10^{-6}	Bell et al., 1940
Shales	1.08×10^{-6}	Bell et al., 1940
Limestones	0.42×10^{-6}	Evans and Goodman, 1941

Brief Chemistry

There are 16 isotopes of radium from ^{213}Ra to ^{230}Ra with no ^{218}Ra or ^{229}Ra. All of the isotopes of radium are unstable, and all of the naturally-occurring radium isotopes occur in the thorium decay series (^{228}Ra, ^{224}Ra), the ^{238}U-radium decay series (^{226}Ra), and the ^{235}U-actinium decay series (^{223}Ra) (Vdovenko and Dubasov, 1973). Radium radionuclide data are given in Table 3-61. All are alpha emitters except ^{228}Ra which is a beta emitter. With the present uranium fuel cycle and the much longer half-life, only the ^{226}Ra isotope is of long-term concern in waste disposal. If future energy production includes thorium fuels, then ^{228}Ra also must be included in waste disposal management plans.

TABLE 3-61. RADIUM RADIONUCLIDE DATA
(VDOVENKO AND DUBASOV, 1973)

Isotope	Half-Life	Decay Mode
^{223}Ra	11.43 days	α
^{224}Ra	3.64 days	α
^{225}Ra	14.8 days	β^-
^{226}Ra	1622 years	α
^{228}Ra	5.77 ± 0.02 years	β^-

Radium is a homolog of the alkaline-earth elements, with a (II) oxidation state. The radius of the +2 radium ion is 1.52 Å compared to 1.43 Å for Ba^{+2}. The compounds formed by radium and their solubilities are similar to barium. For example, the solubility product of radium sulfate is 4.25×10^{-11} at 25°C (Vdovenko and Dubasov, 1973) and the solubility product of barium sulfate is 1.08×10^{-10} at 25°C (Weast, 1976). Metallic radium dissolves in water with the evolution of hydrogen and the formation of readily soluble $Ra(OH)_2$. Of the alkaline-earth metal cations, Ra^{+2} shows the least tendency for complex formation, although 1:1 complexes with citric, tartaric, succinic and several other acids were detected at pH 7.2 to 7.4 by Schubert et al. (1950). It may be assumed that Ra^{+2} is not hydrolyzed in aqueous solutions, in an analogy with Ba^{+2}, although there is no specific literature on the subject.

Solid Phase and Solution Equilibria

The thermodynamic data for radium compounds are available only for radium nitrate, chloride, iodate and sulfate (Parker et al., 1971). However, all of the compounds except sulfate are very soluble. Therefore, solid phase diagrams are not presented for radium compounds. The solubility product for $RaSO_4$ is $10^{-10.37}$ compared to $10^{-9.96}$ for $BaSO_4$.

No thermodynamic data were located for radium hydrolysis or complex ion species. It is expected the radium will behave in the soil solution much like strontium does. The species Ra^{2+} is expected to be the most important over the normal soil pH range from 4 to 8.

Experimental Adsorption Results

Stead (1964) gave a radium Kd value of 6700 ml/g for NTS tuff. Arnold and Crouse (1965) ran batch adsorption tests on some exchange materials that included the natural zeolites, clinoptilolite and chabazite, as represented by a pelletized molecular sieve (AW-500) and barite (barytes), a natural $BaSO_4$.

The results of the adsorption tests were recomputed as distribution coefficients or Kd values. The solution was a lime-neutralized waste that contained 4100 pCi ^{226}Ra/l, 500 mg/l Ca^{+2}, 80 mg/l Mg^{+2}, 1000 mg/l Na^{+}, 2500 mg/l SO_4^{-2} and 900 mg/l Cl^{-} at pH 7.7. The radium Kd values are given in Table 3-62.

TABLE 3-62. RADIUM Kd VALUES FROM LIME-NEUTRALIZED WASTE; 1.25 g EXCHANGER/l OF WASTE (ARNOLD AND CROUSE, 1965)

Exchanger	Mesh Size	Exchanger Loading, pCi/g	Kd, ml/g
Clinoptilolite	20-50	2650	646
Chabazite	20-50	2900	707
Barite	20-50	2000	490

R. J. Serne of PNL, 1974, used soils from Utah and simulated river water to determine radium distribution coefficients. The soils were pre-equilibrated by four washings with the simulated river water composition shown in Table 3-63 minus the $^{226}RaCl_2$. The fifth solution contained the ^{226}Ra as well as the other constituents, and was used for the radium Kd determinations. The Kd values were determined in triplicate to allow measurement of precision. The Utah soils contained 2 to 5% calcite, with quartz and feldspar constituting the bulk of these sandy, arid soils. Minor constituents included hydromica and a small amount of a smectite clay. The radium Kd results are listed in Table 3-64. The Kd correlated with the cation exchange capacities of the soils.

TABLE 3-63. SIMULATED RIVER WATER COMPOSITION (SERNE, 1974)

Constituent	Added as-	mg/l
Ca	$CaSO_4 \cdot 2H_2O$, $CaCl_2$	82
Mg	$MgSO_4$	26
Na	NaCl	75
K	KCl	3.4
HCO_3	$NaHCO_3$	171
SO_4	$CaSO_4 \cdot 2H_2O$, $MgSO_4$	246
Cl	$CaCl_2$, NaCl, KCl	57
U	UO_3	1
Ra	$RaCl_2$ in HCl solution	7 µg/l

TABLE 3-64. RADIUM DISTRIBUTION COEFFICIENTS WITH THE SOLUTION OF TABLE 3-63 (SERNE, 1974)

Soil	Final pH	Kd, ml/g
I	7.9	
	7.9	354 ± 15
	8.0	
II	7.6	
	7.7	289 ± 7
	7.6	
III	7.8	
	7.9	467 ± 15
	7.8	
IV	7.8	
	7.6	214 ± 15
	7.8	

Migration Results

Field Studies--

Granger et al. (1961) and Granger (1963) showed that radium had migrated out of Ambrosia Lake, New Mexico, uranium ores and had been partly reconcentrated in barite ($BaSO_4$) and cryptomelane ($KMn^{2+}Mn^{4+}_{7.25}O_{16} \cdot H_2O$) found in and near some of the ore bodies. The $^{226}Ra^{2+}$ occurred in the Ba^{2+} position in barite and the Mn^{2+} position in cryptomelane. The high concentrations of ^{226}Ra were not associated with parent uranium, which is good evidence for the recent migration of radium. The strongly to weakly radioactive cryptomelane partially replaced mudstone that occurred near the ore bodies and was relatively low in uranium content and enriched in lead. This suggests that the lead is radiogenic and has also migrated with the ^{226}Ra. The mechanism involved in reconcentration of the ^{226}Ra is the substitution of $^{226}Ra^{2+}$ for the chemically very similar Ba^{2+} in barite and Mn^{2+} in cryptomelane. Analyses of the outer 5 to 10 cm of mudstone layers near ore disclosed anomalously high radioactivity coupled with an abnormally high lead content. Within mudstone layers, however, the radioactivity was essentially in balance with the uranium content, and the lead content was low.

Hansen and Huntington (1969) determined radium and thorium distributions in a series of morainal soils in Bench Valley, California. Thorium accumulated immediately beneath horizons containing a high amount of organic material. The thorium apparently migrated as organic complexes. Radium was distributed with the uranium in the high organic layers.

The ground and surface water sampling and analyses for radium given by Wruble et al. (1964) and Kaufmann et al. (1975, 1976) for waters of the Colorado Basin and Grants area, New Mexico, respectively, also illustrate the tendency of the radium daughter to become separated from uranium due initially to uranium mining and milling operations, and to continue migrating due to inherent geochemical differences between uranium and radium.

Laboratory Studies --

Several reports have been concerned with the leaching of ^{226}Ra from uranium recovery process tailings and wastes (Whitman and Porter, 1958; Anonymous, 1960; Feldman, 1961), and other authors have shown that radium can be leached from stream sediments, minerals and uranium mill wastes (Starik and Polevaya, 1958; Starik and Lazerev, 1960). The factors that influence leachability of ^{226}Ra from uranium mill waste solids and river sediments were investigated by Shearer (1962) and Shearer and Lee (1964). Less than 1.5 wt% of the river sediments and mill waste solids were greater than plus 20 mesh size and less than 30 wt% were minus 140 mesh in particle size. The amounts of radium leached with distilled water versus time showed that essentially no radium was leached after 15 min. Diffusion of radium from the interior of the particles was relatively insignificant. By varying the liquid to solid ratio in distilled water-solid leaching equilibria, it was shown that the ratio affects the amount of radium leached. The largest ratio effect was shown with leaching of the acid leach process tailings, less with alkaline leach process tailings and the least from river sediments. It was demonstrated that sulfate was present in the waste solids and that the sulfate was easily solubilized. Trace amounts of barium present led to precipitation of $BaSO_4$ and the coprecipitation of $RaSO_4$. If radium was added prior to the solids-distilled water equilibria, it too was removed from solution by coprecipitation with $BaSO_4$. One-hundredth molar solutions were used in leaching equilibria (100 ml/g river sediment) to determine effects on radium leaching. $MgCl_2$, KCl, NaCl, HCl and water solutions all leached less than 1 µµg of radium while $CaCl_2$ leached 1.2 µµg Ra, $SrCl_2$ 6.3 µµg Ra and $BaCl_2$ 30 µµRa. Apparently the radium on river sediments is in the form of Ra^{+2} and is exchangeable.

Havlik et al. (1968a) investigated the leaching of ^{226}Ra from uranium mill solids and uranium ores. The first report concerned the effects of pH on leaching rates. The authors found, like Shearer and Lee (1964), that equilibrium leaching was rapidly accomplished in 15 to 30 min. Homogenized uranium ores (540 pCi Ra/g and 85 pCi Ra/g) and milling tailings (14 pCi Ra/g) were shaken for varying lengths of time as a 3g solid/30 ml solution ratio. The pH was modified from 1 to 14 with hydrochloric acid, boric acid and sodium hydroxide. At pH 1, 22% of the ^{226}Ra was liberated. At pH 9, the amount leached had decreased to 2.8%. At pH 13, the amount of leached ^{226}Ra increased to 5%.

The second report by Havlik et al. (1968b) studied the leaching of radium from the same solids as affected by leaching solution composition in addition to acidity. Unlike the leaching results of Shearer and Lee (1964), Havlik et al. found that the largest concentrations of radium were leached by 1N KCl and 1N NaCl solutions. $BaCl_2$, $SrCl_2$ and $CaCl_2$ also were used, but with much less radium liberated. The 1N KCl leached 100% of the radium in mill tailings and NaCl, 95%. Uranium ore leaching results were lower, showing 22% and 31% leached by KCl and 14% and 17% leached by NaCl. In all cases, the one normal salt solutions removed more radium than ten normal salt solutions.

Summary

Radium is present as Ra^{2+} over the normal soil pH range (4 to 8) and shows little tendency to form complex species (Schubert et al., 1950). Radium would be expected to substitute for other divalent cations during replacement or precipitation reactions (Granger et al., 1961; Granger, 1963). A direct correlation of cation exchange capacity with adsorption (Arnold and Crouse, 1965; Serne, 1974) and leaching studies with different types of competing cations (Starik and Polevaya, 1958; Starik and Lazerev, 1960; Shearer, 1962; Shearer and Lee, 1964; Havlik et al., 1968b) suggests that an important reaction mechanism for radium adsorption is cation exchange. Radium could be expected to migrate in much the same manner as strontium.

References

Anonymous. 1960. January, 1960. Summary Report on I. The Control of Radium and Thorium in the Uranium Milling Industry, II. Radium-226 Analysis Principles, Interference and Practice, III. Current Winchester Laboratory Projects. WIN-112.

Arnold, W. D. and D. J. Crouse. 1965. Radium Removal from Uranium Mill Effluents with Inorganic Ion Exchangers. I&EC Process Design and Development. 4:335-337.

Bell, K. G., C. Goodman and W. L. Whitehead. 1940. Radioactivity of Sedimentary Rocks and Associated Petroleum. Bull. Am. Assoc. Petroleum Geol. 24:1529.

Butler, J. N. 1964. Ionic Equilibrium. Addison-Wesley, Reading, MA. pp. 180-183.

Davis, G. L. 1947. Radium Content of Ultramafic Igneous Rocks. I. Laboratory Investigation. Am. J. Sci. 245:677.

Evans, R. D. and C. Goodman. 1941. Radioactivity of Rocks. Bull. Geol. Soc. Am. 52:459.

Evans, R. D., C. Goodman, N. B. Keevil. 1942. Radioactivity: The Earth's Heat and Geological Age Measurements. IN: Handbook of Physical Constants. Geol. Soc. Am. Special Papers 36, Sec. 18, p. 267.

Feldman, M. H. 1961. Summary Report, 1959-1961, Winchester Laboratory, Winchester, Massachusetts and Grand Junction, Colorado. WIN-125.

Granger, H. C. 1963. Radium Migration and Its Effect on the Apparent Age of Uranium Deposits at Ambrosia Lake, New Mexico. U.S. Geol. Survey Prof. Paper 475-B, pp. B-60-B-63.

Granger, H. C., E. S. Santos, B. G. Dean and F. B. Moore. 1961. Sandstone-Type Uranium Deposits at Ambrosia Lake, New Mexico - an Interim Report. Econ. Geol., vol. 56, no. 7, pp. 1179-1209.

Hansen, R. O. and G. L. Huntington. 1969. Thorium Movements in Morainal Soils of the High Sierra, California. Soil Science. 108:257-265.

Havlik, B., J. Grafova, and B. Nycova. 1968a. Radium-226 Liberation from Uranium Ore Processing Mill Waste Solids and Uranium Rocks into Surface Streams - I. Health Physics. 14:417-422.

Havlik, B., B. Nycova, and J. Grafova. 1968b. Radium-226 Liberation from Uranium Ore Processing Mill Waste Solids and Uranium Rocks into Surface Streams - II. Health Physics. 14:423-430.

Kaufmann, R. F., G. G. Eadie and C. R. Russell. 1975. Summary of Ground Water Quality Impacts of Uranium Mining and Milling in the Grants Mineral Belt, New Mexico. U.S.E.P.A., Off. Rad. Prog.-Las Vegas Facility. Tech. Note ORP/LV-75-4.71 p.

Kaufmann, R. F., G. G. Eadie and C. R. Russell. 1976. Effects of Uranium Mining and Milling on Ground Water in the Grants Mineral Belt, New Mexico. Ground Water. 14(5):296-308.

Parker, V. B., D. D. Wagman, and W. H. Evans. 1971. Selected Values of Chemical Thermodynamic Properties. U.S. Department of Commerce. NBS Technical Note 270-6.

Schubert, A. J., E. R. Russell, and L. S. Myers. 1950. Dissociation Constants of Radium Inorganic Acid Complexes Measured by Ion Exchange. J. Biol. Chem. 185:387-398.

Senftle, F. E. and N. B. Keevil. 1947. Thorium-Uranium Ratios in the Theory of Genesis of Lead Ores. Trans. Am. Geophys. Union. 28:732.

Shearer, S. D., Jr. 1962. The Leachability of Radium-226 from Uranium Mill Solids and River Sediments. Thesis. Univ. Wisconsin.

Shearer, S. D., Jr. and G. F. Lee. 1964. Leachability of Radium-226 from Uranium Mill Solids and River Sediments. Health Physics. 10:217-227.

Starik, I. E. and K. F. Lazarev. 1960. Effect of Crushing of Minerals on the Extraction of Radioactive Elements. Radiokhimiya. II:749-752.

Starik, I. E. and N. I. Polevaya. 1958. The Leachability of ThX and Rd Th from Minerals. AEC-tr-4208, pp. 108.

Stead, F. W. 1964. Distribution in Groundwater of Radionuclides from Underground Nuclear Explosions. IN: Proc. Third Plowshare Symp. Engineering with Nuclear Explosives. April 21-23, 1964. TID-7695, p. 127-138.

Vdovenko, V. M. and Yu. V. Dubasov. 1973. Analytical Chemistry of Radium. Israel Program for Scientific Translations.

Vinogradov, A. P. 1959. The Geochemistry of Rare and Dispersed Chemical Elements in Soils. Consultants Bureau.

Weast, R. C. 1976. Handbook of Chemistry and Physics. The Chemical Rubber Co.

Whitman, A. and E. S. Porter. 1958. Chemical Stream Pollution from Uranium Mills. WIN-99.

Wruble, D. T., S. D. Shearer, D. E. Rushing and C. E. Sponagle. 1964. Radioactivity in Waters and Sediments of the Colorado River Basin, 1950-1963. Rad. Health Data, 5(11):557-567.

THORIUM

Natural Soil and Rock Distributions

The content of thorium in rocks and soil is given in Table 3-94. The thorium in sedimentary rocks may be due either to the selective adsorption of thorium on clays or its retention in heavy resistate minerals such as monazite. Concentrations of thorium in metamorphic rocks are highly variable. The thorium content of igneous rocks increases from basalts to granites.

TABLE 3-94. THORIUM CONTENT OF COMMON ROCKS AND SOILS

Rock Type	Thorium Average or Range, ppm	Reference
	Igneous Rocks	
Granites	10-20	Rogers, 1964
Intermediate	2-10	Heier and Carter, 1964
Basalt and Gabbros	0.5-2	Heier and Rogers, 1963
	Sedimentary Rocks	
Shales, North America	10-15	Adams and Weaver, 1958
Bauxites	49	Adams and Richardson, 1960
Bentonites	24	Adams and Weaver, 1958
Limestones	1.1	Adams and Weaver, 1958
Sandstones	1.7	Murray and Adams, 1958
Soils	6	Vinogradov, 1959
	Metamorphic Rocks	
Marble	0.03	Pliler, 1956
Slate	7.5	Pliler, 1956
Phyllite	5.5	Pliler, 1956
Schist	7.5	Pliler, 1956
Gneiss	13.1	Billings, 1962

Thorium adsorption studies were performed by several investigators. Holland and Kulp (1954) found red clay, globigerina ooze and green clay readily adsorbed thorium. They concluded that ion exchange was the adsorption mechanism. Adams et al. (1959) suggested that thorium is concentrated by clay minerals. Up to 50 ppm thorium in the aluminum hydroxide and resistate minerals in bauxite was reported by Adams and Richardson (1960).

Brief Chemistry

There are 13 isotopes of thorium with six of them found in nature. Of the six natural thorium isotopes, five are relatively quantitatively unimportant members of the ^{238}U, ^{235}U or ^{232}Th decay series. Thorium-232 is the major isotope, with a half-life of 1.39 x 10^{10} years (Ryabchikov and Golbraikh, 1969). Thorium radionuclide data of interest in waste disposal operations are in given in Table 3-95.

TABLE 3-95. THORIUM RADIONUCLIDE DATA (RYABCHIKOV AND GOLBRAIKH, 1969)

Isotope	Half-Life	Decay Mode
^{227}Th	18.5 days	α
^{228}Th	1.913 years	α
^{229}Th	7340 years	α
^{230}Th	80,000 years	α
^{231}Th	25.5 hours	β^-
^{232}Th	1.41 x 10^{10} years	α
^{234}Th	24.1 days	β^-

Although other oxidation states of thorium are known in the laboratory, only Th(IV) is found in nature. Th(IV) is found as Th^{+4}. The atomic radius of Th^{+4} is 0.99 Å (Ahrens, 1952).

Common insoluble thorium compounds include the hydroxide, fluoride and phosphate. Soluble compounds include the chloride, nitrate and sulfate. Thorium in solution is a small, highly charged ion that undergoes extensive interaction with water and many anions. The solution chemistry of thorium is largely a study of its complex ions. Common anions that form strong complexes with thorium include fluoride, chloride, nitrate, phosphate and sulfate. At pH values above 3, thorium undergoes hydrolysis in aqueous solutions. During the sedimentary cycle, thorium usually becomes separated from uranium because the uranium tends to mobilize in its U(VI) oxidation state until encountering a reducing environment to become immobilized U(IV). Thorium does not undergo a comparable oxidation state change.

Solid Phase and Solution Equilibria

Figure 3-24 relates the activity of Th^{4+} to pH under an assumed weathering environment in equilibrium with various thorium solid phases. The thermodynamic data for $ThO_2(s)$ were selected from Baes and Mesmer (1976). The data for the other compounds were selected from Sillen and Martell (1964). Under the assumptions outlined in Figure 3-24, all thorium compounds except ThF_4 can be arranged in an increasing order of stability throughout the pH range as follows: $Th(HPO_4)_2$, $Th_3(PO_4)_4$, $Th(OH)_4$, and ThO_2. ThF_4 would be least stable in a pH of approximately >7 and most stable approximately pH <4.

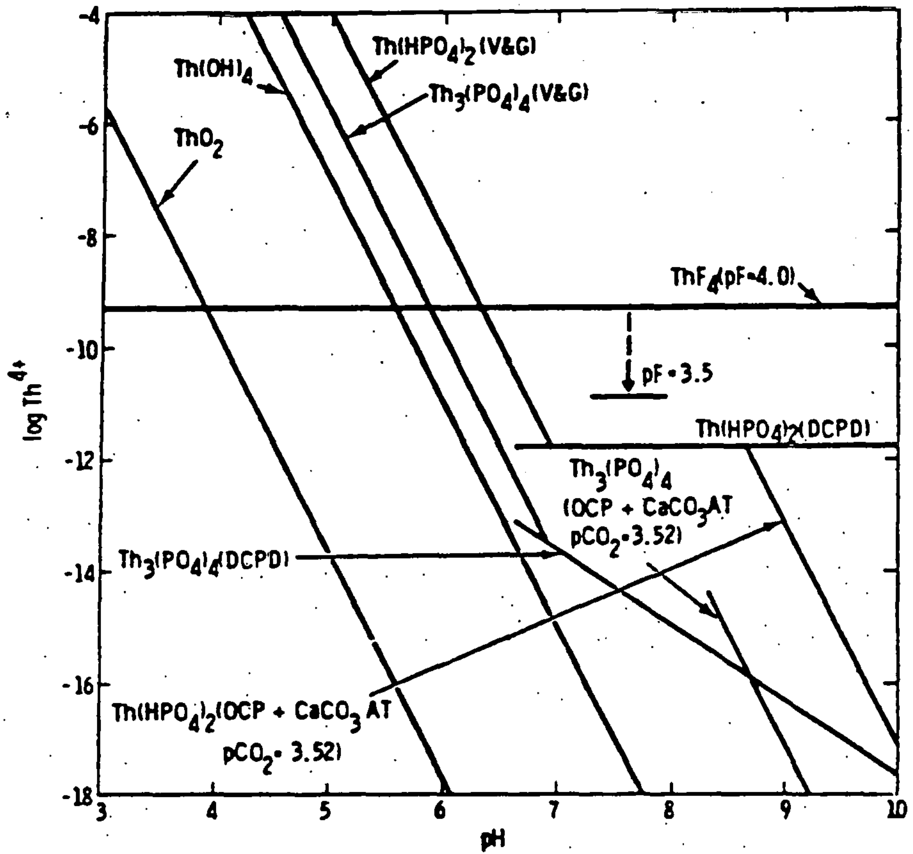


Figure 3-24. The relative stability of various thorium solids in equilibrium with Variscite and Gibbsite (V & G) Dicalcium Phosphate Dihydrate (DCPD) and Octacalcium Phosphate (OCP)

The relative activity of solution species of thorium in equilibrium with $ThO_2(s)$ at assumed activities of various ions is plotted in Figure 3-25. The thermodynamic data for all the hydrolysis species except $Th(OH)_5^-$ were obtained from Baes and Mesmer (1976). The data for all the other species and $Th(OH)_5^-$

were obtained from Sillen and Martell (1964). In general, the total concentration of thorium in solution decreases with an increase of pH from zero to 5. Above 5, pH does not affect thorium concentration in solution due to the formation of $Th(OH)_4^o$. The activity of all positively charged species decreases with an increase in pH, while the activity of the negatively charged species increases with the increase in pH. Under the conditions assumed for Figure 3-25, the total activity of thorium in solution would be expected to be approximately $10^{-9.6}$ moles/liter above pH 5.

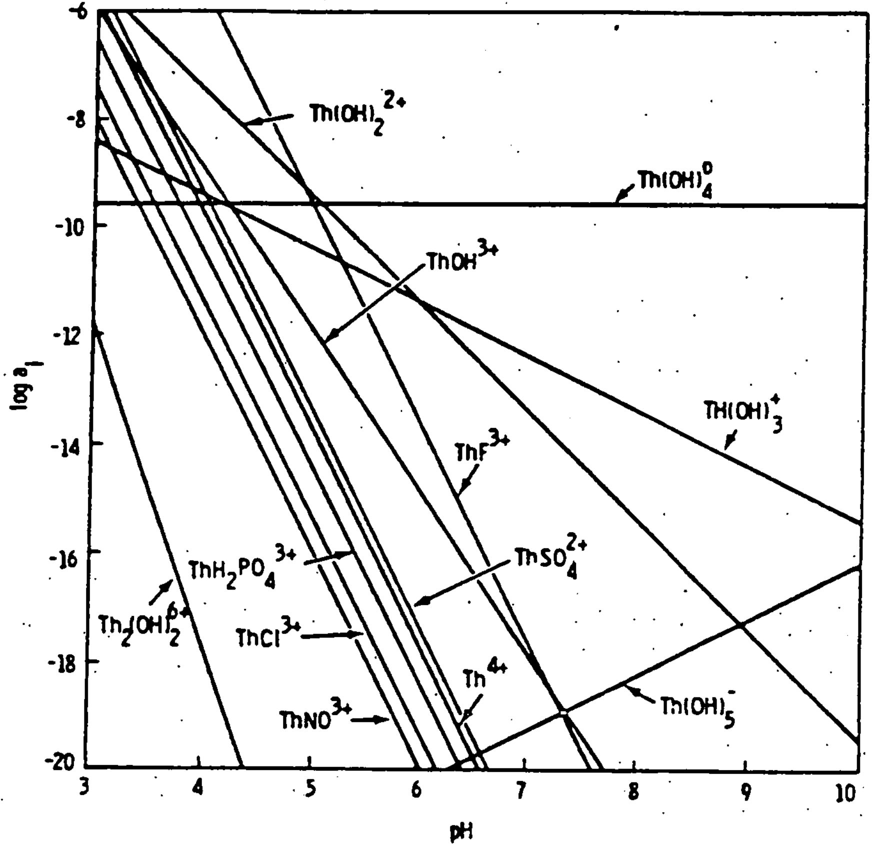


Figure 3-25. Activity of various thorium species in soil in equilibrium with $ThO_2(s)$, $pCl^- = pNO_3^- = pSO_4^{2-} = 3.0$, $pF^- = 4.5$ and $pH_2PO_4^- = 5.0$.

In addition to OH^-, thorium forms various complexes with SO_4^{2-}, PO_4^{3-}, Cl^-, NO_3^-, and F^-. Various anions in increasing order of their importance to forming complexes are: NO_3^-, Cl^-, $H_2PO_4^-$, SO_4^{2-}, and F^-. Figure 3-25 shows that thorium exists as Th^{4+} only in very acidic solutions (pH < 3). Above pH 3, Th^{4+} hydrolyzes very rapidly and it does not contribute significantly to the total thorium concentration. Under the conditions assumed for Figure 3-25, ThF^{3+} would be a dominant solution species at pH < 5, and $Th(OH)_4^{\circ}$ would be a dominant solution species at pH > 5. If fluoride ion is absent from the solution, or its concentration is extremely low, $Th(OH)_2^{2+}$ would mainly control the thorium concentration in solution at pH < 5.

Experimental Adsorption Results

Schulz (1965) found the thorium in soils to be strongly adsorbed by clay particles or present as insoluble oxides and hydroxides. Rubtsov (1966, 1972) found thorium to associate with the fine-grained particles during soil weathering. Katsurayama (1968) determined the distribution coefficient of thorium but data are not presented in the available abstract.

Nishiwaki et al. (1972) spiked seawater and seawater distilled water mixtures with Th^{+4} and measured the adsorption on a medium sand, very fine sand and silt-clay. Twenty grams of soil were contacted with 4 liters of spiked water and mixed until equilibrium was reached. The Kd for thorium increased as the particle size of the soil decreased. Chlorosity of the water did not appear to consistently affect the thorium Kd for the fine sand or silt-clay. The Kd for the medium sand increased as the chlorosity of the water decreased. The chlorosity effect was compounded by a variable pH of the various salt solutions, so that the exact cause of the trend was not determinable. Kd values for the medium sand, very fine sand and silt-clay were 40 to 130, 310 to 470, and 2700 to 10,000 ml/g, respectively.

Rancon (1973) measured the thorium Kd for a soil developed on a schist consisting of quartz and clay with no calcite or organic matter, for a mixed quartz-clay-calcite-organic matter soil and for illite with 100 mg Th/l versus solution pH. For the quartz-clay soil, at pH 6 the Kd was 5×10^5 ml/g, at pH 4 the Kd was about 1×10^3 ml/g and at pH 2 the Kd was about 5 ml/g. The mixed quartz-clay-calcite-organic matter soil could not be lowered in pH without removal of soil calcite, but above pH 8, the thorium Kd dropped from 10^6 ml/g to 100 ml/g at pH 10. Dissolution of humic acids in the soil probably resulted in thorium complexation and a decreased Kd with rising pH. Illite behaved similarly to the quartz-clay soil, but the thorium Kd at pH 1 was about 500 ml/g and about 1×10^5 ml/g at pH 6.5. For soils without calcite or organic material, the thorium Kd decreased as the thorium concentration in solution initially increased. Calcareous soils neutralized even high strength thorium solutions to precipitate $Th(OH)_4$. The quartz-clay soil and illite, for example, gave thorium Kd values of 8 ml/g and 120 ml/g, respectively, in a 1 g Th/l solution, and 60 ml/g and 1000 ml/g, respectively, in a 0.1 g Th/l solution. The drop in thorium Kd was caused by saturation of available exchange sites as a result of increased thorium concentration. There was evidence for the concentration dependence of the thorium Kd down to 1 mg Th/l in the initial contacting solution. In general, three types of soil-thorium adsorption reactions were found: 1) $Th(OH)_4$ precipitation as a result of soil calcite buffering, 2) strong adsorption on clay-containing soils and dilute thorium (<1 g/l) solutions at a pH above 2, and 3) strong adsorption on organic-containing soils at the neutral to acid pH range, but diminishing adsorption into the alkaline pH range.

Bondietti (1974) studied the adsorption of hydrolyzed thorium from waters at pH 6.5 by calcium saturated reference clays (montmorillonite and kaolinite) and calcium humate and found 95% and 99.9+% adsorption, respectively. Desorption studies utilizing calcium citrate removed 10 to 30% of the thorium from the clays but only 1% from the humate. Stronger complexers (DTPA and EDTA) removed 20 to 30% of the thorium from the humate. A mixed organic-hydroxy complex was proposed for the reaction of thorium with humic substances.

Migration Results

Field Studies--

The thorium content of groundwater was reported by Dementyev and Syromyatnikov (1965) to be highest in low salinity, low hardness, low pH, high organic content groundwaters. These characteristics suggest transport of thorium as colloidal suspensions and anionic complexes involving soil acids.

From a fresh granodiorite containing 9.3 ppm thorium and 2.5 ppm uranium, the first stages of weathering resulted in apparent removal of 25% of the thorium and 60% of the uranium (Hansen and Huntington, 1969). An acid leach of the fresh rock removed 90% of the thorium and 60% of the uranium indicating that most of the thorium and uranium are in acid soluble or interstitial materials. After an initial drop in concentration, the total uranium and thorium content of the weathered rock increases by at least a factor of 4 in the uppermost weathered material. Leaching studies showed that thorium was associated with clays formed during weathering and with accessory minerals such as zircon.

Hansen (1970) reported that when freed from minerals by weathering, thorium was leached comparatively slowly. From a fresh granodiorite containing 9.3 ppm thorium, the first stages of weathering resulted in apparent removal of 25% of the thorium. An acid leach of the fresh rock removed 90% of the thorium indicating that most of the thorium was in acid soluble or interstitial materials. The thorium content of the weathered rock increased by a factor of 4 due mostly to association with the clays formed during weathering.

Laboratory Studies--

Desai and Ganguly (1970) showed humic acids from a coastal marine sediment solubilized 100% of the thorium added to an ammonia solution (2.5N). Thorium in this solution without humic acid was observed to predominantly precipitate (95%). The humic acid-thorium complex was noncationic. In an identical experiment, fulvic acid extract was shown to solubilize 59% of thorium added to an ammonia solution. Again, the solubilized organic-thorium fraction was noncationic.

Summary

Under alkaline conditions, $Th(OH)_4$ and ThO_2 maintain low activities in soil solutions (Figures 3-24 and 3-25) and these compounds could form and govern thorium concentration. Thorium hydrolyzes readily even in moderately acidic environments (Figure 3-25) so that Th^{4+} would be present only in very acidic solutions. Laboratory studies also show that thorium tends to precipitate as thorium hydroxide and hydrated thorium oxide in soils (Schulz, 1965; Rancon, 1973).

An increase in thorium content with increase in $CaCO_3$, phosphate and humus content of soils and sediments has been reported (Kuznetsov et al., 1968; Pashneva et al., 1965; Menzel, 1968; Yakobenchuck, 1968; Hansen and Huntington, 1969; Pokidin et al., 1972). However, Tyuryukanova and Kalugina (1971) reported low thorium concentrations in high humus soils (peats and forest podzols) compared with alluvial soils. Thorium adsorption increases with increase in pH (Rancon, 1973) and decrease in soil particle size (Hansen and Huntington, 1969; Hansen, 1970; Nishiwaki et al., 1972; Rubtsov, 1966, 1972; Bondietti, 1974). Strong humic and fulvic acid complexes with thorium occur in the neutral to acidic range (Rancon, 1973; Bondietti, 1974) which are noncationic (Desai and Ganguly, 1970) and mobile. It has been reported also that thorium migrates primarily in the colloidal form (polymeric) in the natural environment (Baranov et al., 1956; Lazarev et al., 1961; Kimura et al., 1968).

References

Adams, J. A. S., J. K. Osmond, and J. J. W. Rogers. 1959. The Geochemistry of Thorium and Uranium. IN: Physics and Chemistry of the Earth. 3. L. H. Ahrens, F. Press, K. Rankama, and S. K. Runcorn (eds.), pp. 298-348.

Adams, J. A. S. and K. A. Richardson. 1960. Thorium, Uranium, and Zirconium Concentrations in Bauxite. Econ. Geol. 55:1653.

Adams, J. A. S. and C. E. Weaver. 1958. Thorium-to-Uranium Ratios as Indicators of Sedimentary Processes -- An Example of Geochemical Facies. Bull. Am. Assoc. Petrol. Geol. 42:387.

Ahrens, L. H. 1952. The Use of Ionization Potentials. Part I. Ionic Radii of the Elements. Geochim. et Cosmochim. Acta. 2:155.

Baes, C. F., Jr. and R. F. Mesmer. 1976. The Hydrolysis of Cations. John Wiley and Sons, New York.

Baranov, V. I., A. B. Ronov, and K. G. Kinashova. 1956. On Geochemistry of Dispersed Thorium and Uranium in Clays and Carbonate Rocks of Russian Platform. Geokhimiya No. 3:3-8.

Billings, G. K. 1962. A Geochemical Investigation of the Valley Spring Gneiss and Packsaddle Schist, Llano Uplift, Texas. Texas J. Sci. 9:328.

Bondietti, E. A. 1974. Adsorption of Pu(IV) and Th(IV) by Soil Colloids. Agronomy Abstracts.

Dementyev, V. S. and N. G. Syromyatnikov. 1965. Mode of Occurrence of Thorium Isotopes in Ground Waters. Geokhimiya. No. 2:211-218.

Desai, M. V. M. and A. K. Ganguly. 1970. Interaction of Trace Elements with Organic Constituents in the Marine Environment. BARC-488, p. 102.

Hansen, R. O. 1970. Radioactivity of a California Terrace Soil. Soil Science. 110:31-36.

Hansen, R. O. and G. L. Huntington. 1969. Thorium Movements in Morainal Soils of the High Sierra, California. Soil Science. 108:257-265.

Heier, K. S. and J. L. Carter. 1964. Uranium, Thorium, and Potassium Contents in Basic Rocks and Their Bearing on the Nature of the Upper Mantle. IN: J.A.S. Adams and W. M. Lowder (eds.). The Natural Radiation Environment, p. 75.

Heier, K. S. and J. J. W. Rogers. 1963. Radiometric Determination of Thorium, Uranium, and Potassium in Basalts and in Two Magmatic Differentiation Series. Geochim. et. Cosmochim. Acta. 27:137.

Holland, H. D. and J. L. Kulp. 1954. The Mechanism of Removal of Thorium and Radium from the Oceans. Geochim. et Cosmochim. Acta. 5:214-224.

Katsurayama, K. 1968. Accumulation of Radioactivity in Rice Fields. Annu. Rep. Res. Reactor Inst. Kyoto University. 1:293-97.

Kimura, Y., H. Morishuma, T. Koga, H. Kawai and Y. Honda. 1968. Studies on the Behavior and Distribution of Radioactive Substances in Coastal and Estuarine Waters. Kinki Daigaku Genshiryaku Kenkyusho Nonpo. 7:21-31 (in Japanese).

Kuznetsov, Y. V., Z. N. Simonyak, A. P. Lisitsyn, and M. S. Frenklikh. 1968. Thorium Isotopes (^{230}Th, ^{232}Th) in the Surface Layer of the Indian Ocean Sediments. Geochem. Int. 5:169-77.

Lazarev, K. F., D. S. Nikolaev, and S. M. Grashchenko. 1961. Concentration of Thorium Isotopes in Sea Water. Radiokhimiya. 3:623-35 (in Russian).

Menzel, R. G. 1968. Uranium, Radium, and Thorium Content in Phosphate Rocks and Their Possible Radiation Hazard. J. Agr. Food Chem. 16:231-34.

Murray, E. G. and J. A. S. Adams. 1958. Amount and Distribution of Thorium, Uranium, and Potassium in Sandstones. Geochim. et. Cosmochim. Acta. 13:260.

Nishiwaki, Y., Y. Honda, Y. Kimura, H. Morishima, T. Koga, Y. Miyaguchi, and H. Kawai. 1972. Behavior and Distribution of Radioactive Substances in Coastal and Estuarine Waters. IN: Radioactive Contamination of the Marine Environment. IAEA-SM-158/11, pp. 177-193.

Pashneva, G. E., T. P. Slavnina, and V. V. Serebrennikov. 1965. Rare Earth and Thorium Content in Soils of Tomsk Region. Izv. Sibirsk. Otd. Akad. Nauk SSSR, No. 4 Ser. Biol.-Med. Nauk. No. 1:48-52 (in Russian).

Pliler, R. 1956. The Distribution of Thorium and Uranium in Sedimentary Rocks and the Oxygen Content of the Precambrian Atmosphere. Thesis. Rice Institute.

Pokidin, V. K., Y. V. Kuznetov, E. A. Prozorovich, and F. A. Asadullaeva. 1972. Radioactivity and Rate of Sediment Formation in the Caspian Sea. Geokhimiya. No. 7:834-43 (in Russian).

Rancon, D. 1973. The Behavior in Underground Environments of Uranium and Thorium Discharged by the Nuclear Industry. IN: Environmental Behavior of Radionuclides Released in the Nuclear Industry. IAEA-SM-172/55, pp. 333-346 (in French).

Rogers, J. J. W. 1964. Statistical Test of the Homogeneity of the Radioactive Components of Granite Rocks. IN: J.A.S. Adams and W. M. Lowder (eds.), The Natural Radiation Environment, p. 51.

Rogers, J. J. W. and J. A. S. Adams. 1974. Thorium Solubilities in Water. Handbook of Geochemistry, K. H. Wedepohl, Ed., Vol. II-4, p. 90-H-1, Springer-Verlag, New York.

Rubtsov, D. M. 1966. Distribution of Thorium in Various Soils. Pochvovedenie. No. 3:55-67 (in Russian).

Rubtsov, D. M. 1972. Thorium and Radium Content in the Silt Fraction of the Podzolic Mountain Soils of Thin Forests. Radioekologicheskie Issledavaniya Prirodnykh Biogeotsenozakh. IN: Verkhovskaya (ed.) Izdatel'stov Nauka, pp. 42-53.

Ryabchikov, D. I. and E. K. Golbraikh. 1969. Analytical Chemistry of Thorium. Translated by A. Aladjem. Ann Arbor-Humphrey Science Publishers.

Schulz, R. K. 1965. Soil Chemistry of Radionuclides. Health Physics. 11:1317-24.

Sillen, L. G. and A. E. Martell. 1964. Stability Constants of Metal-Ion Complexes. Special Publication No. 17. The Chemical Society, London.

Tyuryukanova, E. B. and V. A. Kalugina. 1971. The Behavior of Thorium in Soils. Soviet Journal of Ecology. 2:467-469.

Vinogradov, A. P. 1959. The Geochemistry of Rare and Dispersed Chemical Elements in Soils. Consultants Bureau, Inc.

Yakobenchuk, V. F. 1968. Radioactivity and Chemical Properties of Sod-Podzolic Soils in the Ukrainian Western Polesie. Visn. Sil's' Kogosped. Nauki. 11:45-50 (in Ukranian).

Sasscer, D. S., C. F. Jordan, and J. R. Kline. 1971. Mathematical Model of Tritiated and Stable Water Movement in an Old-Field Ecosystem. IN: Radionuclides in Ecosystems. CONF-710501-P1, pp. 915-923.

Savin, A. M. 1967. Oxygen and Hydrogen Isotope Ratios in Sedimentary Rocks and Minerals. PhD Dissertation, Calif. Inst. of Tech. Pasadena, CA.

Stead, F. W. 1963. Tritium Distribution in Ground Water Around Large Underground Fusion Explosions. Science. 142:1163-1165.

Stewart, G. L. 1967. Fractionation of Tritium and Deuterium in Soil Water. IN: Isotope Techniques in the Hydrologic Cycle, Geophys. Monograph No. 11, 159-168.

Teller, E., W. K. Talley and G. H. Higgins. 1968. The Constructive Uses of Nuclear Explosives. McGraw-Hill, New York.

Wheeler, M. L. and J. L. Warren. 1975. Tritium Containment After Burial of Contaminated Solid Waste. Trans. Am. Nucl. Soc. 22:741.

URANIUM

Natural Soil and Rock Distributions

The range in abundance of uranium in natural rocks is given in Table 3-96. Vinogradov (1959) reported 1 ppm uranium as the average content of uranium in soils. The oxidation of organic matter in black shales tends to precipitate U(IV). Consequently, the black shales usually contain more uranium than red, green or gray shales.

Brief Chemistry

There are 14 known isotopes of uranium from ^{227}U to ^{240}U and one uranium isomer (^{235}U) (Fairbridge, 1972). The half-lives of the uranium isotopes vary from a few minutes to over 4 billion years. Only three of the isotopes occur naturally, and two of these (^{235}U and ^{238}U) are parents of series that end in lead isotopes. The nuclear data on the three natural uranium isotopes are given in Table 3-97. The uranium isotopes of interest in waste disposal are given in Table 3-98. The common oxidation states of uranium are U(III), U(IV), U(V) and U(VI) (Udaltsova, 1963). U(III) is easily oxidized in air to U(IV).

TABLE 3-96. ABUNDANCES OF URANIUM IN NATURAL MATERIALS (ADAMS ET AL., 1959; CLARK ET AL., 1966)

Material	U Concentration Range, ppm	
Igneous Rocks		
Dunites and Peridotites	0.001-0.8	
Gabbro and Diabase	0.3-3.4	Median 0.5
Intermediate (diorite and quartz diorite)	0.1-11.0	Median 1.7
Sialic (granite, syenite, monaozite)	0.15-21.0	Median 3.9
Sedimentary Rocks		
Black shales	3.0-25.0	Median 8.0
Red, gray and green shales	1.2-12.0	Median 3.2
Orthoquartzite	0.2-0.6	Median 0.45
Limestone and dolomite	0.1-9.0	Median 2.2
Bentonite	1.0-21.0	Median 5.0
Bauxite	3.0-27.0	Median 8.0
Halite	0.01-0.02	Median 0.013
Anhydrite	0.25-0.43	Median 0.37
Metamorphic Rocks		
Marble	0.11-0.24	
Slate	1.2-6.1	
Phyllite	1.0-2.7	
Schist	1.8-2.9	
Gneiss	4.5-15.0	
Amphibolite	2.6-4.1	
Granulite	3.2-7.0	

The U(IV) state is fairly stable in aqueous solutions if they are very acidic. Uranium (V) disproportionates to U(IV) and U(VI): $2UO_2^+ + 4H_3O^+ \rightarrow UO_2^{+2} + U^{+4} + 6H_2O$ (Udaltsova, 1963). U(VI), as UO_2^{+2} (Uranyl) at pH ≤ 2.5, is the most

TABLE 3-97. NUCLEAR PROPERTIES OF NATURAL URANIUM ISOTOPES (FAIRBRIDGE, 1972)

Isotopes	% Abundance	Half-Life, yr	Decay Series Parent
^{234}U	0.0056	2.48×10^5	-
^{235}U	0.72	7.13×10^8	4N + 3
^{238}U	99.27	4.51×10^9	4N + 2

TABLE 3-98. URANIUM RADIONUCLIDE DATA (WEAST, 1976)

Isotope	Half-Life	Decay Mode
^{232}U	73.6 years	α, SF
^{233}U	162,000 years	α
^{234}U	247,000 years	α
^{235}U	7.13×10^8 years	α, SF
^{236}U	2.39×10^7 years	α, SF
^{237}U	6.75 days	β^-
^{238}U	4.51×10^9 years	α, SF

stable state of uranium in aerated aqueous solutions (Seaborg and Katz, 1954). At higher pH values, hydrolyzed uranyl ions predominate. Approximately 103 uranium minerals have been confirmed. Uraninite (ideally UO_2) is the primary ore mineral of uranium, but other secondary minerals include carbonates, sulfates, molybdates, phosphates, vanadates, silicates and multiple oxides. Uranyl ions readily form many complexes with anions ordinarily found in soil-water environments such as carbonate, sulfate and fluoride.

Solid Phase and Solution Equilibria

The thermodynamic data for 1) UO_3, Na_2UO_4 and UO_2CO_3 were obtained from Garrels and Christ (1965), 2) $UO_2(OH)_2$, $UO_2(OH)_2H_2O$ and $Na_4UO_2(CO_3)_3$ were obtained from Sillen and Martell (1964), and 3) the remaining species reported in Figure 3-26 were obtained from Palei (1970). The relative stability of several uranium solid phases, in terms of the uranyl ion activity produced by each, is shown in Figure 3-26. Since all the compounds shown in Figure 3-26 are of U(VI), and are plotted as a function of UO_2^{2+}, their curves would not move with changes in oxidation-reduction conditions. U(IV) compounds, such as UO_2, are very soluble in an oxidizing environment and fall outside the boundary of Figure 3-26. With an increase in reducing conditions, U(IV) compounds would become more stable. At a pO_2 of >71, the UO_2 curve would fall just below the $UO_2NH_4PO_4$ curve. U(VI) compounds are stable in an oxidizing environment and U(IV) compounds are stable in a reducing environment. Consistent with the thermodynamic data is the observation that carnotite, a U(VI) mineral is found in the oxidized zones of uranium ore deposits and uraninite, a U(IV) mineral is a primary mineral in reducing ore zones.

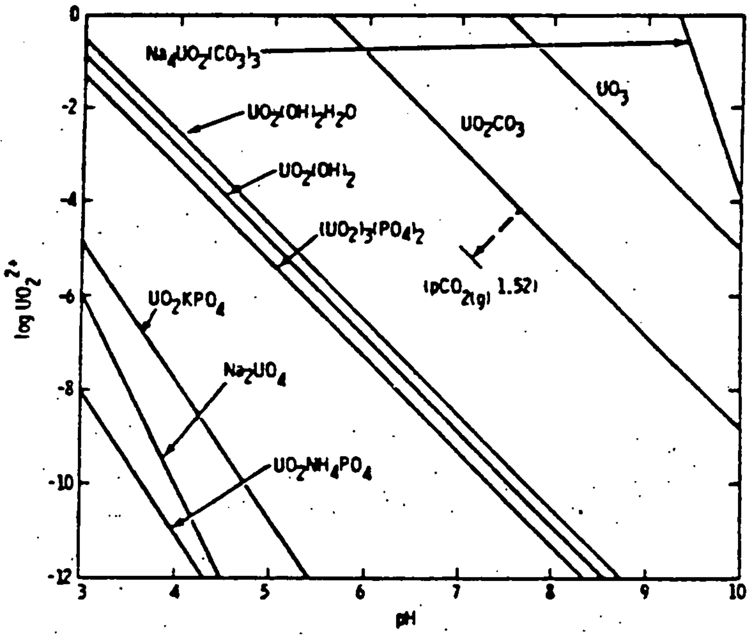


Figure 3-26. The relative stability of various uranium solids in an oxidizing soil environment [$pO_2(g)$ = 0.68 atm], $PCO_2(g)$ = 3.52 atm, pK^+ = pNa^+ = pNH_4^+ = 3.0 and phosphate levels in equilibrium with Variscite and Gibbsite.

The thermodynamic data for UO_2F^+ and $UO_2H_2PO_4^+$ were selected from Palei (1970) and selected data from Sillen and Martell (1964) were used for the remaining species in Figure 3-27. All the species except UO_2^+ shown in Figure 3-27 are of U(VI). Since equilibrium is assumed with Na_2UO_4, a U(VI) compound, a change in oxidation-reduction conditions will affect the position of the curves. In an oxidizing environment (pO_2 = 0.68 atm), the U(IV) species are very low in concentration (log a_i <38) and fall outside of the boundaries of Figure 3-27.

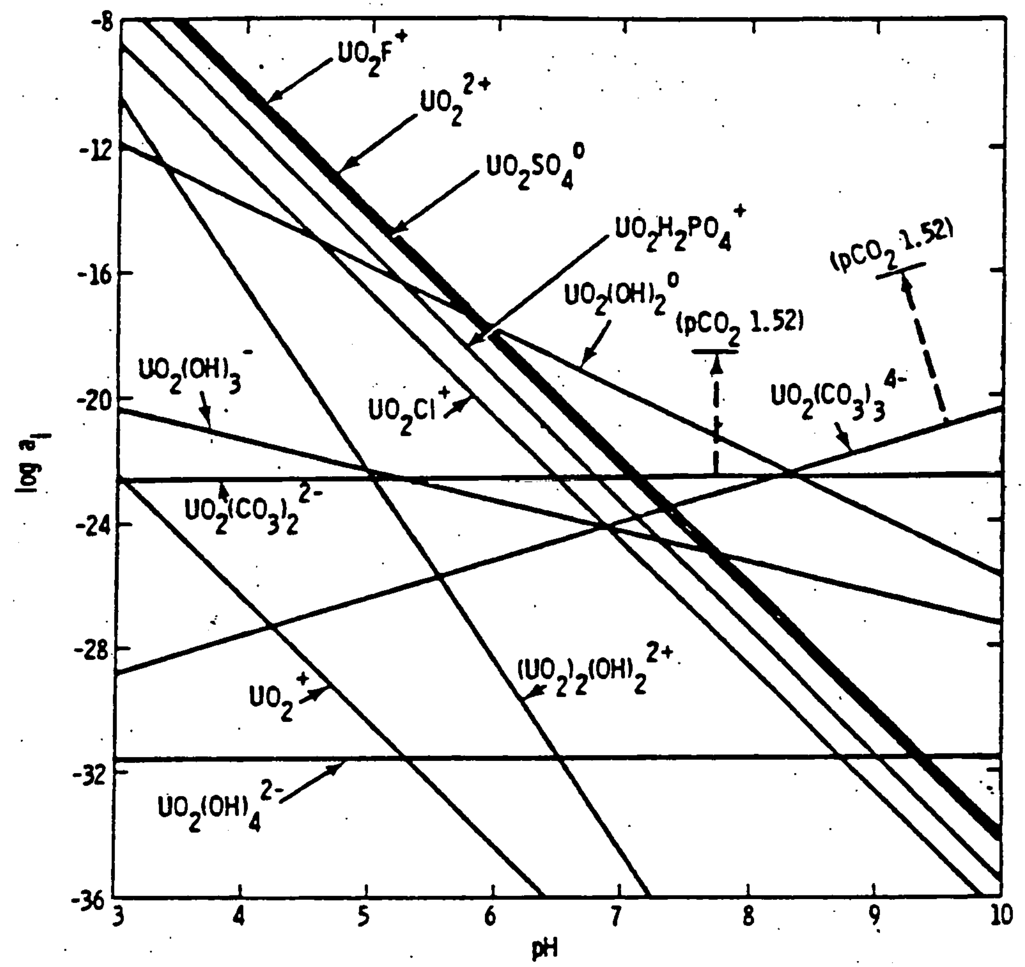


Figure 3-27. Activity of various uranium species in equilibrium with Na_2UO_4 in an oxidizing soil environment [$pO_{2(g)}$ = 0.68 atm], $pCO_{2(g)}$ = 3.52 atm, pCl^- = pSO_4^{2-} = 3.0, pF^- = 4.5 and pH_2PO_4 = 5.0.

Figure 3-27 shows that U(VI) species will control the solution concentration in an oxidizing environment. UO_2^{2+} is the predominant solution species up to a pH of approximately 6. The predominant solution species over pH ranges of from 6 to 8 and >8 are $UO_2(OH)_2^\circ$ and $UO_2(CO_3)_3^{4-}$, respectively.

Experimental Adsorption Results

Uranium adsorption studies have been performed by several investigators. Szalay (1954, 1957) showed that uranium adsorption by decomposing plant debris, peat, lignite and brown coal is quite high. He determined that humic substances in these materials were responsible for the adsorption which was described as an ion exchange-like sorption. Adsorption isotherms for the humic acid were measured.

Goldsztaub and Wey (1955) determined that 7.5 g of uranium was adsorbed from a 1% uranyl nitrate solution per 100 g of calcined montmorillonite and 2.0 g of uranium per 100 g of calcined kaolinite.

Manskaya et al. (1956) described the adsorption of uranium on fulvic acids as a function of pH. The curve of percent adsorption versus pH showed a maximum at pH 6 of about 90% uranium removal. At pH 4 and pH 7, uranium adsorption was down to 30%.

Starik et al. (1958) showed a similar adsorption curve of pH versus uranium adsorption on ferric hydroxide. The maximum of the adsorption curve was at pH 5 with about 50% uranium adsorption, and rapidly decreased above and below pH 5.

Rozhkova et al. (1959) showed similar uranium adsorption curves versus pH for lignite and humic acids. The curve maxima occurred between pH 5 and 6. Dementyev and Syromyatnikov (1968) showed that these adsorption curve maxima result because the pH 6 region is a boundary between anionic and cationic uranium forms and corresponds to: $[UO_2^{+2}] + [UO_2OH^+] \simeq [UO_2(CO_3)_2^{-2}]$, an equality between cation and anion uranium forms in solution.

Horráth (1960) measured the enrichment factor for the adsorption of uranium by peat and obtained an average of 200 to 350. Although it is not possible to determine what methods or exactly what the enrichment factor means from the English translated abstract, it is assumed that the data represent a uranium Kd of 200 to 350 by weight or volume.

Kovalevskii (1967) found the uranium content of noncultivated soils in western Siberia increased with the clay content of the soils. Clay soils contained at least three times as much uranium as sands. Yakobenchuck (1968) correlated the total uranium content in Russian sodpodzilic soils from the Ukraine with other soil constituents. Uranium showed correlation with the oxidizes of silicon, iron, and aluminum suggesting coprecipitation or inclusion.

Masuda and Yamamoto (1971) studied the adsorption of uranium (1 to 100 ppm U) dissolved in water onto volcanic ash, alluvial, and sandy soils. The uranium was almost completely adsorbed on each of the soils. Uranium desorption with salt solutions was extremely difficult especially for the volcanic ash. Similar studies by Yamamoto et al. (1973) on the three soils using uranium (1 to 50 ppm U) and carbonated waters (4 to 109 ppm CO_3^{-2}) showed approximately 100% adsorption and less than 2% desorption.

Rubtsov (1972) determined the uranium content in forest podzolic mountain soils and found a relatively high level of uranium in the <1 μ particle size fraction of the podzolic A_2 horizon. In general, for the soils studied 58% of the total uranium was found in the <1 μ soil fractions. Ritchie, Hawks, and McHenry (1972) found the uranium content of sediments from the Little Tallahatchie River to increase with decreasing particle size.

Rancon (1973) studied the adsorption of uranium using four soils described as follows: 1) a river sediment containing a mixture of quartz, clay, calcite and organic matter, 2) a river peat, 3) a sediment from Cadarache containing a mixture of quartz, clay and calcite with no organic matter, and 4) a soil developed on an altered schist from near LaHague containing a mixture of quartz and clay but no calcite or organic matter. The first two soils were equilibrated with their river waters containing 10 ppm uranium and the last two soils were equilibrated with their respective groundwaters also containing 10 ppm uranium. The resulting uranium distribution coefficients are shown in Table 3-99, which also includes the Kd values on pure quartz, calcite and illite. The clay minerals in Soils 1, 3 and 4 were not identified or the soils further characterized. Rancon also examined the effects of initial uranium concentration on Kd values. Both the uranium concentration and solution pH changed as uranium was added to the solution. At 0.1 mg U/l, the pH was 7.6, for example, and at 1.0 g U/l, the pH was 3.5. Because the pH changes are a function of uranium concentration changes, the results are not easily interpreted. In addition, the Kd concept is invalid above the trace uranium concentration (∿1.0 mg U/l). Uranium adsorption data at 1 ppm versus Kd also are presented. For Soil 4, three peaks were observed: Kd 300 ml/g at about pH 5.5, Kd 2000 ml/g at pH 10 and Kd 270 ml/g at pH 12.

TABLE 3-99. URANIUM Kd VALUES (RANCON, 1973)

Soil	Kd, ml/g
1 - River Sediment (clay, $CaCO_3$, OM)	39
2 - River Peat	33
3 - Sediment (clay, $CaCO_3$)	16
4 - Altered Schist (clay)	270
Quartz	0
Calcite	7
Illite	139

Rancon believed that the adsorption maxima represented by the three peaks also represent electrokinetic potential maxima. Quartz was characterized as inert, calcite was a poor uranium adsorber and clays were the best adsorbers of uranium from solution. Acid, organic-rich soils show much higher uranium Kd values than the alkaline peat (Soil 2) of this study.

Migration Results

Field Studies--

A study of granitic rock weathering by Harriss and Adams (1966) included autoradiographs of fresh and weathered samples of several granitic rocks. There was a definite increase in the density of concentrated radioactive materials with weathering. However, analyses indicated a small loss for uranium. The increased density, therefore, must be due to losses of other materials (alkalies and alkaline earths) during weathering. From a fresh granodiorite containing 2.5 ppm uranium, the initial weathering resulted in losses of 60% of the uranium. An acid leach of fresh rock also removed 60% of the uranium indicating that most of the uranium was in acid soluble or interstitial materials. After an initial drop in concentration, the total uranium content of the weathered rock increased by at least a factor of 4 in the uppermost material.

Laboratory Studies--

Schulz (1965) suggested that uranium may be present in the soil as the divalent uranyl ion, UO_2^{2+}, and will be mobile in soils if present as the uranyl ion.

Masuda and Yamamoto (1971) examined the desorption of uranium from alluvial, sand and volcanic ash soils. The cation exchange capacities were 13.7, 7.7 and 33.0 meq/100 g, respectively, for the alluvial sand and volcanic ash soils. Strong salt solutions and distilled water were used as leachates. Loads of more than 2000 µg U/g of soil were required before desorption by distilled water was 1% of the uranium on the soils. The uranium was amended to the soils as uranyl nitrate before the desorption work. Desorption of uranium was higher with 0.5M $(NH_4)_2SO_4$, 1.34M KCl and 1.44M K_2HPO_4 salt solutions, but reached 50% removal only for the alluvial soil with a high adsorbed uranium content in 1.44M K_2HPO_4 solution. The volcanic ash soil did not attain 5% uranium desorbed in any of the salt solutions.

The effects of carbonate ions on uranium desorption from soils was examined by Yamamoto et al. (1973). They used an alluvial soil, a volcanic ash soil and a sandy soil containing up to 500 µg U/g air-dried soil in their desorption experiments. The desorption of uranium declined as a power function of the amount of uranium on the soil. Desorption results are shown in Table 3-100. Ten grams of soil were magnetically stirred with 100 ml of potassium carbonate solution for 30 min and stood overnight before filtering and a fluorometric uranium analysis. As can be seen in Table 3-92, uranium desorption was very low in the presence of low to moderate environmental carbonate concentrations.

TABLE 3-100. DESORPTION OF URANIUM FROM SOILS WITH DISTILLED WATER AND CARBONATE SOLUTIONS (YAMAMOTO, 1973)

Soil	U Content, µg/g	Distilled Water, %	Carbonate Solution, % 4.3mg CO_3^{-2}/l	43.4mg CO_3^{-2}/l
Alluvial	7.1	0.31	0.62	1.20
	485.0	0.22	0.25	0.41
Sandy	10.1	0.14	0.27	0.90
	488.2	0.12	0.13	0.46
Volcanic Ash	8.3	0.18	0.57	1.15
	500.0	0.09	0.10	0.20

Dall'Aglio et al. (1974) discussed some of the geochemical processes responsible for precipitating secondary uranium minerals. There are a series of processes capable of bringing about a very effective separation and concentration of uranium. The most important process is the attainment of high UO_2^{+2} activity in circulating waters because of low concentration or depletion of CO_3^{-2} which is the most effective uranyl complexing agent. The microorganisms involved were soil microflora and bacteria of mine-waters and granites. Some species were identified. Batch cultures were used to study uranium insolubilization involving biodegradation of uranium complexing organic compounds. The authors suggested that the redeposition of uranium by bacteria may be the origin of some uranium deposits.

Uranium solubilization and insolubilization from granites by heterotrophic bacteria were investigated by Magne et al. (1974). Microbial activity increased the solubilization of uranium from 2 to 97 times by biosyntheses of complexing and chelating compounds.

Grandstaff (1976) arrived at a rate expression for the effects of surface area and uraninite composition, oxygen content of the solution, carbonate content, organics content, pH and system temperature on uraninite dissolution. R, the rate of uraninite (UO_2) dissolution $= \frac{-d(\text{uran})}{dt} = 10^{20.25}\ (SS)(RF)^{-1}$ $(10^{3.38-10.8\ NOC})\ (a\Sigma_{CO_2})\ (D.O.)\ (a_{H+})\ \exp\ (-7045/T)\text{day}^{-1}$, where SS is the specific surface area (cm^2/g), RF is an organic retardation factor, NOC is the mole fraction of nonuranium cations in the uraninite, D.O. is the dissolved oxygen content of the water (ppm), CO_2 is the total dissolved carbonate and T is the absolute temperature. The rate expression was used to predict UO_2 dissolution rates under varying conditions in the absence of organics in the contacting water with good results. Dissolved inorganic species in artificial seawater other than hydrogen ion activity (pH), total carbonate and dissolved oxygen did not appreciably affect the UO_2 dissolution rate. The organic retardation factor must be determined experimentally for each organic-containing solution by comparing the calculated UO_2 dissolution rate without organics in the environment with the observed dissolution rate, or RF = R calculated/R observed. Retardation values of up to 420 were obtained. The magnitude of the affects of the several factors in the expression on the oxidation rate of U(IV) to U(VI) could be of assistance in understanding uranium mobility in soil and rock environments.

Summary

The common oxidation states of uranium are U(III), U(IV), U(V) and U(VI) (Udaltsova, 1963). However, in the geologic environment U(IV) and U(VI) are the most important oxidation states. In oxidizing environments U(VI) compounds such as KUO_2VO_4 (Garrels and Christ, 1965), $UO_2NH_4PO_4$, Na_2UO_4 and UO_2KPO_4 (Figure 3-26) are stable and can precipitate. U(IV) would precipitate as UO_2 in a reducing environment. U(VI) solution species govern uranium concentrations and movement in oxidizing environments (Figure 3-27). Uranium retention by soils and rocks in alkaline conditions is poor because the predominant uranium species at pH > 6 in oxidizing environments (Figure 3-27) are either neutral or negatively charged. An increase in CO_2 pressure in soil solutions reduces uranium adsorption and can increase uranium concentration. The cation exchange properties of soils could contribute to the adsorption of uranium in the neutral to acidic pH range due to the presence of UO_2^{2+}. Oxidation-reduction conditions and pH would be important parameters of uranium mobilization and immobilization.

The above theoretically based predictions are substantiated by experimental results. Uranium has been reported to be solubilized and highly mobile in carbonate-containing waters (Brown and Keller, 1952; Naumov, 1961; Ermolaev et al., 1965; Legin et al., 1966; Haglund, 1968, 1969). Soluble uranium [U(VI)] can:

1. precipitate in the presence of phosphorus as evidenced by a direct correlation of uranium and phosphate content in soils and rocks (Bell, 1960; Sakanoue, 1960; Habashi, 1962; Kuznetsov et al., 1968; Menzel, 1968; Mihalik, 1968), and
2. be adsorbed by the soil organic component and/or reduced to U(IV) followed by precipitation (Breger et al., 1955; Kolodny, 1969; Kolodny and Kaplan, 1970; Baturin, 1971; Dall'Aglio, 1971; Dorta and Rona, 1971; Gabelman, 1971; Baturin and Kochenov, 1973; Mo et al., 1973).

An increase in uranium content with a decrease in soil or sediment particle size was reported by several workers (Kovalevskii, 1967; Mizuno and Mochizuki, 1970; Ritchie et al., 1972).

The uranyl ion can be adsorbed on clay minerals (Goldsztaub and Wey, 1955; Kovalevskii, 1967; Rubtsov, 1972; Ritchie et al., 1972; Rancon, 1973) and other adsorbent materials (Masuda and Yamamoto, 1971; Yamamoto et al., 1973), but also is inclined to form complexes with anions, such as carbonate, that are commonly found in the soil solution (Figure 3-27). Uranyl salts also have been shown to substitute for Ca^{2+} during replacement of calcite by apatite (Ames, 1960), and Ca^{2+} competes with UO_2^{2+} for available sites during ion exchange reactions on inorganics such as calcite (Rancon, 1973) resulting in low uranium Kd values. However, uranyl ion adsorption on organic materials and humic substances is quite high (Szalay, 1954, 1957; Manskaya et al., 1956; Rozhkova et al., 1959; Rancon, 1973) especially at acid pH values. The most important parameters of uranium migration and retention are system Eh and pH.

References

Adams, J. A. S., J. K. Osmond, and J. J. W. Rogers. 1959. The Geochemistry of Uranium and Thorium. Phys. Chem. Earth. 3:298.

Ames, L. L., Jr. 1960. Some Cation Substitutions During the Formation of Phosphorite from Calcite. Economic Geology. 55:354-362.

Baturin, G. N. 1971. Uranium in Solutions of Ooze Deposits in the Southeastern Atlantic. Dokl. Akad. Nauk. SSSR. 198:1186-1188 (in Russian).

Baturin, G. N. and A. V. Kochenov. 1973. Uranium in Interstitial Waters of Marine and Oceanic Sediments. Geokhimiya. No. 10:1529-1536 (in Russian).

Bell, K. G. 1960. Deposition of Uranium in Salt-Pan Basins. U.S.G.S. Professional Paper No. 354-G:161-169.

Breger, I. A., M. Deul, and S. Rubinstein. 1955. Geochemistry and Mineralogy of a Uraniferous Lignite. Econ. Geol. 50:206-226.

Brown, J. H. and W. D. Keller. 1952. Uranium in the Clay of a Black Radioactive Shale. Science. 116:632-633.

Clark, S. P., Jr., Z. E. Peterman and K. S. Heier. 1966. Abundances of Uranium, Thorium, and Potassium. IN: Handbook of Physical Constants. S. P. Clark (ed.). Geol. Soc. Am. Mem. 97.

Dall'Aglio, M. 1971. Study of the Circulation of Uranium in the Supergene Environment in the Italian Alpine Range. Geochim. et Cosmochim. Acta. 35:47-59.

Dall'Aglio, M., R. Gragnani, and E. Locardi. 1974. Geochemical Factors Controlling the Formation of the Secondary Minerals of Uranium. IAEA-SM-183/21. pp. 33-48.

Dement'yev, V. S. and N. G. Syromyatnikov. 1968. Conditions of Formation of a Sorption Barrier to the Migration of Uranium in an Oxidizing Environment. Geochemistry International. 5:394-400.

Dorta, C. C. and E. Rona. 1971. Geochemistry of Uranium in the Cariaco Trench. Bull. Mar. Sci. 21:754-765.

Ermolaev, N. P., A. P. Zhidikora, and V. A. Zarinski. 1965. On the Silicate Form of Uranium Transport in Aqueous Solutions. Geokhimiya No. 7:813-826 (in Russian).

Fairbridge, R. W. (ed.). 1972. The Encyclopedia of Geochemistry and Environmental Sciences. Van Nostrand Reinhold Company.

Gabelman, J. W. 1971. Sedimentology and Uranium Prospecting. Sediment. Geol. 6:145-186.

Garrels, R. M. and C. L. Christ. 1965. Solutions, Minerals, and Equilibria. Harper and Row, New York.

Goldsztaub, S. and R. Wey. 1955. Adsorption of Uranyl Ions by Clays. Bull. Soc. Franc. Mineral. Crist. 78:242.

Grandstaff, D. E. 1976. A Kinetic Study of the Dissolution of Uraninite. Econ. Geol. 71(8):1493-1506.

Habashi, F. 1962. Correlation Between the Uranium Content of Marine Phosphates and Other Rock Constituents. Econ. Geol. 57:1081-1084.

Haglund, D. S. 1968. The Distribution of Uranium in Recent Carbonate Sediments and Skeletons of Organisms and the Effect of Diagenesis on Uranium Redistribution. Rensselaer Polytechnic Institute. Thesis.

Haglund, D. S., G. M. Friedman, and D. S. Miller. 1969. The Effect of Freshwater on the Redistribution of Uranium in Carbonate Sediments. J. Sediment. Petrol. 39:1283-1296.

Harriss, R. C. and J. A. S. Adams. 1966. Geochemical and Mineralogical Studies on the Weathering of Granitic Rocks. Am. J. Science. 264:146-173.

Horráth, E. 1960. Investigations of Uranium Adsorption to Peat in Natural Waters Containing U-Traces. Magyar Tudomanyos Akad. Atommag Kutató Intézete, Közlemenyek. 2:177-183 (in Hungarian).

Kolodny, Y. 1969. Studies in Geochemistry of Uranium and Phosphorites. University of California. Thesis.

Kolodny, Y. and I. R. Kaplan. 1970. Deposition of Uranium in the Sediment and Interstitial Water of an Anoxic Fjord. CONF-700965-1.

Kovalevskii, A. L. 1967. Dependence of the Content of Some Trace Elements on the Clayiness of Soils. Mikroelem. Biosfere Ikh Primen. Sci. Khoz. Med. Sib. Dal'nego Vostoka, Dokl. Sib. Knof., 2nd. 1964. O. V. Makew. Buryat. Khizhn. Izd. Ulan-Ude, USSR.

Kuznetsov, Y. V., Z. N. Simonyak, A. P. Lisitsyn, and M. S. Frenklikh. 1968. Thorium Isotopes (^{230}Th, ^{232}Th) in the Surface Layer of the Indian Ocean Sediments. Geochem. Int. 5:169-177.

Latimer, W. M. 1952. The Oxidation States of the Elements and Their Potentials in Aqueous Solutions. Second Edition. Prentice-Hall, Inc., Englewood Cliffs, NJ.

Legin, V. K., Y. V. Kuznetsov and K. F. Lazarev. 1966. Uranium Occurrence in Marine Sediments. Geokhimiya No. 5:606-608 (in Russian).

Magne, R., J. R. Berthelin, and Y. Dommergues. 1974. Solubilization and Insolubilization of Uranium from Granites by Heterotrophic Bacteria. IN: Formation of Uranium Ore Deposits. IAEA-SM-183/18, p. 73.

Manskaya, S. M., G. V. Drozdora, and M. P. Yemel'yanova. 1956. Fixation of Uranium by Humic Acids and Melanoidins. Geokhimiya. No. 4.

Masuda, K. and T. Yamamoto. 1971. Studies on Environmental Contamination by Uranium. II. Adsorption of Uranium on Soil and Its Desorption. J. Radiat. Res. 12:94-99.

Menzel, R. G. 1968. Uranium, Radium, and Thorium Content in Phosphate Rocks and Their Possible Radiation Hazard. J. Agr. Food Chem. 16:231-34.

Mihalik, P. 1968. Uranium Compounds in the Dominion Reefs, and Their Association with Phosphorus Compounds. NIM-415.

Mizuno, A. and T. Mochizuki. 1970. Distribution of Uranium in the Sediments from San'in Off-Shore, Southwestern Japan. Chishitsu Chosasho Geppo. 21:287-292 (in Japanese).

Mo, T., A. D. Suttle, and W. M. Sackett. 1973. Uranium Concentrations in Marine Sediments. Geochim. et Cosmochim. Acta. 37:35-51.

Naumov, G. B. 1961. Some Physico-Chemical Peculiarities of Uranium Behavior in Hydrothermal Solutions. Geokhimiya. No. 2:115-132 (in Russian).

Palei, P. N. (comp.). 1962. Analytical Chemistry of Uranium. Trans. by N. Kaner. 1963. Israel Program for Scientific Translations Ltd. 421 p.

Racon, D. 1973. The Behavior in Underground Environments of Uranium and Thorium Discharged by the Nuclear Industry. IN: Environmental Behavior of Radionuclides Released in the Nuclear Industry. IAEA-SM-172/55. pp. 333-346 (in French).

Ritchie, J. C., P. H. Hawks, and J. R. McHenry. 1972. Thorium, Uranium, and Potassium in Upper Cretaceous, Pateocene, and Eocene Sediments of the Little Tallahatchie River Watershed in Northern Mississippi. Southeast Geol. 14:221-231.

Rozhkova, Ye.V., Ye. G. Razumnaya, M. B. Serebryakova and O. V. Shchebak. 1959. Role of Sorption in Concentration of Uranium in Sedimentary Rocks. Tr. II. Mezhdunar, knof. po miro nmu ispol'z. atom. energii. 3.

Rubtsov, D. M. 1972. Thorium and Uranium Content in the Clay Fraction of Podzolic Mountain Soils of Thin Forests. Radioekol. Issled Prir. Biogeotsenozakh. pp. 53-66 (in Russian).

Sakanoue, M. 1960. Geochemical Studies on the Radioactive Sediments. II. Uranium Content of Natural Waters from the Ningyo-Pass Mining Area. Nippon Kagakii Zasshi 81:896-898.

Sakanoue, M. 1960. Geochemical Studies on the Radioactive Sediments. III. Uranium, Phosphorus, and Arsenic in the Sedimentary Bed at Ningyo Pass. Nippon Kagaku Zasshi 81:898-902.

Schulz, R. K. 1965. Soil Chemistry of Radionuclides. Health Physics. 11:1317-1324.

Seaborg, G. T. and J. J. Katz (ed.). 1954. The Actinide Elements. McGraw Hill Book Company, Inc.

Sillen, L. G. and A. E. Martell. 1964. Stability Constants of Metal Ion Complexes. Second ed. Spec. Publ. No. 17. The Chemical Society, London.

Starik, I. Ye., F. Ye Starik and A. N. Apollonova. 1958. Adsorption of Traces of Uranium on Iron Hydroxide and Its Desorption by the Carbonate Method. Zh. Neorgan. Khimii. 3(1).

Szalay, A. 1954. The Enrichment of Uranium in Some Brown Coals in Hungary. Acta Geol. Acad. Sci. Hungary. 2:299-311.

Szalay, A. 1957. The Role of Humus in the Geochemical Enrichment of U in Coal and Other Bioliths. Acta Phys. Acad. Sci. Hungary. 8:25-35.

Udaltsova, N. I. 1963. General Information on Uranium. IN: Analytical Chemistry of Uranium. Israel Program for Scientific Translations. pp. 3-7.

Vinogradov, A. P. 1959. The Geochemistry of Rare and Dispersed Chemical Elements in Soils. Consultants Bureau.

Weast, R. C., Editor. 1976. Handbook of Chemistry and Physics. The Chemical Rubber Co., Cleveland, Ohio. pp. B344-B345.

Yakobenchuk, V. F. 1968. Radioactivity and Chemical Properties of Sod-Podzolic Soils in the Ukranian Western Polesie. Visn. Sil's Kogosped. Nauki. 11:45-50 (in Ukranian).

Yamamoto, T., E. Yunoki, M. Yamakawa, and M. Shimizu. 1973. Studies on Environmental Contamination by Uranium. 3. Effects of Carbonate Ion on Uranium Adsorption to and Desorption from Soils. J. Radiat. Res. 14:219-224.

A Proposal for Estimation of Soil Leaching and Leaching Constants for Use in Assessment Models[1]

C. F. BAES, III AND R. D. SHARP[2]

ABSTRACT

A model to predict annual-average, order-of-magnitude leaching constants for solutes in agricultural soils is given. The leaching constants, specific by geographic location and solute, are for use in exposure, dose, or risk assessment models to account for removal from the root zone via leaching in a manner analogous to radiological decay. The model presented here relates annual-average water infiltration (V_w) with soil bulk density (ϱ) and porosity (θ) and the distribution coefficient (K_d) for the solute. Annual infiltration is determined from site-specific estimates of total precipitation (P), irrigation (I), and evapotranspiration (E). A review, discussion, and determination of generic default model input parameters is included. The parameters ϱ and θ vary within a factor of three, and are lognormally distributed with estimated geometric mean values of 1.35 g cm^{-3} and 0.48 cm^3 cm^{-3}, respectively. Geographic distributions of U.S. county-averaged P, I, and E were determined from historical weather station records, the 1974 Agricultural Census, and an evapotranspiration model, respectively. Variability in site-specific annual-average estimates of these parameters is expected to be within a factor of three, also. The most variable and unpredictable parameter is K_d, which may range from one to many orders of magnitude, depending on solute and soil characteristics, including pH. Estimated distributions of K_d for 27 elements in soils $4.5 \leq \text{pH} \leq 9.0$, and correlations between K_d and pH for Cu, Zn, Cd, and Pb are also included. Finally, eight comparisons of model predictions with leaching constants, determined directly from observed data, are made for Pu, Sr, and Tc. In all comparisons, except one study of Pu, the model predicted leaching constants within an order of magnitude of observed values. For Pu the model underpredicted, and for Sr and Tc the model overpredicted leaching rates.

Additional Index Words: **first-order approximation, net infiltration, distribution coefficient, bulk density, volumetric water content, parameter variability.**

Baes, C. F., III, and R. D. Sharp. 1983. A proposal for estimation of soil leaching and leaching constants for use is assessment models. J. Environ. Qual. 12:17–28.

The assessment of human health risks from toxic materials in terrestrial environments invariably includes exposure pathways involving transport through contaminated surface soils (USNRC, 1977; Booth et al., 1971; Fletcher and Dotson, 1971; Moore et al., 1979). In many instances, the relative contribution of these pathways to the total exposure is a function of the material's residence time in surface soils. For example, if a toxic substance is quickly removed from root zone soil, then exposure pathways from ingestion of food produced on contaminated soils may be relatively unimportant. In this case, however, pathways involving drinking water drawn from wells or springs near contaminated soils may be significant if leaching is involved. Thus, inclusion of removal processes such as decay, degradation, or leaching is important to assessment modelers, and to regulators who determine compliance with standards or propose remedial actions to mitigate exposures.

The intent of assessment models such as those contained in the USNRC's Reg. Guide 1.109 (USNRC, 1977) is to predict environmental concentrations and subsequent doses from given releases of radionuclides or pollutants. Because equilibrium conditions and annual perspectives are inherent in model assumptions, assessment models are usually simple multiplicative chains of parameters based on broad generalizations. Such generalizations are acceptable, because the goal of the model is order-of-magnitude, average-condition predictions rather than process-level simulation. Thus, a simple model that will predict the net effect is generally preferable to a more complex formulation which simulates the process, but requires a heavier investment of input parameters, time, and expense to use.

For example, in its present form, the NRC model for plant uptake of radionuclides contained in soil assumes a uniform contamination throughout a 15-cm deep root zone (USNRC, 1977). Plant concentrations are calculated from the soil concentration corrected for nuclide removal via decay. No other removal processes from soil are considered. However, for some relatively mobile nuclides with extremely long half-lives, such as ^{99}Tc, inclusion of additional removal process, such as successive harvesting of crops or leaching below the root zone, would be appropriate for long-term dose assessments (Schwarz and Hoffman, 1980). For assessment purposes, these removal processes may be conceptualized as occurring at constant rates, and the corresponding harvesting and leaching constants may be combined with decay constants in the model.

A satisfactory assessment model that predicts first-order leaching constants will reasonably approximate the net effect of leaching processes on solute concentration in root-zone soil over a period of time with a minimum of user-input requirements. That is, the model should be based on parameters readily and routinely measured, widely available, or well documented in the literature.

The soils literature abounds with solute movement models, which are based on equations describing hydrodynamic dispersion, molecular diffusion (often combined with the former into the term "dispersion"), convection with infiltrating water, and adsorption or exchange of solutes by the soil (Boast, 1973; van Genuchten and Wierenga, 1976; Travis, 1978). Convection-dispersion equations (Crank, 1956; Bear, 1972) have been applied to solute movement, both in steady-state and time-dependent or transient forms. The steady-state convection-dispersion equations may be solved analytically or with relatively simple numerical techniques

---

[1] Research sponsored by the Office of Radiation Programs, U.S. Environmental Protection Agency, under Interagency Agreement EPA-89-F-2-A106 under Union Carbide contract W-7405-eng-26 with the U.S. Department of Energy. Although the research described in this report has been funded wholly or in part by the U.S. Environmental Protection Agency, it has not been subjected to the Agency's required peer and policy review, and therefore does not necessarily reflect the views of the Agency and no official endorsement should be inferred. Received 18 April 1981.

[2] Research scientist, Environmental Sciences Division and programming analyst, Computer Sciences Division, respectively, Oak Ridge National Laboratory, Box X, Oak Ridge, TN 37830.

(van Genuchten and Wierenga, 1974). These models have been shown to perform reasonably well in simulating short-term tritium and 2,4,5-T herbicide movement in hand-packed soil columns (van Genuchten and Wierenga, 1977; van Genuchten et al., 1977) and in field soils (Van de Pol et al., 1977) for simplified water input conditions, and when model paramemters are empirically adjusted or optimized to achieve good agreement between model results and experimental data. Unfortunately, the steady-state convection-dispersion model has been found unsatisfactory for simulation of long-term solute movement under normal field conditions (Omoti and Wild, 1979), and the necessity of adjusting unknown model parameters to achieve short-term fits to experimental data precludes the model's use for general assessments.

Transient- or time-dependent convection-dispersion equations have been developed to avoid simplifications of their steady-state forms and to better simulate normal field conditions. Although the time-dependent form has been shown to better simulate solute movement over long time periods in the field than the steady-state form (Wierenga, 1977), it is also inappropriate for assessment applications. The transient convective-dispersion models must be solved by numerical techniques, which require large computers, much computer time, and, usually, great expense. Also, these models require much more information about parameter values and correlations between parameters than is generally available on a site-specific basis. Furthermore, their ability to predict solute movement in various environments without empirically fitting parameter values has not been demonstrated. Thus, any improvements the transient models have over steady-state models is offset by the effort and expense necessary to use them.

In the absence of an appropriate simple leaching model for assessment applications, the objective of this paper is to (i) present a simple analytical leaching model that predicts order-of-magnitude leaching constants, analogous to radiological decay constants for solutes, (ii) discuss the model input parameters and their variability in agricultural environments, and (iii) compare model predictions with observations of leaching.

THE MODEL

In real systems, processes that render pollutants unavailable for transport through exposure pathways involving soil are neither continuous nor of a first-order nature. However, the traditional assessment modeling approach has been to approximate these soil removal processes by first-order rate constants (Booth et al., 1971; Schwarz and Hoffman, 1980). In such an approach, the soil concentration of a pollutant at a given time (C_t^s) is given by:

$$C_t^s = C_o^s \exp(-\lambda_r t) \qquad [1]$$

where C_o^s = the initial soil concentration, and λ_r = the sum of all first-order removal processes (y^{-1}). Such removal processes may include radiological decay (λ_i), removal via uptake by and harvesting of crops (λ_h), and leaching (λ_l) such that

$$\lambda_r = \lambda_i + \lambda_h + \lambda_l. \qquad [2]$$

Removal constants are derived by approximating observed data with exponential regression equations or by calculating mean residence times of pollutants in soil. The latter approach yields removal constants via the relationship

$$\lambda_r = 1/T_r \qquad [3]$$

where T_r = the mean residence time of the pollutant in soil (y). The mean residence time of a solute being leached from a soil layer may be given by

$$T_l = d/V_s \qquad [4]$$

where d = the depth from which the solute is removed via leaching (cm), and V_s = effective solute migration velocity (cm y^{-1}). The rate of movement of the solute being leached may be related to the infiltration rate of water by a retardation factor (K) such that

$$K = (V_w/\theta)/V_s, \qquad [5]$$

where V_w = the infiltration rate of water in soil (cm y^{-1}), and θ = the volumetric water content of the soil (mL cm^{-3}). The retardation factor is defined as (Hashimoto et al., 1964)

$$K = 1 + \varrho K_d/\theta, \qquad [6]$$

where ϱ = the bulk density of the soil (g cm^{-3}) and K_d = the distribution coefficient for the solute (ml g^{-1}). The distribution coefficient is a partitioning coefficient that assumes a linear relationship between the concentration of a solute in the solid (S) and solution (C) phases. That is,

$$S = K_d C. \qquad [7]$$

Solute removal from solution may be by adsorption, precipitation, etc.

By combining Eq. [3 to 6], the first-order leaching constant is given by

$$\lambda_l = \frac{V_w/\theta}{d(1 + \varrho K_d/\theta)}. \qquad [8]$$

The model represented by Eq. [8] is a relatively simple approximation of an extremely complex process involving physical, chemical, and biological processes. However, within the context of the generalizations and simplifications inherent in currently accepted assessment models (USNRC, 1977), we propose its use as a first approximation of pollutant removal from soil via leaching. The remainder of our discussion will examine the model parameters and their variability in agricultural systems.

NET INFILTRATION

Water infiltration V_w in Eq. [1] is dependent on hydrologic factors other than soil type. The total annual average precipitation provides an estimate of the water applied to a nonirrigated soil. However, irrigation water applied to agricultural land, evaporation, surface runoff, storage in soil, and transpiration through crop plants, all affect the quantity of water percolating downward through agricultural surface soils. Therefore, V_w may be estimated by

$$V_w = P + I - (E + R_s + S_s) \qquad [9]$$

where
P = the total annual precipitation (cm y^{-1}),
I = the total annual irrigation (cm y^{-1}),
E = the total evapotranspiration (surface evaporation and plant transpiration) (cm y^{-1}),
R_s = annual surface runoff (cm y^{-1}), and
S_s = annual average soil storage of water (cm y^{-1}).

In Eq. [9], R_s is water not percolating vertically through surface soil. Surface runoff in this context is not equivalent to annual runoff, as reported by the U.S. Geological Survey, which is measured in terms of fluvial discharge of surface rivers and streams draining large watersheds (Busby, 1963). Annual runoff is comprised of both surface runoff and infiltrating water, which runs into drainage streams (subsurface runoff). Surface runoff is primarily the cause of soil erosion, and occurs mostly when precipitation intensity, duration, and amount are all great (Baver, 1959). Recently, surface runoff and soil erosion have been identified as serious economic problems in the United States (Risser, 1981). However, for simplicity, R_s will be assumed to be negligible.

In a generic sense, surface runoff is difficult to predict because of its dependence on climate, slope, soil composition, vegetative cover, and human intervention. The latter influence is extremely difficult to predict. Here it is assumed that post-1930 farming techniques such as contour plowing, terracing, etc., have made surface runoff negligible. As a first approximation $(R_s + S_s)/P$ is assumed to be negligible, hence Eq. [9] reduces to

$$V_w = P + I - E. \qquad [10]$$

BULK DENSITY

The distribution of soil bulk density (Fig. 1) is based on an analysis of values reported by Free et al. (1940) for 68 soils, and Holtan et al. (1968) for 154 soils. The soils from these references are the surface horizons of pasture or cropland soils from 24 of the continental 48 states, representing all of the nine geographic census regions of the conterminous United States. Soils from idle, waste, municipal, or forest land or horizons below the surface horizon were not included in our analysis. Distributions were derived after the method of Hoffman and Baes (1979). The reported data were best approximated by lognormal distributions.

All bulk density measurements were made using liquid displacement techniques at 0.3 bar moisture tension (Holtan et al., 1968) or at field moisture at the time of sampling (Free et al., 1940). Holtan et al. (1968) included bulk densities determined after oven-drying the soil, but they were not used in this analysis because the 0.3 bar moisture tension method is more comparable to the field moisture method used by Free et al. (1940). The oven-dry-bulk density determinations averaged ~4% greater than the 0.3 bar moisture bulk density determinations.

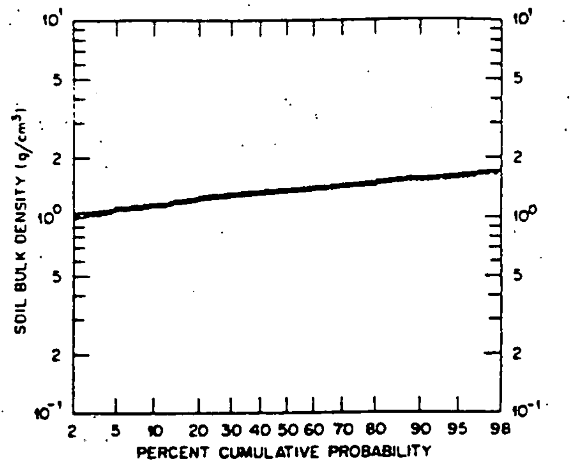


Fig. 1—Lognormal probability plot of soil bulk density based on analysis of 222 agricultural soils in the United States.

The distribution of pasture and cropland soil bulk density may be described by a lognormal distribution, with a mean and standard deviation of the logtransformed values of 0.30 and 0.12, respectively. The median value of the lognormal distribution is 1.35 g cm^{-3}. Estimates of bulk density distributions for silt loams, clays and clay loams, sandy loams, gravelly silt loams, and loams are presented in Table 1.

The variability of ρ, as measured by σ, may be reduced by grouping the soils into five soil types reported by Holtan et al. (1968) and Free et al. (1940); i.e., silt loams, clays and clay loams, sandy loams, gravelly silt loams, and loams. This grouping is an arbitrarily imposed regime on a continuous spectrum of soil characteristics. The variability of the distributions is, therefore, a product of the inherent variability among soils and the arbitrary assignment of them into categories.

There are slight, but distinct, differences among the five soil types with respect to the values of ρ. The ordering is approximately sandy loams > loams > silt loams > clays and clay loams > gravelly silt loams.

VOLUMETRIC WATER CONTENT

The analysis of volumetric water content, θ is based on total soil porosity. The relationship

$$\alpha = 1 - (\rho/2.65) \qquad [11]$$

where α = the total porosity of the soil ($cm^3\ cm^{-3}$), and 2.65 = the particle density (g cm^{-3}) of the soil, was used to derive total porosities of the soils analyzed for bulk density. These values were used to estimate distributions of total porosity for the five soil types (Table 1). Because of the relationship between soil bulk density and total porosity, the ordering of total soil porosity for the five soil types is the inverse of the ordering for bulk density.

Total porosity defines an upper limit of volumetric water content (saturation of all pore space). The cor-

Table 1—Estimated distributions of soil bulk density for five soil types.

Soil type	No. observed	μ†	σ‡	exp(μ) (0.50)§	exp($\mu + \sigma$) (0.84)	exp($\mu + 2\sigma$) (0.98)	Observed range
Bulk density, ρ				g cm^{-3}			
Silt loams	99	0.28	0.11	1.33	1.49	1.67	0.86 to 1.67
Clays & clay loams	49	0.26	0.11	1.30	1.45	1.62	0.94 to 1.54
Sandy loams	37	0.40	0.09	1.50	1.64	1.81	1.25 to 1.76
Gravelly silt loams	15	0.20	0.10	1.22	1.34	1.48	1.02 to 1.42
Loams	22	0.35	0.08	1.42	1.53	1.66	1.16 to 1.58
All soils	222	0.30	0.12	1.35	1.52	1.71	0.86 to 1.76
Total porosity, α¶				cm^3 cm^{-3}			
Silt loams	99	−0.71	0.11	0.493	0.549	0.613	0.370 to 0.675
Clays & clay loams	49	−0.68	0.10	0.506	0.558	0.615	0.419 to 0.645
Sandy loams	37	−0.84	0.12	0.430	0.487	0.551	0.336 to 0.528
Gravelly silt loams	15	−0.62	0.08	0.536	0.580	0.627	0.464 to 0.615
Loams	22	−0.77	0.09	0.462	0.502	0.547	0.411 to 0.562
All soils	222	−0.73	0.12	0.484	0.547	0.617	0.336 to 0.675

† The mean of the logarithms of the observed values.
‡ The standard deviation of the logarithms of the observed values.
§ Values in parentheses correspond to cumulative probability.
¶ Based on $\alpha = 1 - (\rho/2.65)$, where 2.65 g cm^{-3} is the particle density for all soils.

responding lower limit of volumetric water content is zero (complete desiccation). The actual quantity of water in soil at any time will be a function of such environmental and physical influences as rainfall, humidity, sunlight, temperature, atmospheric pressure, vegetative cover, tillage effects, soil structure, texture, and composition. In geographic areas with high annual precipitation, it is probable that an average annual value of θ is near field capacity. In arid or semi-arid regions, an average annual value of θ is likely near or below the wilting point. In such areas, precipitation is usually supplemented with irrigation to maintain optimum soil moisture conditions for agricultural practice.

Wierenga (1977) stated that water contents near field capacity were appropriate for his model. For the model presented here, any θ between field capacity and wilting point is probable for agricultural soils. In the absence of available data, a value midway between field capacity and wilting point appears reasonable. Table 2 presents an analysis of volumetric water contents at field capacity and wilting point for the 154 pasture and cropland soils reported by Holtan et al. (1968). Field capacity θ was taken to be moisture retained at 0.3 bar tension, and wilting point was taken to be moisture retained at 15 bar tension (Marshall, 1959; Richards, 1954; Lyon et al., 1956). From Table 2, water retention is roughly ordered clay loams > silt loams and loams > sandy loams. Midpoint vlaues of θ for silt loams, clays and clay loams, sandy loams, and loams are 0.24, 0.29, 0.15, and 0.23 mL cm^{-3}, respectively. An average value for all the soils combined is 0.23 mL cm^{-3}.

DISTRIBUTION COEFFICIENT

From Eq. [7], the distribution coefficient (K_d) is the ratio of concentration of a substance in soil to its concentration in water. Although easy to calculate from empirical measurement, K_d is the most variable of the model parameters. Because of the inherent uncertainties in estimates of K_d for various materials, a brief discussion of the parameter and its determination is appropriate.

The first source of variability in the parameter is associated with the laboratory methods used to determine K_d. Generally, the two most common techniques for determination of K_d are the column and

Table 2—Estimates of volumetric water content, θ, at field capacity and wilting point for four soil types.

	No. observed	μ†	σ‡	exp(μ) (0.50)§	exp($\mu + \sigma$) (0.84)	exp($\mu + 2\sigma$) (0.98)	Observed range
Field capacity¶							
Silt loams	76	−1.06	0.14	0.345	0.396	0.455	0.243 to 0.454
Clays & clay loams	33	−1.02	0.16	0.360	0.423	0.499	0.255 to 0.448
Sandy loams	24	−1.53	0.27	0.217	0.284	0.372	0.124 to 0.329
Loams	17	−1.14	0.15	0.319	0.371	0.431	0.226 to 0.394
All soils#	154	−1.14	0.24	0.321	0.408	0.519	0.124 to 0.454
Wilting point††							
Silt loams	76	−2.06	0.33	0.127	0.176	0.244	0.060 to 0.297
Clays & clay loams	33	−1.52	0.21	0.218	0.269	0.331	0.145 to 0.325
Sandy loams	24	−2.56	0.44	0.077	0.121	0.188	0.029 to 0.158
Loams	17	−2.03	0.26	0.131	0.170	0.220	0.082 to 0.167
All soils#	154	−2.03	0.45	0.131	0.206	0.324	0.029 to 0.325

† The mean of the logarithms of the observed values.
‡ The standard deviation of the logarithms of the observed values.
§ Values in parentheses correspond to cumulative probability.
¶ Measured at 0.3 bar tension.
# Includes values for four gravelly silt loams.
†† Measured at 15 bar tension.

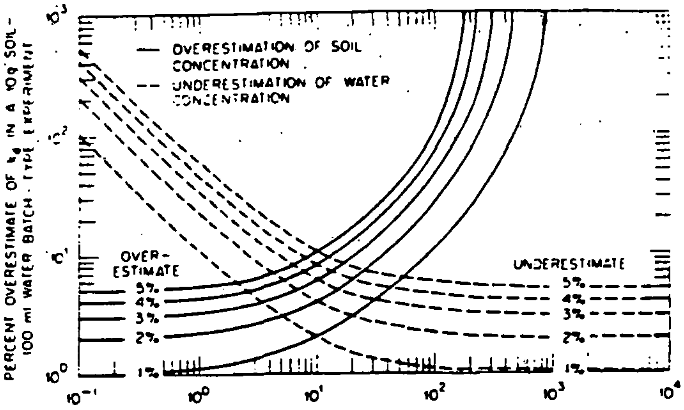


Fig. 2—Percent error in K_d estimation from 1 to 5% overestimates of soil concentration or underestimates of water concentration in a 10 g – 100 mL batch-type K_d experiment.

batch methods, although other methods have been employed to measure distributions of chemical forms (Francis et al., 1977) or distribution among soil fractions (Sheppard et al., 1977). In the column method, a solution of material in water is applied to a column containing uniformly packed soil. The K_d of the material is determined from comparison of the 50% breakthrough curves for the water and material according to the equation

$$\frac{V_s}{V_w} = \frac{\theta}{1 + \rho K_d/\theta}. \qquad [12]$$

In the batch method, soil and water are shaken with the material for a period of time until equilibrium distribution between soil and water is achieved or assumed. Because of nonequilibrium or the influences of convection and diffusion in the column method, these two techniques may give different results for nonionic elemental forms (Inoue and Morisawa, 1976b). Thus, in searching the literature for K_d values, various biases and confounding factors inherent in the laboratory methods used to determine K_d are reflected in the values reported.

A second factor responsible for variation or imprecision in K_d measurement is a result of the parameter being a ratio of two concentrations. A small amount of error in measurement of either the soil or water concentration of material may produce a large amount of error in the resultant ratio. For example, in a batch-type experimental system of 10 g soil, 100 mL H_2O, and 100 μg of material, for which the true K_d is 190 mL g^{-1}, a 1% overestimate of the soil concentration (95.95 μg in soil) yields a K_d of 237 mL g^{-1}, or approximately a 25% overestimate of K_d. The relative error in a K_d estimate from a given percent error in measurement of soil concentration increases rapidly with increasing K_d (Fig. 2). The same is true with a given percent underestimate of the water concentration as the true K_d of the material decreases. Thus, if an investigator measures only one fraction of the soil-water system and determines the concentration of the other fraction by default, significant errors may be introduced into the K_d estimate from very small experimental errors of measurement. This magnification of experimental error undoubtly contributes a significant amount of variability in K_d estimates for materials which are strongly or weakly adsorbed.

Table 3—Estimates of the relationship between K_d and pH in pure clays.

Element	K_d† (ml g^{-1})	Relationship with pH	r^2	Reference
Cu	360	$K_d = 155 + 110 \ln \mathrm{pH}$	0.13	Frost and Griffin, 1977b
Zn	12	$K_d = 0.62 \exp(0.453\ \mathrm{pH})$	0.20	Frost and Griffin, 1977b
Cd	14	$K_d = 0.242 \exp(0.61\ \mathrm{pH})$	0.34	Frost and Griffin, 1977b
Pb	890	$K_d = 6.57 \times 10^{-3} \exp(1.79\ \mathrm{pH})$	0.81	Griffin and Shimp, 1976

† K_d determined at pH = 6.6.

A third source of variability in K_d is its variation with soil type. Soils with different pH, clay content, organic matter content, free iron and manganous oxide contents, or particle size distributions will likely yield different K_d values. For example, in a study by Griffin and Shimp (1976) of Pb absorption by pure clay minerals, pH was shown to be an extremely important determinant of K_d. From their data, an exponential relationship between K_d and pH of the clays can be seen (Table 3). At pH >7.0, Pb K_d is on the order of 10^3, and below this pH, K_d ranges from 10^1 to 10^2. Soil pH has also been shown to influence K_d for Pu and Cm (Relyea and Brown, 1978; Rhodes, 1957a; Nishita, 1978); Ru, Y, Zr, Nb, and Ce (Rhodes, 1957b); As and Se (Bishop and Chisholm, 1962; Frost and Griffin, 1977a); and Mn, Fe, Zn, Co, Cu, Cd, and Ca (Graham, 1973; Reddy and Perkins, 1974; Frost and Griffin, 1977b).

Another source of variation in K_d is the time factor involved with its determination. A batch-type K_d determination is usually made over a period of a few to several hours, until equilibrium is achieved or assumed. If equilibrium is not achieved within this short time period, some error is introduced. Errors from nonequilibrium K_d determination made after 24 h, however, are relatively insignificant (Rhodes, 1957a; Relyea and Brown, 1978; Wildung and Rhodes, 1963). A more significant error may be introduced by using short-term K_d determinations to simulate leaching over time periods of months or years. Gast et al. (1979) found that sorption of ^{99}Tc by low organic soils tended to significantly increase over a 5- to 6-week period. Treatments of the soil with dextrose (H_2O_2) and steam sterilization, and sorption variation with temperature, all indicated that microbiota played either a direct or indirect role in sorption. Heterotrophic bacteria capable of solubilizing PbS, ZnS, CdS, and CnS have been reported by Cole (1979). Microbial influences on the solubility of transuranics has also been suggested by Wildung and Garland (1977). If microbial action is, indeed, important over the long term, then the applicability of K_d experiments carried out with oven-dried and sieved soil to models of leaching in agricultural soils over long time periods must be questioned.

An analysis of the literature was performed to ascertain appropriate distributions of K_d for various elements (Table 4). Because of the variation of K_d with soil pH, the 222 soils analyzed for bulk density and volumetric water content were used to determine a typical range of pH for agricultural soils. In these soils, pH was found to be normally distributed with a mean pH of 6.7, and 95% of the values between a pH of 4.7 to 8.7. Thus, the criterion was adopted of discarding K_d values measured in soils outside of the pH range of 4.5 to 9. The K_d determinations used to generate Table 4 represent a diverse mixture of reported soils, pure clays (pure minerals were excluded), extracting solutions (commonly H_2O, $CaCl_2$ or NaCl), laboratory techniques, and magnification of experimental error. Also, unavoidably, single measurements have been combined with replicates, means, and means of means to derive K_d distributions. When many references have been used to generate the distribution, greater assurance can be given that the distribution is a representative distribution, because it is not heavily biased by one or two experimental designs or techniques. Where a single or a few references were used, less assurance can be given.

On the basis of distributions computed for Cs and Sr (Fig. 3), a lognormal distribution for K_d has been assumed for all elements. If Cs and Sr are typical, then K_d may vary by as much as three orders of magnitude in soils of pH 4.5 to 9. Therefore, the median value [exp(μ)] may not be an appropriate value of K_d for the model in all applications.[1] For example, the median K_d value for Pb (Table 4) is approximately 100 mL g^{-1}. However, based on the relationship between K_d for Pb and clay pH (Table 3) an appropriate K_d for Pb at pH = 6.7 is ~ 1000 mL g^{-1}, which is an order of magnitude higher than the median. Thus, a model user must determine whether a pH-K_d relationship, based on regression analysis of data from a single reference or a distribution based on many references, is more appropriate for an assessment application.

NET INFILTRATION

Site-specific net infiltration estimates are based on total precipitation, irrigation, and estimated evapotranspiration (Eq. 10). Total annual precipitation by county for all states east of a line from the Dakotas through Texas was determined by Olson et al. (1980) from interpolation of 1941-1970 monthly climatic norms for weather stations using the SYMAP program (Dougenik and Sheehan, 1975). Total precipitation by

---

[1] For lognormally distributed parameters with a mean and standard deviation of the logtransformed data of μ and σ, respectively, the median value of the parameter is given by exp(μ) and the mean value by exp($\mu + \sigma^2/2$).

Table 4—Estimates of the distribution of K_d for various elements in agricultural soils and clays of pH 4.5 to 9.0.

Element	No. observed	μ†	σ‡	exp(μ) (0.50)§	Observed range	References
				mL g^{-1}		
Mg	58	1.7	0.52	5.5	1.6 to 13.5	Graham and Silva, 1979; Mokwunye and Melsted, 1973
K	10	1.7	0.49	5.5	2.0 to 9.0	Graham and Silva, 1979
Ca	10	1.4	0.78	4.1	1.2 to 9.8	Graham and Silva, 1979
Cr(II)	15	7.7	1.2	2,200	470 to 150,000	Wildung and Rhodes, 1963
Cr(VI)	18	3.6	2.2	37	1.2 to 1,800	Wildung and Rhodes, 1963; Hajek, 1964; Schmaltz, 1972
Mn	45	5.0	2.7	150	0.2 to 10,000	Graham, 1973; Inoue and Morisawa, 1976a, b; Sims et al., 1979
Fe	30	4.0	1.7	55	1.4 to 1,000	Rancon, 1972; Graham, 1973; Inoue and Morisawa, 1976a, b
Co	57	4.0	2.3	55	0.2 to 3,800	Graham and Killion, 1962; Wildung and Rhodes, 1963; Rancon, 1972; Schmalz, 1972; Graham, 1973; Inoue and Morisawa, 1976a, b
Cu	55	3.1	1.1	22	1.4 to 333	Graham, 1973; Frost and Griffin, 1977b
Zn	146	2.8	1.9	16	0.1 to 8,000	Graham, 1973; Reddy and Perkins, 1974; Inoue and Morisawa, 1976a, b; Frost and Griffin, 1977b
As(III)	19	1.2	0.61	3.3	1.0 to 8.3	Frost and Griffin, 1977a
As(V)	37	1.9	0.52	6.7	1.9 to 18	Frost and Griffin, 1977a
Se(IV)	19	1.0	0.65	2.7	1.2 to 8.6	Frost and Griffin, 1977a
Sr	218	3.3	2.0	27	0.15 to 3,300	Nishita et al., 1956; Klechkovskii, 1957; Rhodes, 1957a, b; McHenry, 1958; Gailledreau, 1963; Jacobs, 1963; Wildung and Rhodes, 1963; Juo and Barber, 1970; Rancon, 1972; Schmalz, 1972; Tamura, 1972; Inoue et al., 1975; Inoue and Morisawa, 1976a, b; Dames and Moore¶
Mo	17	3.0	2.1	20	0.37 to 400	Inoue and Morisawa, 1976b
Tc	24	−3.4	1.1	0.033	0.0029 to 0.28	Gast et al., 1979
Ru	17	5.4	1.0	220	48 to 1,000	Rhodes, 1957b; Wildung and Rhodes, 1963
Ag	16	4.7	1.3	110	10 to 1,000	Inoue and Morisawa, 1976a, b
Cd	28	1.9	0.86	6.7	1.26 to 26.8	Frost and Griffin, 1977b
Cs	135	7.0	1.9	1,110	10 to 52,000	Nishita et al., 1956; Klechkovskii, 1957; Prout, 1958; Jacobs, 1963; Wildung and Rhodes, 1963; Rogowski and Ramura, 1965; Tamura, 1966; Rancon, 1972; Schmalz, 1972; Tamura, 1972; Inoue and Morisawa, 1976a, b; Dames and Moore¶
Ce	16	7.0	1.3	1,100	58 to 6,000	Wildung and Rhodes, 1963; Rhodes, 1957 b
Pb	125	4.6	1.7	99	4.5 to 7,640	Tso, 1970; Griffin and Shimp, 1976
Po	6	6.3	0.65	540	196 to 1,063	Tso, 1970
Th	17	11	1.5	60,000	2,000 to 510,000	Rancon, 1973; Bondietti et al., 1976; Dahlman et al., 1976
U	24	3.8	1.3	45	10.5 to 4,400	Rancon, 197; Bondietti et al., 1976; Dahlman et al., 1976
Np	44	2.4	2.3	11	0.16 to 929	Routson et al., 1975; Dahlman et al., 1976; Sheppard et al., 1977; Nishita et al., 1979
Pu	40	7.5	2.3	1,800	11 to 300,000	Rhodes, 1957a, b; Prout, 1958; Tamura, 1972; Bondietti et al., 1976; Dahlman et al., 1976; Nishita, 1978; Relyea and Brown, 1978; Nishita et al., 1979
Am	46	6.7	3.0	810	1.0 to 47,230	Routson et al., 1975; Nishita and Hamilton, 1977; Sheppard et al., 1977; Nishita et al., 1979
Cm	31	8.1	1.9	3,300	93.3 to 51,900	Sheppard et al., 1977; Nishita, 1978; Nishita et al., 1979

† The mean of the logarithms of the observed values.
‡ The standard deviation of the logarithms of the observed values.
§ Percent cumulative probability.
¶ March 1977 report, Assessment of the Levels, Potential Origins and Transport Routes of the Radioactivity Measured in the Vicinity of the Maxey Flats Low-Level Radioactive Waste Disposal Site. Prepared for the Executive Department for Finance and Administration, Commonwealth of Kentucky.

county for western states was generated by weighted interpolation of 1941-1970 annual climatic norms for the three nearest weather stations to the county centroid. Weighting factors for each weather station were determined using an inverse relationship to distance between weather station and county centroid. A geographical representation of average-annual precipitation by county for the U.S. is shown in Fig. 4.

Total annual irrigation by county (Fig. 5) was determined from the 1974 Census of Agriculture (U.S. De-

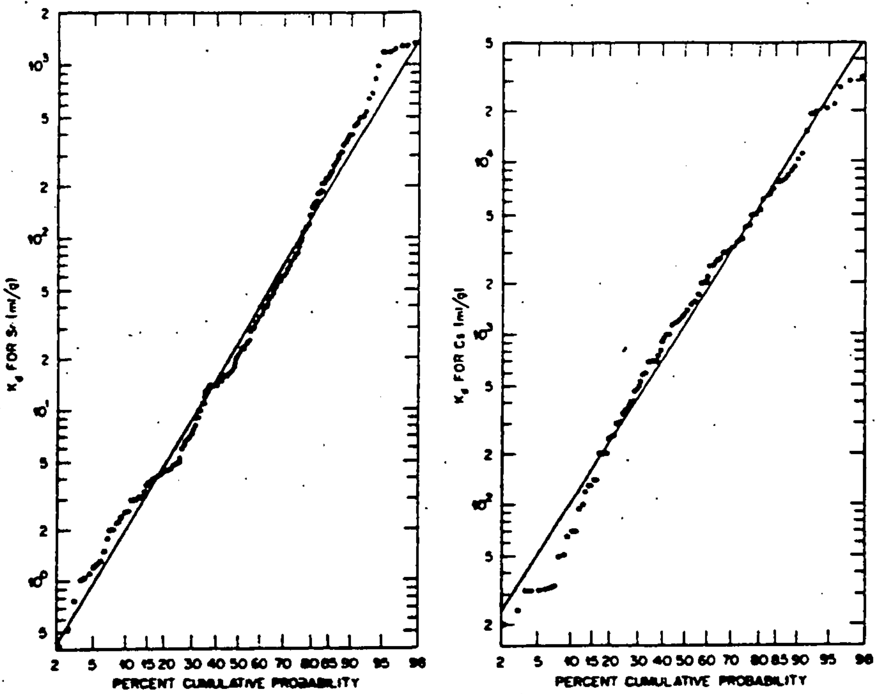


Fig. 3—Lognormal probability plots of K_d for Sr and Cs in soils of pH 4.5 to 9.0.

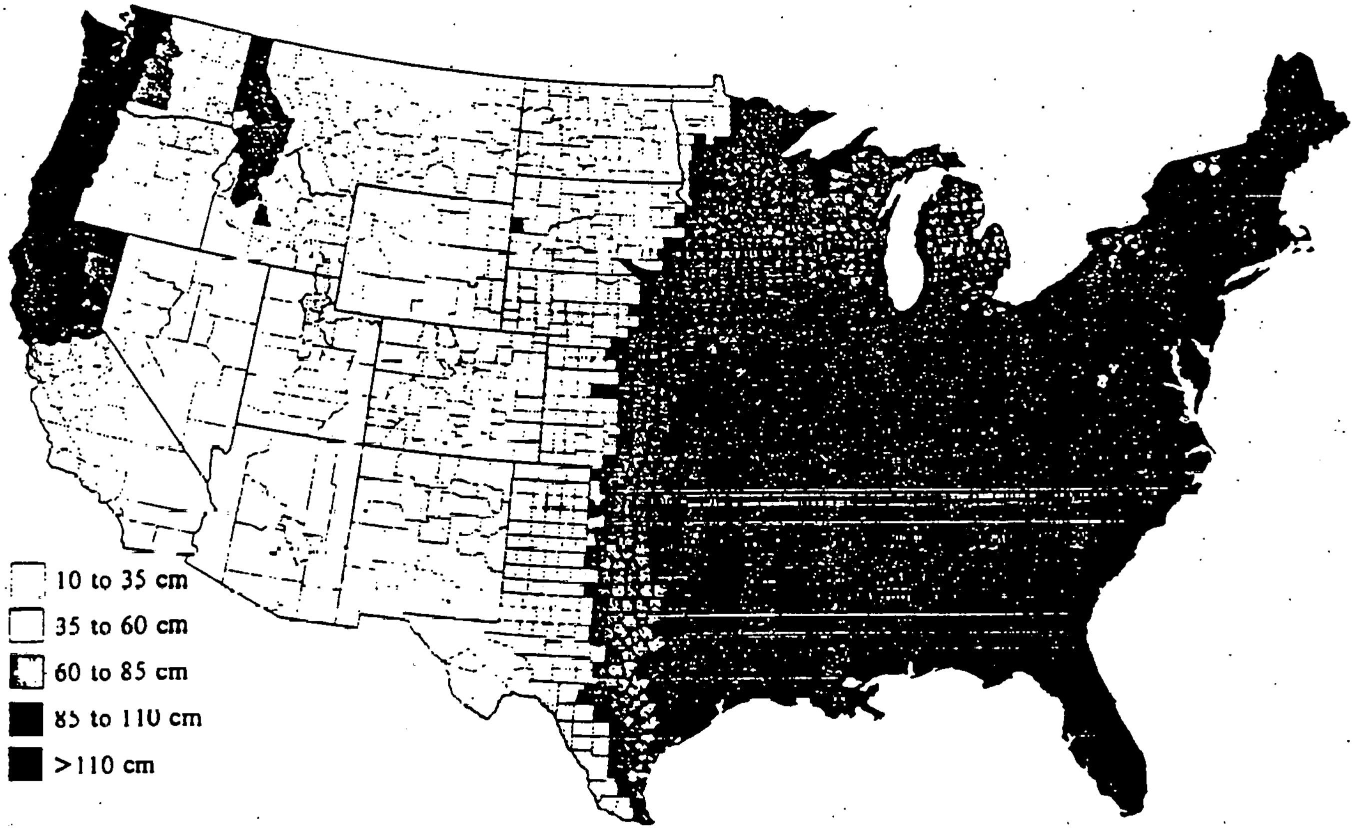


Fig. 4—Geographical distribution of annual average precipitation in cm by county for the conterminous United States.

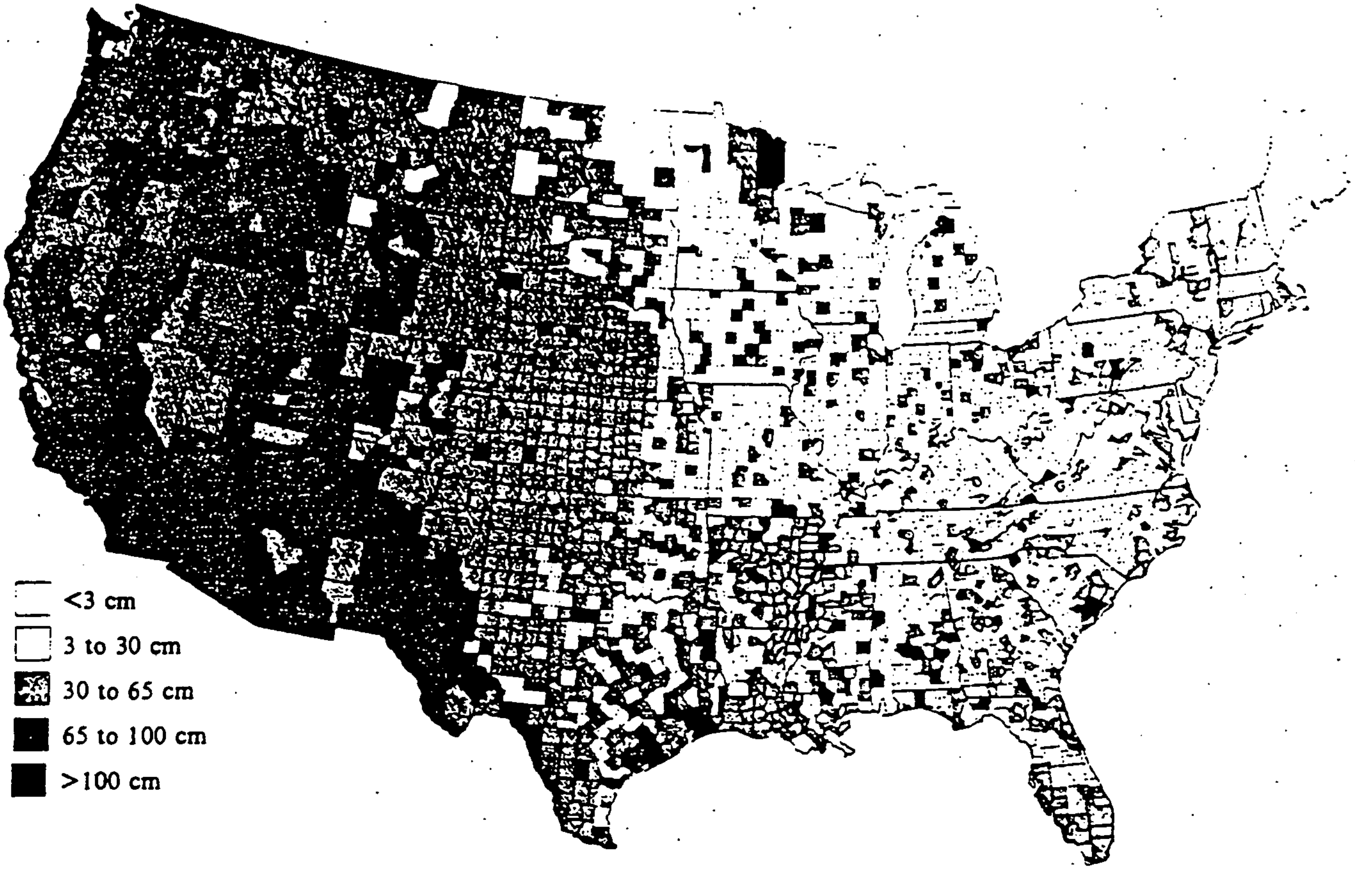


Fig. 5—Geographical distribution of estimated annual average irrigation in cm by county for the conterminous United States.

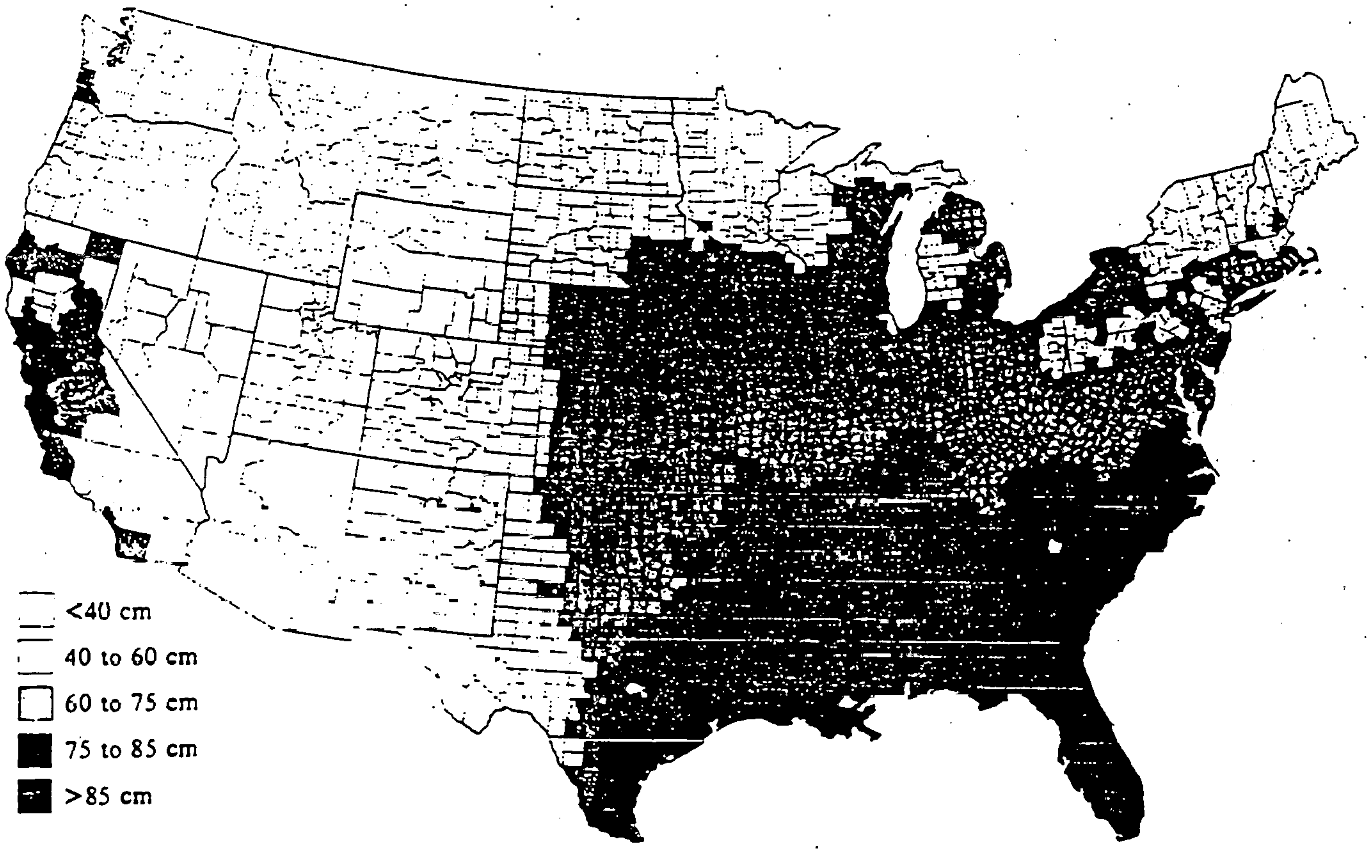


Fig. 6—Geographical distribution of estimated annual average evapotranspiration in cm by county for the conterminous United States.

partment of Commerce, 1977, Table 13) by dividing the "Estimated quantity of irrigation water applied" by the "Total land irrigated." The resulting acre-feet per acre irrigated was converted to centimeters of water. In Fig. 5 the values and corresponding shaded areas are for irrigated land only; and, therefore, caution should be exercised in interpretation of Fig. 5.

Evapotranspiration by county (Fig. 6) was estimated using a model developed by Morton (1978). The model requires as input annual precipitation, and sea level pressure (or altitude), and monthly dew point, ambient air temperatures, and fraction of maximum possible sunshine. These parameters were either taken from Olson et al. (1980) for eastern states or were interpolated from the three nearest weather stations in a manner similar to the precipitation data. Irrigation was not included with precipitation in the model input parameters, although it is considered in Eq. 10. This discrepancy will add a small amount of error to the evapotranspiration by county calculation. Because the Morton model is designed for large land areas and does not provide for local discontinuities, it was assumed that irrigation water is an insignificant fraction of total precipitation over the entire county. This assumption is supported by the observation that nationally only 3–4% of all farmland is irrigated and farmland is approximately 40% of total land area (USDC, 1977). However, if in some U.S. counties irrigated land is a significant fraction of the total land area, then our calculations may be inappropriate. Monthly evapotranspiration estimates were checked for continuity and summed into an annual-average value in centimeter equivalents of water.

According to Morton, the evapotranspiration model has been verified over a wide range of environments and compares satisfactorily with annual precipitation less annual runoff for 81 river basins in Canada, 36 river basins in southern U.S.A., three river basins in Ireland, and two river basins in Kenya. Wallace (1978) compared the model with the Thronthwaite-Mather (1955) and Penman (1948) approaches to modeling arid environments. Morton, however, warns against use of the model near sharp environmental discontinuities. Therefore, estimates of evapotranspiration near coastlines and mountain ranges are suspect.

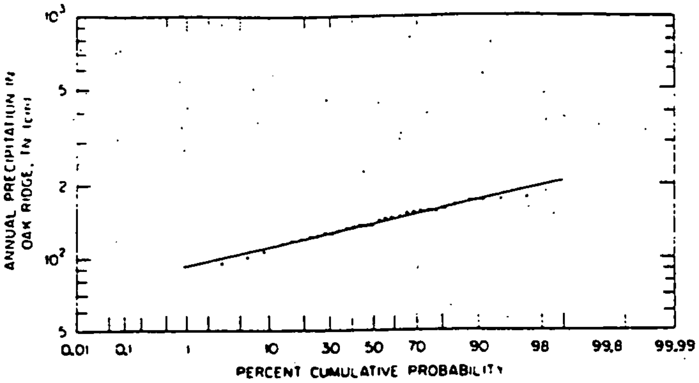


Fig. 7—Lognormal probability plot of annual precipitation in Oak Ridge, Tennessee based on annual normals from 1948 to 1979.

The variability in annual average infiltration by location was estimated by assuming that it is approximately equal to the annual variation in total precipitation. Analysis of annual precipitation rates for Oak Ridge, Tennessee, from 1948 to 1979 (Fig. 7) indicates that total precipitation may be described by a lognormal distribution with a mean and standard deviation of the log-transformed values of 4.93 and 0.17, respectively. If the variability in annual precipitation in Oak Ridge is typical for the continental U.S. and is approximately equal to that for net infiltration, then the variability of model parameters is ordered approximately as $\varrho \equiv \theta < V_w \ll K_d$.

The validity of any model is tested by comparison of model predictions with observed values. Unfortunately, appropriate long-term field studies of leaching are relatively few. Furthermore, most field studies of solute movement through soil are carried out over very short time periods or include various input and removal processes such as atmospheric deposition, biological activity, or soil erosion. For example, in one seven-year follow-up study of lead migration in cropland soil near Urbana, Illinois (Stevenson and Welch, 1979), removal rates can be calculated, but they are largely attributable

Table 5—Comparison of leaching constants predicted by the model with values derived from five studies.

Element†	Depth	Soil texture	V_w	λ_l Observed	λ_l Predicted	P/O‡	Reference
	cm		cm y^{-1}	y^{-1}	y^{-1}		
Pu	5	Sandy loam	40	1.7×10^{-3} (0.78)§	3.0×10^{-5}	1.8×10^{-2}	Jakubick, 1976
Pu	10	Sandy loam	42	4.7×10^{-3}	1.6×10^{-3}	3.4×10^{-1}	Bennett, 1978
Sr	15	Sandy loam	<70	4.4×10^{-3} (0.52)	$< 1.2 \times 10^{-1}$	<27	Squire, 1966
Sr	15	Loam	<70	6.6×10^{-3} (0.55)	$< 1.2 \times 10^{-1}$	<18	Squire, 1966
Sr	15	Clay	<70	1.8×10^{-2} (0.80)	$< 1.3 \times 10^{-1}$	<7.2	Squire, 1966
Sr	5	Sandy loam	<46	7.9×10^{-2} (0.69)	$< 2.3 \times 10^{-1}$	<2.9	Baum, 1977 & 1978
Sr	20	Sandy loam	<46	3.9×10^{-2} (0.53)	$< 5.7 \times 10^{-2}$	<1.5	Baum, 1977 & 1978
Tc	15	Silt loam	64	1.3×10^{0}	1.5×10^{1}	12	Hoffman et al., 1982

† K_d = Median value [exp(μ)] of estimated lognormal distribution (Table 4). ϱ = Median value of estimated lognormal distribution (Table 1), and θ = the average of median values of volumetric water content at field capacity and at wilting point (Table 2).

‡ Predicted-to-observed ratio. A value of 1.0 is perfect a correlation.

§ Values in parantheses are coefficient of determination (r^2) for exponential regression fit [Eq. (1)] of reported data.

to lateral redistribution of lead from successive tillage of the relatively small plots.

Review of available literature indicates only five studies which are appropriate for comparison of model predictions and observed leaching rates (Table 5). This comparison was achieved by deriving an "observed" leaching constant for the soil depth over which measurements through time were taken. A simple exponential regression analysis of the reported concentrations was used to obtain a leaching constant appropriate to describe a first-order removal rate according to Eq. [1]. The derived leaching constants were compared with the model (Table 5) using the estimates of the median values [exp(μ)] of the parameters given in Table 1, 2, and 4.

In Squire's study (1966) of Sr movement in various undisturbed soils over eight years, soils were packed into large concrete cylinders and allowed to remain undisturbed. Strontium profiles and rainfall were recorded approximately every 12 months. Analysis of the soil profiles showed that leaching constants of less than 4.4 $\times$ 10^{-3}, 6.6 $\times$ 10^{-3}, and 1.8 $\times$ 10^{-2} are approximate averages for the top 15 cm of sandy loam, loam and clay soils, respectively. Evapotranspiration at the site is unknown, but the model prediction, using rainfall for V_w and median estimates of K_d from Table 3 shows order of magnitude agreement with observed Sr movement.

In the Federal Republic of Germany yearly ^{90}Sr concentrations have been measured since 1971 in cultivated agricultural soils and summarized by Baum (1977 and 1978). In one agricultural soil the ^{90}Sr content of the 0–5 cm profile has decreased annually from 43% of the total to 26%, with a corresponding λ_l of 7.9 $\times$ 10^{-2} y^{-1}. Using median estimates of model parameters the model overpredicts the derived leaching constant by a factor of about three. In another soil the model overpredicts the derived leaching constant for the 0–20 cm soil depth by about a factor of two.

The leaching constant estimated by Jakubick (1976) for Pu (as PuO_2) migration in the top 5 cm of a sandy loam soil in the vicinity of New York as reported by the USAEC (1974) is 1.7 $\times$ 10^{-2} y^{-1}. Long Island was assumed to be the site of the measurements, and no irrigation was assumed. Although the migration study was carried out over 17 years, the relatively high leaching constant reported for Pu and the low predicted-to-observed ratio suggests either the presence of confounding removal processes or an inappropriate K_d estimate for Pu. Jakubick noted that the PuO_2 form that he studied was roughly 100 times more mobile than the $Pu(NO_3)_4$ form which has a K_d of roughly 5 $\times$ 10^3. Thus, it appears that the K_d value used in Table 5 (1800 ml/g) is too high for PuO_2.

In the study by Bennett (1978) and Hardy (1974) biological activity and other removal mechanisms besides leaching were reported to be minimal because worm holes, evidence of burrowing insects, or cracks from swelling soil were not observed. In 1972, 92.5% of the Pu in the soil at North Eastham, Massachusetts, was in the top 10 cm. In 1976 only 90.8% remained. Using those two measurements, a leaching coefficient of 4.7 $\times$ 10^{-3} y^{-1} was derived and compared to model predictions. The model predicts a removal rate within a factor of three of the observed rate, but caution must be exercised when using an observed value based on only two measurements in time.

In a recent study by Hoffman et al. (1982), $^{95m}TcO_4^-$ was applied to plots of bare soil. Downward movement was observed over a period of 213 days, and a first-order leaching constant of 1.3 y^{-1} was calculated for the top 15 cm of soil. Using estimates of precipitation and evapotranspiration for Oak Ridge, Tennessee, a V_w of 64 cm y^{-1} was combined with ϱ and θ for silt loam soils and a K_d estimate of 0.033 for Tc to predict a leaching constant of 15 y^{-1}. The model overpredicts the actual leaching rate. This overprediction is very likely because of the K_d estimate. Evidence from Gast et al. (1979) and others (O. Hoffman, Jr., private communication) indicates that under long term field conditions Tc K_d is likely to be much greater than the value employed in the model.

These studies notwithstanding, the paucity of long term investigations of solute migration in the field under a diverse range of environments precludes verification of the model. Furthermore, studies of solute distribution coefficients over long time periods in agricultural soils are needed in order to reduce the uncertainty associated with selecting an appropriate estimate of K_d. Until such studies are performed the simple model for solute leaching constants given by Eq. [8] remains unverified, although for Sr the model appears to predict leaching constants within an order of magnitude of observed values. For elements such as Ce, Cs, Pu, and Th, model verification may be difficult because of their suspected immobility in soil and the long time required to observe and quantify leaching.

It is clear that assessment models need to incorporate a quantification of the long-term effects of leaching in their dose and risk calculations. To date most investigations of leaching have addressed short-term dynamics of solute movement in soil without providing appropriate long-term models based on readily available information. The model presented here is an attempt to fill the void. Its merit will be determined only after further verification, beyond that presented here, is made. However, the best approach will be determined only after appropriate long-term field studies are performed to define the environmental and element-specific parameters which best quantify long-term leaching.

ACKNOWLEDGMENTS

The authors wisk to acknowledge C. B. Nelson of the U.S. EPA and R. J. Luxmoore, G. T. Yeh, D. C. Kocher, and F. O. Hoffman of the Oak Ridge National Laboratory for their ideas and help in preparation of this article. Also, we especially wish to recognize E. L. Etnier and Oak Ridge National Laboratory for her contribution in gathering literature references.

LITERATURE CITED

1. Baum, G. R. 1977. Umweltradioaktivitat und Strahlenbelastung. Jahresbericht 1977. Der Bundesminister des Innern. Referat Offentlichkeitsarbeit, Graurheindorfer Strafse 198, 5300 Bonn. Jan. 1980.
2. Baum, G. R. 1978. Umweltradioaktivitat und Strahlenbelastung. Jashresbericht 1978. Der Bundesminister des Innern. Referat Analyse und Information. Offentlichkeitsarbeit, Graurheindorfer Strafse 198, 5300 Bonn, Feb. 1981.

3. Baver, L. D. 1959. Soil physics (3rd ed.). John Wiley and Sons Inc., New York.
4. Bear, J. 1972. Dynamics of fluids in porous media. New York, Elsevier.
5. Bennett, B. G. 1978. Environmental aspects of Americium. U.S. Department of Energy, New York. EML-348.
6. Bishop, R. F., and D. Chisholm. 1962. Arsenic accumulation in Anapolis Valley orchard soils. Can. J. Soil Sci. 42:77-80.
7. Boast, C. W. 1973. Modeling the movement of chemicals in soils by water. Soil Sci. 115:224-230.
8. Bondietti, E. A., S. A. Reynolds, and M. H. Shanks. 1976. Interaction of plutonium with complexing substances in soils and natural waters (IAEA-SM-199/51). *In* Transuranium nuclides in the environment, Proceedings of the symposium held in San Francisco, 17-21 Nov. 1975. International Atomic Energy Agency, Vienna, 1976. p. 273-287.
9. Booth, R. S., S. V. Kaye, and P. S. Rohwer. 1971. A systems analysis methodology for predicting dose to man from a radioactively contaminated terrestrial environment. p. 877-893. *In* D. J. Nelson (ed.) Proc. Third National Symposium on Radioecology. Vol. 1. 10-12 May 1971. Environ. Sci. Div. Oak Ridge Natl. Lab., Oak Ridge, Tenn.
10. Busby, M. W. 1963. Yearly variations in runoff for the conterminous United States, 1931-60. U.S. Geological Survey Water-Supply Paper 1669-S. U.S. Government Printing Office, Washington, D.C.
11. Cole, M. A. 1979. Solubilization of heavy metal sulfides by heterotrophic soil bacteria. Soil Sci. 127:313-317.
12. Crank, J. 1956. The mathematics of diffusion. London, Oxford University Press, London.
13. Dahlman, R. C., E. A. Bondietti, and L. D. Eyman. 1976. Biological pathways and chemical behavior of plutonium and other actinides in the environment. p. 47-80. *In* A. M. Friedman (ed.) Environmental behavior of the actinide elements. American Chemical Society Symposium Series 35.
14. Dougenik, J. A., and D. E. Sheehan. 1975. SYMAP user's reference manual. Laboratory for Computer Graphics and Spatial Analysis, Harvard University, Cambridge, Mass.
15. Fletcher, J. F., and W. L. Dotson. 1971. HERMES—A digital computer code for estimating regional radiological effects from the nuclear power industry. JEDL-TME-71-168. Handford Engineering Laboratory. Richland, Wash. December 1971.
16. Francis, C. W., M. Reeves III, R. S. Fisher, and B. A. Smith. 1977. Soil chromatograph K_d values. p. 403-431. *In* Waste isolation safety assessment program, Task 4, Contractor information meeting proceedings, 20-23 Sept. 1977. Battelle Memorial Institute. Human Affairs Research Centers, Seattle. PNL-SA-6957.
17. Free, G. R., G. M. Browning, and G. W. Musgrave. 1940. Relative infiltration and related physical characteristics of certain soils. U.S. Department of Agriculture Technical Bull. 729. USDA.
18. Frost, R. R., and R. A. Griffin. 1977a. Effect of pH on adsorption of arsenic and selenium from landfill leachate by clay minerals. Soil Sci. Soc. Am. J. 41:53-57.
19. Frost, R. R., and R. A. Griffin. 1977b. Effect of pH on adsorption of copper, zinc, and cadmium from landfill leachate by clay minerals. J. Environ Sci. Health. A12:139-156.
20. Gailledreau, C. 1963. Reactions physico-chemiques lors du mouvement souterrain des radioisotopes. p. 270-275. *In* The International Symposium on the Retention and Migration of Radioactive Ions in Soils, Saclay, France. 16-18 Oct. 1962. Center for Nuclear Studies, Saclay. The University Press of France.
21. Gast, R. G., E. R. Landa, L. J. Thorvig, D. F. Grigal, and J. C. Balogh. 1979. The behavior of Technetium-99 in soils and plants. Dep. of Soil Science, University of Minnesota, St. Paul. COO-2447-6.
22. Graham, E. R. 1973. Selective distribution and labile pools of micronutrient elements as factors affecting plant uptake. Soil Sci. Soc. Am. Proc. 37:70-74.
23. Graham, E. R., and D. D. Killion. 1962. Soil colloids as a factor in the uptake of cobalt, cesium, and strontium by plants. Soil Sci. Soc. Am. Proc. 26:545-547.
24. Graham, E. R., and C. G. Silva. 1979. Labile pools and distribution coefficients for soil calcium, magnesium, and potassium determined with exchange equilibria and radioisotopes. Soil Sci. 128:17-22.
25. Griffin, R. A., and N. F. Shimp. 1976. Effect of pH on exchange-adsorption or precipitation of lead from landfill by clay minerals. Environ. Sci. Technol. 10:1256-1261.
26. Hajek, B. F. 1964. Some adsorption, diffusion, and dispersion characteristics of anions in soils. p. 4.1-4.4. *In* D. W. Pearch and J. K. Green (ed.) Hanford Radiological Sciences Research and Development Annual Report for 1964. BNWL-361V. Battelle-Northwest Laboratories, Richland, Wash.
27. Hardy, E. P. 1974. Depth distributions of global fallout Sr-90, Cs-137, and Pu-239, -240 in sandy loam soil. p. I-2-I-10. *In* E. P. Hardy (ed.) Health and Safety Laboratory, Fallout Program, Quarterly Summary Report. Health and Safety Laboratory, U.S. Atomic Energy Commission, New York, N.Y. HASL-286.
28. Hashimoto, I., K. B. Deshpande, and H. C. Thomas. 1964. Peclet numbers and retardation factors for ion exchange columns. Ind. Eng. Chem. Fundam. 3:213-218.
29. Hoffman, F. O., C. T. Garten, Jr., J. W. Huckabee, and D. M. Lucas. 1982. Interception and retention of technetium by vegetation and soil. J. Environ. Qual. 11:134-141.
30. Hoffman, F. O., and C. F. Baes III. 1979. A statistical analysis of selected parameters for predicting food chain transport and internal dose of radionuclides. Oak Ridge National Laboratory, Oak Ridge, Tenn. ORNL/NUREG/TM-282. October 1979.
31. Holtan, H. N., C. B. England, G. P. Lawless, and G. A. Schumaker. 1968. Moisture-tension data for selected soils on experimental watersheds. USDA, Agricultural Research Service. ARS 41-144.
32. Inoue, Y., and S. Morisawa. 1976a. Migration of radionuclides in a model saturated zone, (I). Nippon Genshiryoku Gakkai Shi 18&42-50.
33. Inoue, Y., and S. Morisawa. 1976b. Distribution coefficient k_d of radionuclide between sample soil and water. Nippon Genshiryoku Gakkai Shi 18:524-534.
34. Inoue, Y., S. Morisawa, and Y. Mahara. 1975. Radionuclide migration in aerated zones, (I). Migration characteristics of nuclides contained in percolating water. Nippon Genshiryoku Gakkai Shi 17:376-384.
35. Jacobs, D. G. 1963. Ion exchange in the deep-well disposal of radioactive wastes. p. 43-54. *In* the International Symposium on the Retention and Migration of Radioactive Ions in Soils, Saclay, France. 16-18 Oct. 1962. Center for Nuclear Studies, Saclay. The University Press of France.
36. Jakubick, A. T. 1976. Migration of plutonium in natural soils (IAEA-SM-199/3). p. 47-62. *In* Transuranium Nuclides in the Environment, Proceedings of the symposium held in San Francisco, 17-21 Nov. 1975. International Atomic Energy Agency, Vienna. 1976.
37. Juo, A. S. R., and S. A. Barber. 1970. The retention of strontium by soils as influenced by pH, organic matter and saturation cations. Soil Sci. 109:143-148.
38. Klechkovskii, V. M. 1957. On the behavior of radioactive fission products in soil, their absorption by plants and their accumulation in crops (Translated from Russian). U.S. Atomic Energy Commission, Washington, D.C.
39. Lyon, T. L., H. O. Buckman, and N. C. Brady. 1956. The nature and properties of soils. (5th ed.). The Macmillan Co., New York.
40. McHenry, J. R. 1958. Ion-exchange properties of strontium in a calcareous soil. Soil Sci. Soc. Am. Proc. 22:514-518.
41. Marshall, T. J. 1959. Relations between water and soil. Tech. Communication #50. Commonwealth Agric. Bureau. Farnham Royal, England.
42. Mokwunye, A. U., and S. W. Melsted. 1973. Magnesium fixation and release in soils of temperate and tropical origins. Soil Sci. 116:349-362.
43. Moore, R. E., C. F. Baes, III, L. M. McDowell-Boyer, A. P. Watson, F. O. Hoffman, J. C. Pleasant, and C. W. Miller. AIRDOS-EPA: A computerized methodology for estimating environmental concentrations and dose to man from airborne releases of radionuclides. ORNL-5532. Oak Ridge National Laboratory. June 1979.
44. Morton, F. I. 1978. Estimating evapotranspiration from potential evaporation: practicality of an iconoclastic approach. J. Hydrol. 38:1-32.
45. Nishita, H. 1978. Extractability of plutonium-238 and curium-242 from a contaminated soil as a function of pH and certain soil components. CH_3COOH-NH_4OH system. p. 403-416. *In* D. C. Adriano, and I. L. Brisbin, Jr. (ed.) Environmental chemistry and cycling processes. Proceedings of a symposium held at

Augusta. Georgia. 28 Apr.-1 May 1976. Technical Information Center. U.S. Dep. of Energy.

46. Nishita. H., and M. Hamilton. 1977. Factors influencing the chemical extractability of ^{241}Am from a contaminated soil. p. 77-96. *In* M. G. White, and P. B. Dunaway (ed.) Transuranics in natural environments. A Symposium at Gatlinburg, Tenn., October 1976. Nevada Applied Ecology Group. U.S. Energy Res. and Dev. Administration, Las Vegas. NVO-178.

47. Nishita. H., B. W. Kowalewsky, A. J. Steen, and K. H. Larson. 1956. Fixation and extractability of fission products contaminating various soils and clays: I. Sr-90, Y-91, Ru-106, Cs-137, and Ce-144. Soil Sci. 81:317-326.

48. Nishita, H., A. Wallace. E. M. Romney, and R. K. Schulz. 1979. Effect of soil type on the extractability of ^{237}Np, ^{239}Pu, ^{241}Am and ^{244}Cm as a function of pH. Laboratory of Nuclear Medicine and Radiation Biology, University of California, Los Angeles. UCLA 12-1192.

49. Olson, R. J., C. J. Emerson, and M. K. Nungesser. 1980. GEO-ECOLOGY: A county-level environmental data base for the conterminous United States. Environmental Sciences Division Publication no. 1537. Oak Ridge National Laboratory, Oak Ridge, Tenn. ORNL/TM-7351.

50. Omoti, U., and A. Wild. 1979. Use of fluorescent dyes to mark the pathways of solute movement through soils under leaching conditions: 2. field experiments. Soil Sci. 128:98-104.

51. Penman, H. L. 1948. Natural evaporation from open water, bare soil and grass. Proc. R. Soc. London, Ser. A, 193:120-145.

52. Prout, N. E. 1958. Adsorption of radioactive wastes by Savannah River plant soil. Soil Sci. 85:13-17.

53. Rancon, D. 1973. Compartment dans les milieux souterrains de l'uranium et du thorium rejetes par l'industrie nucleaire. p. 333-346. *In* C. N. Welsh (ed.) Environmental behavior of radionuclides released in the nuclear industry. Proceedings of a symposium. Aix-en-Provence. 14-18 May 1973. IAEA-SM-172/55. Vienna, Austria.

54. Rancon, D. 1972. Utilisation pratique du coefficient de distribution pour la mesure de la contamination radioactive des mineraux, des roches, du sol et des eaux souterraines. Departement Surete Nucleaire, Centre d'Etudes Nucleaires de Cadarache, Saclay, France. Rapport CEA-R-4274.

55. Reddy, M. R., and H. F. Perkins. 1974. Fixation of zinc by clay minerals. Soil Sci. Soc. Am. Proc. 38:229-231.

56. Relyea, J. F., and D. A. Brown. 1978. Adsorption and diffusion of plutonium in soil. p. 479-495. *In* D. C. Adriano, and I. L. Brisbin, Jr. (ed.) Environmental chemistry and cycling processes. Proceedings of a symposium held at Augusta, Ga. 28 Apr.-1 May 1976. Technical Information Center, U.S. Dep. of Energy.

57. Rhodes, D. W. 1957a. Adsorption of plutonium by soil. Soil Sci. 84:465-471.

58. Rhodes, D. W. 1957b. The effect of pH on the uptake of radioactive isotopes from solution by a soil. Soil Sci. Soc. Am. Proc. 21:389-392.

59. Richards, L. A. 1954. Diagnosis and improvement of saline and alkali soils. U.S. Department of Agriculture Handbook 60. USDA. Washington, D.C.

60. Risser, J. 1981. A renewed threat of soil erosion: it's worse than the Dust Bowl. Smithson. 11:120-131.

61. Rogowski, A. S., and T. Tamura. 1965. Movement of ^{137}Cs by runoff, erosion and infiltration on the alluvial captina silt loam. Health Phys. 11:1333-1340.

62. Routson, R. C., G. Jansen, and A. V. Robinson. 1975. Sorption of ^{99}Tc, ^{237}Np and ^{241}Am on two subsoils from differing weathering intensity areas. Battelle Pacific Northwest Laboratories. Richland, Wash. BNWL-1889.

63. Schmalz, B. L. 1972. Radionuclide distribution in soil mantle of the lithosphere as a consequence of waste disposal at the national reactor testing station. Idaho Operations Office. U.S. Atomic Energy Commission. IDO-10049.

64. Schwarz, G., and F. O. Hoffman. 1980. An examination of the effect in radiological assessments of high soil-plant concentration ratios for harvested vegetation. Health Phys. 39:983-986.

65. Sheppard, J. C., J. A. Kittrick, M. J. Campbell, and T. L. Hardt. 1977. Determination of distribution ratios and diffusion coefficients of Neptunium, Americium and Curium in soil-aquatic environments. Washington. State University. Pullman, Wash. RLO-2221-T-12-3.

66. Sims, J. L., P. Duangpatra, J. H. Ellis, and R. E. Phillips. 1979. Distribution of available manganese in Kentucky soils. Soil Sci. 127:270-274.

67. Squire, H. M. 1966. Long-term studies of strontium-90 in soils and pastures. Rad. Bot. 6:49-67.

68. Stevenson, F. J., and L. F. Welch. 1979. Migration of applied lead in a field soil. Environ. Sci. Technol. 13:1255-1259.

69. Tamura, T. 1966. Development and applications of minerals in radioactive waste disposal. p. 425-439. *In* L. Heller and A. Weiss (ed.) Proceedings of the International Clay Conference. Jerusalem, Israel. Vol. 1. Israel Program for Sci. Translocations, Jerusalem.

70. Tamura, T. 1972. Sorption phenomena significant in radioactive-waste-disposal. p. 318-330. *In* T. D. Cook (ed.) Underground waste management and environmental implications. Proceedings of a symposium 6-9 Dec. 1971. Houston. American Association of Petroleum Geologists and U.S. Geological Surv. Tulsa, Okla.

71. Thronthwaite, C. W., and J. R. Mather. 1955. The water balance. Climatol. 8:1-104.

72. Travis, C. C. 1978. Mathematical description of adsorption and transport of reactive solutes in soil: A review of selected literature. ORNL-5403. Oak Ridge National Laboratory, Oak Ridge, Tenn. October 1978.

73. Tso, T. C. 1970. Limited removal of Po^{210} and Pb^{210} from soil and fertilizer by leaching. Agron. J. 62:663-664.

74. U.S. Atomic Energy Commission. 1974. Environmental statement. Liquid metal fast breeder reactor program. Volume II. WASH-1535. p. 4.G-76-4.G-88. U.S. Gov. Printing Office.

75. U.S. Department of Commerce. 1977. 1974 Census of agriculture. Bureau of the Census, Agriculture Division. Washington, D.C.

76. U.S. Nuclear Regulatory Commission. 1977. Regulatory Guide 1.109. Calculation of annual doses to man from routine releases of reactor effluents for the purpose of evaluating compliance with 10 CFR, Part 50, Appendix I (Revision 1). Office of Standards Development. U.S. NRC, Washington, D.C.

77. Van de Pol, R. M., P. J. Wierenga, and D. R. Nielsen. 1977. Solute movement in a field soil. Soil Sci. Soc. Am. J. 41:10-13.

78. van Genuchten, M. Th., and P. J. Wierenga. 1974. Simulation of solute movement in soils. N. Mex. Agric. Exp. Stn. Bull. 628.

79. van Genuchten, M. Th., and P. J. Wierenga. 1976. Mass transfer studies in sorbing porous media: I. Analytical solutions. Soil Sci. Soc. Am. Proc. 40:473-480.

80. van Genuchten, M.Th., and P. J. Wierenga. 1977. Mass transfer studies in sorbing porous media: II. Experimental evaluation with tritium (^3H_2O). Soil Sci. Soc. Am. J. 41:272-278.

81. van Genuchten, M.Th., P. J. Wierenga, and G. A. O'Connor. 1977. Mass transfer studies in sorbing porous media: III. Experimental evaluation with 2,4,5-T. Soil Sci. Soc. Am. J. 41:278-285.

82. Wallace, R. W. 1978. A comparison of evapotranspiration estimates using DOE Hanford climatological data. Pacific Northwest Laboratory. Richland, Wash. PNL-2698.

83. Wierenga, P. J. 1977. Solute distribution profiles computed with steadystate and transient water movement models. Soil Sci. Soc. Am. J. 41:1050-1055.

84. Wildung, M. W., and D. W. Rhodes. 1963. Removal of radioisotopes from solution by earth materials from eastern Idaho. Phillips Petroleum Co., Atomic Energy Division. National Reactor Testing Station. U.S. Atomic Energy Commission. IDO-14624, November 1963.

85. Wildung, R. E., and J. R. Garland. 1977. The relationship of microbial processes to the fate and behavior of transuranic elements in soils, plants, and animals. Battelle Pacific Northwest Laboratories. PNL-2416. October 1977.

J. Lord

TECHNICAL NOTES

To provide a place within ASCE for publication of technical ideas that have not advanced to the point where they warrant publication as a technical paper in a journal, the publication of technical notes was authorized by the Board of Direction on October 16–18, 1967, under the following guidelines:

1. Five copies of an original manuscript are to be submitted to the Manager, Journals, ASCE, 345 East 47th Street, New York, NY 10017-2398, along with a request by the author that it be considered as a technical note.
2. Four of the copies will be sent to an appropriate technical division or council for review.
3. If the division or council approves the contribution for publication, it will be returned to Society Headquarters with appropriate comments.
4. The journal staff will prepare the material for use in the earliest possible issue of the journal, after proper coordination with the author.
5. Each technical note is not to exceed 2,500 word-equivalents. As an approximation, each full double-spaced manuscript page of text, tables, or figures is the equivalent of 300 words.
6. The technical notes are grouped in a special section of the journal.
7. A 175-word information retrieval abstract and key words are necessary for technical notes.
8. The final date on which a discussion should reach the Society is given as a footnote with each technical note.
9. Technical notes will be included in *Transactions*.
10. Technical notes will be included in ASCE's annual and cumulative subject and author indexes.

The manuscripts for technical notes must meet the following requirements:

1. Titles must have a length not exceeding 70 characters and spaces.
2. The manuscript should be typed double-spaced on one side of 220 mm by 280 mm paper. Five copies of all figures and tables must be included.
3. The author's full name, Society membership grade, and a footnote reference stating present employment must appear on the first page of the paper. Authors need not be Society members.
4. All mathematics must be typewritten and special symbols must be properly identified. The letter symbols used must be defined where they first appear, in figures, tables, or text.
5. Standard definitions and symbols must be used. Reference must be made to the lists published by the American National Standards Institute and to the *ASCE Author's Guide to Journals, Books, and Reference Publications*.
6. Figures must be drawn in black ink on one side of 220 mm by 280 mm paper. Because figures will be reproduced with a width of between 76 mm and 110 mm, the lettering must be large enough to be legible at this width. Photographs must be submitted as glossy prints. Explanations and descriptions must be made within the text for each figure.
7. Tables must be typed double-spaced on one side of 220 mm by 280 mm paper. An explanation of each table must appear in the text.
8. References cited in text must be typed double-spaced at the end of the technical note in alphabetical order in an Appendix. References.
9. Each author may use the International System of Units (SI), and units acceptable in SI, or other units. When SI units are used, no other units are required. When other units are used, the SI units shall be given in parentheses; in a supplementary or dual-unit table; or an appendix.

GROUNDWATER VELOCITY MAGNITUDE IN RADIONUCLIDE TRANSPORT CALCULATIONS

By Daniel J. Goode,[1] Associate Member, ASCE

INTRODUCTION

Analytical solutions have been developed for many conceptual models of solute transport in groundwater (Bear 1979). Although these models usually rely on assumptions too restrictive for accurate description of actual field situations, they are useful in understanding groundwater transport and in evaluating the relative importance of the subsurface processes affecting transport. In addition, these simple models are often used for generic and screening-type analyses of groundwater contamination problems (Kent et al. 1985). For example, the Nuclear Regulatory Commission assesses potential doses resulting from the disposal of very slightly contaminated material in the ground using analytical solutions for one- and two-dimensional groundwater transport (Codell and Schreiber 1979; Codell et al. 1982; Goode et al. 1986). This note presents a method for determining a "worst-case" groundwater velocity value for two conceptual models of decaying radionuclide transport, resulting in maximum calculated point concentration.

For "conservative" screening-type analysis, hydrogeologic properties are typically not known, and assumed parameter values are selected to result in calculated concentrations which are very unlikely to be exceeded in reality. If this type of "conservative" analysis yields performance measures that meet established criteria, then no further information may be required. Otherwise, further site investigation and more realistic analyses can be performed. Parameter selection demands considerable judgment because of the extreme variability of hydrogeologic characteristics from site to site. Fortunately, "worst-case" values can be chosen for some parameters resulting in theoretically maximum or peak calculated performance measure.

Concentrations at a point in an aquifer down gradient from a radionuclide source are affected by, among other processes, dilution in the flowing groundwater and radioactive decay. The travel time of radionuclides from the source to any location determines the extent of radioactive decay. Thus, a higher velocity value results in less decay and higher concentration. On the other hand, groundwater velocity is often considered proportional to flux or specific discharge: $V = q/n$, where q is the specific discharge through the aquifer and n is porosity. The released source mass is diluted by this through-flow, thus a higher velocity value results in more dilution and reduced concentration. These two effects counteract each other and, for certain conceptual models, a groundwater velocity value can

---

[1] Hydrologist, U.S. Nuclear Regulatory Commission; presently with U.S. Geological Survey, 431 Nat. Ctr., Reston, VA 22092.

Note. Discussion open until January 1, 1989. To extend the closing date one month, a written request must be filed with the ASCE Manager of Journals. The manuscript for this paper was submitted for review and possible publication on August 5, 1987. This paper is part of the *Journal of Hydraulic Engineering*, Vol. 114, No. 8, August, 1988. ©ASCE, ISSN 0733-9420/88/0008-0933/$1.00 + $.15 per page. Paper No. 22697-1.

be determined which will result in maximum concentration at a specified location for a given set of parameters. To simplify notation, this velocity value is herein called the maximum point concentration velocity (MPCV). Performance measures other than maximum point concentration, such as radionuclide mass flux, could also be used to develop different "worst-case" velocity values.

A meaningful MPCV exists only if flux is considered proportional to velocity. In many site-specific cases, groundwater flux can be estimated from recharge or pumping estimates, or from observed hydraulic gradients. Because the amount of dilution is then fixed, a higher velocity value (corresponding to lower porosity) results in less decay and higher concentration, and the MPCV is infinitely high. However, in this case, a more appropriate velocity can be estimated from flux and porosity estimates.

Likewise, radionuclides with very long half-lives do not decay significantly, irrespective of the travel time or velocity. For this case, peak concentration is calculated with minimum-volume flow rate and infinitely low velocity.

This note presents a method for determining a velocity value (MPCV) that will result in maximum concentration at a specified location for a solute subject to rapid or moderate first-order decay when dilution is considered proportional to velocity. A closed-form approximation is developed for two-dimensional advective-dispersive transport. This method is only appropriate when no site-specific information on groundwater velocity is available.

ONE-DIMENSIONAL PLUG-FLOW MODEL

A simple conceptualization of radionuclide transport in groundwater is one-dimensional advection (or plug-flow) with linear equilibrium sorption and first-order radioactive decay. This conceptual model ignores dispersion. Initially, concentration is zero for all locations (x). At time $t = 0$, radionuclide mass (measured as activity in curies) is injected at a constant rate M (Ci/T/L) per unit width into the aquifer at $x = 0$. Ahead of the advected front, which is retarded due to sorption, concentration remains zero, because dispersion is ignored. At and behind the advected front, the concentration is:

$$C(t) = C_0 \exp(-\lambda t) \quad \dots \dots \dots \dots (1)$$

or, substituting $C_0 = M/nbV$ and $t = xR_d/V$,

$$C(x) = \frac{M}{nbV} \exp\left(\frac{-x\lambda R_d}{V}\right) \qquad \text{for } x \leq \frac{Vt}{R_d} \quad \dots \dots \dots \dots (2)$$

where C (in Ci/L^3) = radionuclide concentration, in curies (Ci) per unit volume of water; V (in L/T) = uniform groundwater velocity in the x direction; b (L) = aquifer's saturated thickness; n (L^3/L^3) = porosity; R_d (−) = retardation coefficient; and λ (T^{-1}) = radioactive decay rate. The coefficient R_d accounts for sorption of the radionuclide by the aquifer medium and the radionuclides move with an apparent velocity V/R_d (see, e.g., Bear [1979]).

For this case, the sensitivity of Eq. 2 with respect to V is:

TABLE 1. Parameters for Example Problems

Parameter (1)	Symbol (2)	Value (3)
Porosity	n	0.1
Thickness	b	1 m
Source strength	M	1 Ci/yr/m
Retardation	R_d	10
Decay rate	λ	0.021 yr^{-1}
Half-life	—	33 yr

TABLE 2. Calculated Concentrations for One-Dimensional Plug-Flow Model[a]

V (m/yr) (1)	C (x = 100 m) (Ci/m^3) (2)	C (x = 1,000 m) (Ci/m^3) (3)
1	7.58E-9	6.28E-91
2.1	2.16E-5	1.77E-43
10	0.122	7.58E-10
21	0.175	2.16E-5
100	0.081	1.22E-2
210	0.043	1.75E-2
1,000	9.79E-3	8.11E-3
2,100	4.71E-3	4.31E-3

[a] V_1 (x = 100) = $x\lambda R_d$ = 21 m/yr; V_1(x = 1,000) = 210 m/yr.

$$\frac{\partial C}{\partial V} = \left(\frac{x\lambda R_d}{V^2} - \frac{1}{V}\right)C \quad \dots \dots \dots \dots (3)$$

At a local maximum of C, this derivative equals zero. For a nontrivial solution $C \neq 0$ thus, setting Eq. 3 equal to zero and dividing by C yields:

$$V_1 = x\lambda R_d \quad \dots \dots \dots \dots (4)$$

where V_1 = the MPCV for the one-dimensional plug-flow model.

This MPCV depends on the distance from the source at which concentrations are estimated (i.e., the receptor location), the decay rate, and the retardation coefficient. The parameters for an example problem are shown in Table 1. Table 2 illustrates the variation of peak concentration for several velocities at two locations. For this case, an order-of-magnitude underestimate of MPCV results in much lower concentrations than an order-of-magnitude overestimate.

TWO-DIMENSIONAL ADVECTION-DISPERSION MODEL

A common conceptual model of groundwater transport includes dispersion, or spreading of the radionuclide, both along the flow path (longitudinal) and perpendicular to the flow path (transverse). When dispersion is considered, the MPCV is not equal to Eq. 4 because longitudinal dispersion essentially decreases the travel time of the peak, moving some radionuclides faster than the average.

The ratio of MPCV for the two-dimensional advection-dispersion model to the MPCV for the one-dimensional plug-flow model is a function of x/B only (Fig. 1). This term $x/B = x/2\alpha_x$ is one-half the dimensionless distance down gradient from the source relative to the longitudinal dispersivity.

The form of the one-dimensional MPCV with respect to the one-dimensional analytical solution suggests an approximation (V_a) to the MPCV for the two-dimensional advection-dispersion model:

$$V_a = \frac{2B\lambda R_d}{2\left(\frac{B}{x}\right) + \left(\frac{B}{x}\right)^2} \qquad (10)$$

For large x/B, the second term in the denominator in Eq. 10 is small, and V_a reduces to V_1 (Eq. 4) in the limit.

Fig. 1 shows the ratio of V_2 (Eq. 7) to V_1 (Eq. 4) and to V_a (Eq. 10) as a function of dimensionless distance from the source. V_a is a better approximation of V_2 than V_1, because it incorporates some of the affect of dispersion. Recalling that $B = 2\alpha_x$, V_a is about 90% of V_2 for $x = 2\alpha_x$, and V_a is greater than 99% of V_2 for $x = 20\alpha_x$.

The variation of steady-state centerline concentration for different assumed groundwater velocities is shown in Fig. 2. The exact analytical solution (to which Eq. 6 is an approximation, see Wilson and Miller [1978]) is used for a case governed by the parameters in Table 1. In addition, the longitudinal and transverse dispersivities are 20 m and 4 m, respectively. It should be noted from Eq. 8 that MPCV is independent of transverse dispersivity, although the absolute sensitivity (Eq. 7) is a function of α_y through its affect on $\hat{C}$. For the case considered, centerline concentration is more sensitive to velocity for velocities less than MPCV, when decay is most important. The sensitivity is less when velocity is above MPCV and dilution effects become dominant. These characteristics are also shown above in Table 2, for the plug-flow model.

SUMMARY

For certain generic or screening-type models of radionuclide transport in groundwater, a groundwater velocity value (MPCV) can be determined which will result in maximum calculated concentration at a specified location. The MPCV is a function of the distance from the source, the radioactive decay rate, the retardation coefficient, and the longitudinal dispersivity. For the two-dimensional advection-dispersion model considered, the MPCV can be determined by a simple approximation multiplied by the ratio shown in Fig. 1. For many cases, the MPCV for a simple one-dimensional advection model may provide a reasonable estimate of the MPCV for more complex models which can be verified by computing concentrations corresponding to higher and lower velocity values. This method is presented in terms of transport of a decaying radionuclide in groundwater, although it may also be applied to other problems such as surface water transport and transport of organic solutes subject to first-order biodegradation. MPCVs could also be developed for other conceptual models including three-dimensional transport, different source configurations, or additional geochemical processes. This method is only appropriate when no site-specific information on groundwater velocity is available.

ACKNOWLEDGMENT

The contents of this note represent the opinion of the writer and do not necessarily represent the official policy of the U.S. Nuclear Regulatory Commission.

APPENDIX. REFERENCES

Bear, J. (1979). *Hydraulics of groundwater*. McGraw-Hill Book Co., New York, N.Y.

Codell, R. B., Key, K .T., and Whelan, G. (1982). "A collection of mathematical models for dispersion in surface water and groundwater." *Report NUREG-0868 (NTIS)*, U.S. Nuclear Regulatory Commission, Washington, D.C.

Codell, R. B., and Schreiber, D. L. (1979). "NRC models for evaluating the transport of radionuclides in groundwater." *Management of low-level radioactive waste*, M. W. Carter, et al., eds., Pergamon Press, New York, N.Y., 1193–1212.

Goode, D. J., et al. (1986). "Onsite disposal of radioactive waste, vol. 3, estimating potential groundwater contamination." *Report NUREG-1101 (NTIS)*, U.S. Nuclear Regulatory Commission, Washington, D.C.

Kent, D. C., Pettyjohn, W. A., and Prickett, T. A. (1985). "Analytical methods for the prediction of leachate plume migration." *Ground Water Monitoring Rev.*, 5(1), 46–58.

Wilson, J. L., and Dettinger, M. D. (1979). "Tables of the 'sensitivity' integral." *Technical Note No. 21*, Ralph M. Parsons Lab., Dept. of Civil Eng., Massachusetts Inst. of Technology, Cambridge, Mass.

Wilson, J. L., and Miller, P. J. (1978). "Two-dimensional plume in uniform groundwater flow." *J. Hydr. Div.*, ASCE, 104(4), 503–514.

Wilson, J. L., and Miller, P. J. (1979). Closure to "Two-dimensional plume in uniform groundwater flow." *J. Hydr. Div.*, ASCE, 105(12), 1567–1570.

PLUNGING AND STREAMING FLOWS IN POOL AND WEIR FISHWAYS

By Nallamuthu Rajaratnam[1] and Christos Katopodis,[2] Members, ASCE, and Arbind Mainali[3]

INTRODUCTION

A pool-and-weir fishway consists of a number of pools formed by a series of weirs. Water flows from the headwater side to the tailwater region

---

[1] Prof., Dept. of Civ. Engrg., Univ. of Alberta, Edmonton, Alberta, Canada, T6G 2G7.

[2] Habitat Protection Engr., Dept. of Fisheries and Oceans, Winnipeg, Manitoba, Canada, R3T 2N6.

[3] Grad. Student, Dept. of Civ. Engrg., Univ. of Alberta, Edmonton, Alberta, Canada, T6G 2G7.

Note. Discussion open until January 1, 1989. To extend the closing date one month, a written request must be filed with the ASCE Manager of Journals. The manuscript for this paper was submitted for review and possible publication on May 11, 1987. This paper is part of the *Journal of Hydraulic Engineering*, Vol. 114, No. 8, August, 1988. ©ASCE, ISSN 0733-9420/88/0008-0939/$1.00 + $.15 per page. Paper No. 22697-2.

DRAFT 004583 *12/06/91*

DATA COLLECTION HANDBOOK FOR ESTABLISHING RESIDUAL RADIOACTIVE MATERIAL GUIDELINES

by

C. Yu, C. Loureiro, J.-J. Cheng, Y.P. Chia, and L.G. Jones

Environmental Assessment and Information Sciences Division
Argonne National Laboratory, Argonne, Illinois

December 1991

work sponsored by

U.S. Department of Energy
Assistant Secretary for Environment, Safety, and Health and
Assistant Secretary for Nuclear Energy,
Office of Environmental Guidance and
Office of Environmental Restoration and Waste Management

CONTENTS

CONTENTS (Cont'd)

CONTENTS (Cont'd)

CONTENTS (Cont'd)

CONTENTS (Cont'd)

CONTENTS (Cont'd)

CONTENTS (Cont'd)

TABLES

TABLES (Cont'd)

FIGURES

NOTATION

The following is a list of acronyms, initialisms, and abbreviations (including units of measure) used in this document.

ACRONYMS, INITIALISMS, AND ABBREVIATIONS

DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
ICRP	International Commission on Radiological Protection
LLD	lower limit of detection
NCI	National Cancer Institute
NAS	National Academy of Sciences
NMFS	National Marine Fisheries Service
NRC	U.S. Nuclear Regulatory Commission
SI	International System of Units
REV	representative elementary volume
USDA	U.S. Department of Agriculture

UNITS OF MEASURE

°C	degree(s) Celsius
cm	centimeter(s)
cm^3	cubic centimeter(s)
d	day(s)
g	gram(s)
gal	gallon(s)
h	hour(s)
in.	inch(s)

keV	kiloelectron volt(s)
kg	kilogram(s)
km^2	square kilometer(s)
L	length
l	liter(s)
lb	pound(s)
M	mass
m	meter(s)
m^2	square meter(s)
m^3	cubic meter(s)
mi^2	square mile(s)
ml	milliliter(s)
mrem	millirem(s)
pCi	picocurie(s)
s	second(s)
T	time
ug	microgram(s)
yr	year(s)

DATA COLLECTION HANDBOOK FOR ESTABLISHING RESIDUAL RADIOACTIVE MATERIAL GUIDELINES

by

C. Yu, C. Loureiro, J.-J. Cheng, Y.P. Chia, and L.G. Jones

ABSTRACT

A pathway analysis computer code named RESRAD has been developed for implementing of the U.S. Department of Energy's Residual Radioactive Material Guidelines. The parameters used in the RESRAD code include hydrogeological, meteorological, geochemical, geometrical (size, area, depth, etc.), and material related (soil, concrete). The parameter definition, typical range, variation, measurement methodology, and input screen location are discussed in this handbook.

1 INTRODUCTION

In supporting the U.S. Department of Energy (DOE) order establishing residual radioactive material guidelines (DOE Order 5400.5), Argonne National Laboratory has developed a computer program names RESAD (Gilbert et al. 1989). The program and its manual have been used by DOE and other agencies and their contractors for derivation of cleanup criteria and for dose calculations. Since its first release in June 1989, many new features and pathways have been added to the RESRAD code. (The manual is currently under revision.) The purpose of this handbook is to provide guidance to RESRAD users on gathering, evaluating, and selecting, input data for the RESRAD code.

A sensitivity analysis on RESRAD parameters has been conducted; the results are presented in a report entitled RESRAD Parameter Sensitivity Analysis (Cheng et al. 1991).

Parameter sensitivities for a generic run are presented in the Sensitivity Analysis report. A built-in sensitivity analysis capability has been added to the RESRAD code. This capability provides an easy way for users to study RESRAD parameter sensitivity. Users are referred to the revised manual and the RESRAD Parameter Sensitivity Analysis report for description and use of this enhanced feature of RESRAD code.

A total of 51 parameters are discussed in this report. Table 1.1 lists the applicable pathways and the data input screen locations for RESRAD input parameters.

For each parameter, this handbook discusses its definition, typical range, the default value used in RESRAD, relation (if any) to other parameters, and the measurement methodology, if they can be measured. The intent of this handbook is to provide users with a better understanding of each input parameter regarding its typical range, variation, and how the parameter is used in RESRAD code.

TABLE 1.1 Applicable Pathways and Data Input Screen Locations for RESRAD Input Parameters[a]

Parameter	Pathways									Input Screen
	External Gamma	Inhalation	Plant Ingestion	Meat Ingestion	Milk Ingestion	Aquatic Foods	Drinking Water	Radon	Soil Ingestion	
Soil density										
-cover material	used	—	—	—	—	—	—	used	—	R013
-contaminated zone	used	used	used	used	used	used	used	used	used	R013
-unsaturated zone	—	—	used	used	used	used	used	used	—	R015
-saturated zone	—	—	used	used	used	used	used	used	—	R014
-building foundation material	—	—	—	—	—	—	—	used	—	R021
Total porosity										
-cover material	—	—	—	—	—	—	—	used	—	R021
-contaminated zone	used	used	used	used	used	used	used	used	used	R013
-unsaturated zone	—	—	used	used	used	used	used	used	—	R015
-saturated zone	—	—	used	used	used	used	used	used	—	R014
-building foundation material	—	—	—	—	—	—	—	used	—	R021
Effective porosity										
-contaminated zone	used	used	used	used	used	used	used	—	used	R013
-saturated zone	—	—	used	used	used	used	used	—	—	R014
-unsaturated zone	—	—	used	used	used	used	used	—	—	R015
Hydraulic conductivity										
-contaminated zone	used	used	used	used	used	used	used	used	used	R013
-unsaturated zone	—	—	used	used	used	used	used	—	—	R015
-saturated zone	—	—	used	used	used	used	used	—	—	R014
Volumetric water content										
-cover material	—	—	—	—	—	—	—	used	—	R021
-building foundation material	—	—	—	—	—	—	—	used	—	R021
Effective radon diffusion coefficient										
-cover material	—	—	—	—	—	—	—	used	—	R021
-contaminated zone	—	—	—	—	—	—	—	used	—	R021
-building foundation material	—	—	—	—	—	—	—	used	—	R021
Radon emanation coefficient	—	—	—	—	—	—	—	used	—	R021
Precipitation rate	used	used	used	used	used	used	used	used	used	R013
Runoff coefficient	used	used	used	used	used	used	used	used	used	R013

TABLE 2.1 Typical Values of Dry Density of Various Soil Types and Concrete

Soil Type	Dry Density, ρ_d (g/cm^3)
Sand	1.52
Sandy loam	1.44
Loam	1.36
Silt loam	1.28
Clay loam	1.28
Clay	1.20
Concrete	2.40

Sources: Linsley et al. 1982 and Poffijn 1988.

2.1.3 Total (Wet) Density

The total, or wet, density of soil is the ratio of the total mass of soil to its total volume and can be defined as follows:

$$\rho_t = \frac{M_t}{V_t} = \frac{M_s+M_l}{V_s+V_l+V_g} \tag{2.3}$$

Total density differs from dry density in that it is strongly dependent on the moisture content of the soil. For a dry soil, total density approximates the value of dry density.

2.2 MEASUREMENT METHODOLOGY

For use in RESRAD, only the dry densities of five distinct materials (cover layer, contaminated zone, unsaturated and saturated zones, and building foundation material) are needed as input parameters. However, because information on both soil particle and total (i.e., wet) density is required for the calculation of total porosity of the soil material, descriptions of the techniques and procedures for measuring both types of densities follow.

2.2.1 Soil Particle Density Measurement

The soil particle density of a soil sample is calculated on the basis of the measurement of two quantities: (1) M_s, the mass of the solid phase of the sample (dried mass) and (2) V_s, the volume of the solid phase (Blake 1986). Assuming that water is the only volatile in a soil sample, the mass (M_s) can be obtained by drying the sample (usually at 105°C) until it reaches a constant weight, W_s. This method may not be valid for organic soils or soils with asphalt.

The solid phase volume, V_s, can be measured in different ways. One way is to measure the volume directly by observing the resulting increase in the volume of water as the sample of dried soil is introduced into a graduated flask that initially contains pure water (or another liquid). After making sure that the soil/water mixture is free from air bubbles,

M_g = the mass of gases (negligible compared with the masses of the water and matrix phases), and

M_t = the total mass.

Similarly, within the REV, the volumes associated with the soil phases can be defined as follows:

V_s = the volume of solids;

V_l = the volume of liquids;

V_g = the volume of gases;

V_p = $V_l + V_g$, the volume of pore space; and

V_t = $V_s + V_l + V_g$, the total volume.

These mass and volume definitions can be used to define the concepts of soil particle density, dry soil density, and total (wet) soil density. The dimensional units of soil density are mass per unit of cubic length ($M{\cdot}L^{-3}$).

Soil is a typical heterogeneous multiphase porous system, and the concept of average density can be used to define three parameters: (1) density of solids, or soil particle density, ρ_s; (2) bulk or dry density, ρ_b; and, (3) total or wet density, ρ_t.

2.1.1 Soil Particle Density

The soil particle density, ρ_s, or the density of solids, represents the density of the soil (i.e., mineral) particles collectively and is expressed as the ratio of the solid phase mass to the volume of the solid phase of the soil. Soil particle density is defined as follows:

$$\rho_s = M_s/V_s$$

In most mineral soils, the soil particle density has a short range of 2.6 - 2.7 g/cm^3 (Hillel 1980a). This is close to the density of quartz, which is usually the predominant constituent of sandy soils. A typical value of 2.65 g/cm^3 has been suggested to characterize the soil particle density of a general mineral soil (Freeze and Cherry 1979). Aluminosilicate clay minerals have particle density variations in the same range. The presence of iron oxides and other heavy minerals increases the value of the soil particle density. The presence of solid organic materials in the soil decreases the value.

2.1.2 Bulk (Dry) Density

The bulk or dry density of soil is the ratio of the mass of the solid phase of the soil (i.e., dried soil) to its total volume (solid and pore volumes together) and is defined as follows:

$$\rho_b = \frac{M_s}{V_t} = \frac{M_s}{V_s + V_l + V_g} \tag{2.1}$$

The bulk density, ρ_b, is related to the soil particle density, ρ_s, by the total soil porosity, for example:

$$\rho_b = (1-n)\ \rho_s \tag{2.2}$$

where n is the total porosity (see Section 3.1) and 1-n is the ratio of the solid volume (V_s) to the total volume ($V_l + V_g + V_s$).

From the above definition, it should be obvious that the value of dry density is always smaller than the value of soil particle density. For example, if the volume of pores ($V_l + V_g$) occupies half of the total volume, the value of dry density is half the value of the soil particle density.

The dry density of most soils varies within the range of 1.1 - 2.0 g/cm^3. In sandy soils, dry density can be as high as 2.0 g/cm^3, while in clayey soils and aggregated loams, it can be as low as 1.1 g/cm^3 (Hillel 1980b). Because of its high degree of aggregation (small total porosity), concrete has, in general, a higher dry density then soil. Typical values of dry density in different types of soils and in concrete are shown in Table 2.1. Dry density depends on the structure of the soil matrix (or its degree of compaction or looseness) and on its swelling/shrinkage characteristics.

TABLE 1.1 (Cont'd)

Parameter	Pathways									Input Screen
	External Gamma	Inhalation	Plant Ingestion	Meat Ingestion	Milk Ingestion	Aquatic Foods	Drinking Water	Radon	Soil Ingestion	
Basic radiation dose limit	used	used	used	used	used	used	used	used	used	R011
Dilution length for airborne dust	—	used	used	used	used	—	—	—	—	R017
Seafood consumption rate	—	—	—	—	—	used	—	—	—	R018
Fruit, vegetable, and grain consumption rates	—	—	used	—	—	—	—	—	—	R018
Inhalation rate	—	used	—	—	—	—	—	—	—	R017
Leafy vegetable consumption rate	—	—	used	—	—	—	—	—	—	R018
Livestock water intake rate										
-meat	—	—	—	used	—	—	—	—	—	R019
-milk	—	—	—	—	used	—	—	—	—	R019
Meat and poultry consumption rate	—	—	—	used	—	—	—	—	—	R018
Occupancy and shielding factor (external gamma)	used	—	—	—	—	—	—	—	—	R017
Elapsed time of waste placement	used	used	used	used	used	used	used	used	used	R011
Shape factor, external gamma	used	—	—	—	—	—	—	—	—	R017
Initial concentrations of principal radionuclide	used	used	used	used	used	used	used	used	used	R012
Drinking water intake rate	—	—	—	—	—	—	used	—	—	R018

[a] "—" indicates that the parameter is not used in the pathway calculations.

2 SOIL DENSITY

2.1 DEFINITION

The density of soil, as with any other material, is the ratio of the soil mass to the soil volume. This definition is applicable to any kind of homogeneous monophasic material. For heterogeneous and multiphasic materials, such as porous media, application of this concept can lead to different results, depending on the exact way the mass and volume of the system are defined.

In its general form, the soil system contains three natural phases: (1) the solid phase, which constitutes the soil matrix (i.e., mineral particles); (2) the liquid phase, which is often represented by water, and which could more properly be called the soil solution; and (3) the gaseous phase, which contains air and other gases, that is, the soil atmosphere. The mass and volumes associated with these three phases must be defined before the definitions of the different densities that characterize the soil system can be formalized.

Consider a representative elementary volume (REV) of soil that satisfies the following criteria (Bear 1972 and Marsily 1986):

1. A sufficiently large volume of soil containing a large number of pores, such that the concept of mean global properties is applicable; and
2. A sufficiently small volume of soil so that the variation of any parameter of the soil from one part of the domain to another can be approximated by continuous functions.

Within a REV, the masses of the phases composing the soil can be defined as follows:

M_s = the mass of solids,

M_l = the mass of liquids,

TABLE 1.1 (Cont'd)

Parameter	Pathways									Input Screen
	External Gamma	Inhalation	Plant Ingestion	Meat Ingestion	Milk Ingestion	Aquatic Foods	Drinking Water	Radon	Soil Ingestion	
Building foundation thickness	—	—	—	—	—	—	—	used	—	R021
Foundation depth below ground surface	—	—	—	—	—	—	—	used	—	R021
Fraction of time spent indoors	—	—	—	—	—	—	—	used	used	R021
Fraction of time spent outdoors	—	—	—	—	—	—	—	used	used	R021
Area of contaminated zone	used	used	used	used	used	used	used	used	used	R011
Cover depth	used	used	used	used	used	—	—	used	used	R013
Distribution coefficients	used	used	used	used	used	used	used	used	used	R016
Fractions of annular areas within contaminated area	used	—	—	—	—	—	—	—	—	R017
Radionuclide concentration in groundwater	used	used	used	used	used	used	used	used	used	R012
Leach rate	used	used	used	used	used	used	used	used	used	R016
Livestock fodder intake										
-meat	—	—	—	used	—	—	—	—	—	R019
-milk	—	—	—	—	used	—	—	—	—	R019
Mass loading for inhalation	—	used	—	—	—	—	—	—	—	R017
Milk consumption rate	—	—	—	—	used	—	—	—	—	R018
Occupancy factor for inhalation	—	used	—	—	—	—	—	—	—	R017
Depth of roots	—	—	used	used	used	—	—	—	—	R019
Soil ingestion rate	—	—	—	—	—	—	—	—	used	R018
Thickness of contaminated zone	used	used	used	used	used	—	—	used	used	R011

TABLE 1.1 (Cont'd)

Parameter	External Gamma	Inhalation	Plant Ingestion	Meat Ingestion	Milk Ingestion	Aquatic Foods	Drinking Water	Radon	Soil Ingestion	Input Screen
	Pathways									
Irrigation rate	—	—	used	—	—	—	—	used	—	R013
Evapotranspiration coefficient	used	used	used	used	used	used	used	used	used	R013
Soil-specific b parameter										
-contaminated zone	used	used	used	used	used	used	used	used	used	R013
-unsaturated zone	—	—	used	used	used	used	used	—	—	R015
-saturated zone	—	—	used	used	used	used	used	—	—	R014
Erosion rate										
-cover material	used	used	used	used	used	—	—	used	used	R013
-contaminated zone	used	used	used	used	used	—	—	—	used	R013
Hydraulic gradient	—	—	used	used	used	used	used	—	—	R014
Length of contaminated zone parallel to the aquifer flow	—	—	used	used	used	used	used	—	—	R011
Watershed area for nearby stream or pond	—	—	used	used	used	used	used	—	—	R013
Water table drop rate	—	—	used	used	used	used	used	—	—	R014
Well-pump intake depth	—	—	used	used	used	used	used	—	—	R014
Radon vertical dimension of mixing	—	—	—	—	—	—	—	used	—	R021
Average annual wind speed	—	—	—	—	—	—	—	used	—	R021
Average building air exchange rate	—	—	—	—	—	—	—	used	—	R021
Building room height	—	—	—	—	—	—	—	used	—	R021
Building indoor area factor	—	—	—	—	—	—	—	used	—	R021
Thickness of uncontaminated unsaturated zone	—	—	used	used	used	used	used	—	—	R015

the observed expansion in volume (i.e., the replaced volume of water) should be equal to V_s, the solid phase volume. The problem with this approach is that the techniques used to eliminate air bubbles from the mixture (such as heating) can also disturb the total volume and thus introduce errors into the calculations.

Another way to measure the solid phase volume (V_s), is based on evaluating the mass and density of water (or another fluid) displaced by the sample (after being oven dried). This second approach has been used for quite some time and is simple, direct, and accurate if done carefully (Blake 1986). It is based on the fact that, if V_{dw}, the volume of water displaced by the solids, is equal to V_s, then:

$$V_{dw} = \frac{M_{dw}}{\rho_w} = V_s = \frac{M_s}{\rho_s} \tag{2.4}$$

and

$$\rho_s = \rho_w \frac{M_s}{M_{dw}} \tag{2.5}$$

where M_{dw} is the mass of the displaced water and ρ_w is the water density. Therefore, to obtain the soil particle density it is necessary to evaluate the water density at the specific pressure and temperature conditions and to measure M_s and M_{dw}.

The value of M_{dw} is obtained by using a graduated volumetric flask and by making the following measurements:

M_f = mass of the empty flask;

M_{fs} = mass of the flask plus the dried soil sample;

M_{fsw} = mass of the flask plus the soil and filled with water up to a fixed volume, V_f;

M_{fw} = mass of the flask filled with pure water up to the fixed volume V_f.

Then, M_{dw}, or the mass of the displaced water can be calculated as follows:

$$M_{dw}=(M_{fs}-M_{f})-(M_{fsw}-M_{fw}) \tag{2.6}$$

And therefore, substituting M_{dw} into the expression for soil particle density, ρ_s, yields:

$$\rho_s = \rho_w \left[\frac{M_s}{(M_{fs}-M_{f})-(M_{fsw}-M_{fw})}\right] \tag{2.7}$$

This method is very precise, but it requires careful measurements of volumes and masses. Possible errors can result not only from determining the masses and volumes but from nonrepresentative sampling.

2.2.2 Dry Density Measurement

The dry (bulk) density (ρ_b) of a soil sample is evaluated on the basis of two measured values: (1) M_s, the oven-dried mass of the sample and (2) V_t, the field volume or the total volume of the sample. As stated previously, for the calculation of soil particle density (ρ_s), mass (M_s) is measured after drying the sample at 105°C until a near constant weight is reached. This laboratory technique determines directly the dry density of a soil sample. Possible direct methods of measuring the dry density include the clod, core, and excavation methods. These methods essentially consist of drying and weighing a known volume of soil.

Variations of these methods are related to different ways of collecting the soil sample and measuring volume. In the core method (Blake 1986), a cylindrical shaped metal sampler is introduced into the soil with care to avoid disturbing the sample. At the desired depth in the soil, a known field volume (V_t) of soil material is collected as it exists in-situ. The sample is then oven-dried and weighed to obtain the mass. The value of the dry density is calculated by dividing the mass by the volume. Problems in using this technique include sampling difficulties, such as the presence of gravels in the soil, and the possibility of disturbing the structure of the soil during the sampling process when the sampler is introduced into the ground.

In the excavation method (Blake 1986), the dry density of the soil is determined by excavating a hole in the ground, oven-drying and weighing the amount of soil removed from the ground to determine the mass, and measuring the volume of the excavation. The volume (V_t) can be determined in different ways. One is to use the sand-funnel method in which a selected type of sand with a known volume per unit mass is used to completely fill the hole. Then by measuring the total mass of sand needed to fill the hole, the volume can be determined. Another possible way to measure the volume (V_t) is to use the rubber-balloon method. In this technique, a balloon is placed within the hole and filled with a liquid (water) up to the borders of the hole. The volume of the excavated soil sample is then equal to the volume of the liquid in the balloon.

An advantage of using the excavation method to measure dry densities of soils other than the core method is that it is more suitable for heterogeneous soils with gravels.

An indirect method of measuring soil density, applicable for in-situ rather than laboratory determinations, is called the radiation method or gamma-ray attenuation densitometry (Blake 1986). This method is based on the principle that the amount of gamma radiation being attenuated and scattered in the soil depends on soil properties, including the combined densities of the solid-liquid components of the medium. By measuring the radiation that is transmitted through the medium or that is scattered by soil components and reaches a detector placed away from the source and by using proper calibration, the wet density of the soil, ρ_t, can be determined. To determine the dry density, ρ_b, a correction of the result is needed to remove the contribution present from the liquid phase in the soil.

The radiation method used for measuring soil density has several advantages over other related laboratory techniques: (1) in-situ evaluation of soil density, (2) minimum disturbance of the soil, (3) relatively short measurement time, (4) more applicable for deeper subsoil determinations because it requires minimal excavation, and (5) it is a nondestructive technique with the possibility of performing continuous or repeated measurements at the same spot. The radiation method also has some disadvantages compared with the other methods. Because it is a more sophisticated technique, it requires expensive equipment and highly trained operators who must be able to handle the frequent calibration procedures, the electronics, and the sampling equipment. The system operator must be trained in the radiation aspects and radiological protection procedures of the entire operation.

2.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD one variable is assigned to represent the dry density of each of five materials in grams per cubic centimeter (g/cm^3): (1) cover material, (2) contaminated zone, (3) unsaturated zone, (4) saturated zone, and (5) building foundation material (i.e., concrete). For the first four types of soil, a default value of 1.6 g/cm^3 is assigned for the dry density, a value that is in the upper range of variability and is representative of a sandy soil. Although the building foundation material (i.e., concrete) has a solid phase density (i.e., particle density) similar to that of the soil, because of its small total porosity, concrete has, in general, a higher dry density than soils. In RESRAD a default value of 2.4 g/cm^3 is assigned for the dry density of the foundation building material. This default value is provided for generic use of the RESRAD code. For more precise use of the code, site-specific data should be used.

If the type of soil is known, then Table 2.1 can be used for a slightly more accurate determination of the input data values for dry density. If there is no information about the type of soils, however, then the values for dry density should be experimentally determined by using one of the methods described in Section 2.2.2.

3 TOTAL POROSITY

3.1 DEFINITION

The total porosity of a porous medium is the ratio of the pore volume to the total volume of a representative sample of the medium. Assuming that the soil system is composed of three phases — solid, liquid (water), and gas (air) — where V_s is the volume of the solid phase, V_l is the volume of the liquid phase, V_g is the volume of the gaseous phase, $V_p = V_l + V_g$ is the volume of the pores, and $V_t = V_s + V_l + V_g$ is the total volume of the sample, then the total porosity of the soil sample is defined as follows:

$$n = \frac{V_p}{V_t} = \frac{V_l+V_g}{V_s+V_l+V_g} \tag{3.1}$$

Porosity is a dimensionless quantity and can be reported either as a decimal fraction or as a percentage. Table 3.1 lists representative total porosity ranges for various geologic materials. A more detailed list of representative porosity values (total and effective porosities) is presented in Table 3.2. In general, total porosity values for unconsolidated materials lie in the range of 0.25 - 0.7 (25% - 70%). Coarse-textured soil materials such as gravel and sand tend to have a lower total porosity than fine-textured soils such as silts and clays. The total porosity in soils is not a constant quantity because the soil, particularly clayey soil, alternately swells, shrinks, compacts, and cracks.

3.2 MEASUREMENT METHODOLOGY

On the basis of its definition, the total porosity of a soil sample can be most easily evaluated by measuring directly the pore volume (V_p) and the total volume (V_t). The total volume is easily obtained by measuring the total volume of the sample. The pore volume can, in principle, be evaluated directly by measuring the volume of water needed to completely

TABLE 3.1 Range of Porosity Values

Soil Type	Porosity, n (%)
Unconsolidated deposits	
Gravel	25-40
Sand	25-50
Silt	35-50
Clay	40-70
Rocks	
Fractured basalt	5-50
Karst limestone	5-50
Sandstone	5-30
Limestone, dolomite	0-20
Shale	0-10
Fractured crystalline rock	0-10
Dense crystalline rock	0-5

Source: Freeze and Cherry 1979.

TABLE 3.2 Representative Porosity Values

Material	Total Porosity, n Range	Total Porosity, n Arithmetic Mean	Effective Porosity, n_e Range	Effective Porosity, n_e Arithmetic Mean
Sedimentary Material				
Sandstone (fine)			0.02 - 0.40	0.21
Sandstone (medium)	0.14 - 0.49	0.34	0.12 - 0.41	0.27
Siltstone	0.21 - 0.41	0.35	0.01 - 0.33	0.12
Sand (fine)	0.25 - 0.53	0.43	0.01 - 0.46	0.33
Sand (medium)	-	-	0.16 - 0.46	0.32
Sand (coarse)	0.31 - 0.46	0.39	0.18 - 0.43	0.30
Gravel (fine)	0.25 - 0.38	0.34	0.13 - 0.40	0.28
Gravel (medium)	-	-	0.17 - 0.44	0.24
Gravel (coarse)	0.24 - 0.36	0.28	0.13 - 0.25	0.21
Silt	0.34 - 0.51	0.45	0.01 - 0.39	0.20
Clay	0.34 - 0.57	0.42	0.01 - 0.18	0.06
Limestone	0.07 - 0.56	0.30	~0 - 0.36	0.14
Wind-Laid Material				
Loess	-	-	0.14 - 0.22	0.18
Eolian sand	-	-	0.32 - 0.47	0.38
Tuff	-	-	0.02 - 0.47	0.21
Igneous Rock				
Weathered granite	0.34 - 0.57	0.45	-	-
Weathered gabbro	0.42 - 0.45	0.43	-	-
Basalt	0.03 - 0.35	0.17	-	-
Metamorphic Rock				
Schist	0.04 - 0.49	0.38	0.22 - 0.33	0.26

Source: McWorter and Sunada 1977.

saturate the sample. In practice, however, it is always difficult to saturate the soil sample exactly and completely and, therefore, the total porosity of the sample is rarely evaluated by a direct method. Instead, the total porosity is usually evaluated indirectly by using the following expression (Danielson 1986):

$$n = \left[1 - \frac{\rho_b}{\rho_s}\right] \tag{3.2}$$

where n is given as a decimal fraction, ρ_s is the solid phase (soil particle) density, and ρ_b is the dry bulk density of the sample.

Under this approach, the values of ρ_s and ρ_d are evaluated by laboratory or in-situ measurements (see Section 2.2) and are then used to calculate the total porosity n.

3.3 RESRAD DATA INPUT REQUIREMENTS

To use RESRAD, the user is required to define or use the default values of the total porosity of five materials: (1) the cover material, (2) the contaminated zone, (3) the unsaturated zone, (4) the saturated zone, and, (5) the building foundation material (i.e., concrete). In RESRAD, the total porosities are entered as decimal fractions rather than as percentages. RESRAD adopts the following values as defaults: n = 0.4 for the first four materials listed above and n = 0.1 for the building foundation (i.e., concrete).

4 EFFECTIVE POROSITY

4.1 DEFINITION

The effective porosity n_e, also called the kinematic porosity, of a porous medium is defined as the ratio of the part of the pore volume where the water can circulate to the total volume of a representative sample of the medium. The effective porosity can also be defined as the ratio of the volume of water drained by gravity from a representative sample of the medium to the total volume of the sample.

The definition of effective (kinematic) porosity is linked to the concept of pore fluid displacement rather than to the percentage of the volume occupied by the pore spaces. The empty volume occupied by the pore fluid that can circulate through the porous medium is smaller than the total pore space and, consequently, the effective porosity is always smaller than the total porosity. In a saturated soil system composed of two phases (solid and liquid) where (1) V_s is the volume of the solid phase, (2) $V_w = (V_{aw} + V_{fw})$ is the volume of the liquid phase, (3) V_{aw} is the volume of the water adsorbed onto the soil particle surfaces, (4) V_{fw} is the volume of the water that is free to move through the saturated system, and (5) $V_t = (V_s + V_{aw} + V_{fw})$ is the total volume, the effective porosity can be defined as follows:

$$n_e = \frac{V_{fw}}{V_t} = \frac{V_{fw}}{V_s+V_{fw}+V_{aw}} \tag{4.1}$$

A soil parameter related to the effective soil porosity is the field capacity, or the irreducible volumetric water content, θ_r, which is defined as the ratio of the volume of water retained in the soil sample after all downward gravity drainage has ceased to the total volume of the sample. Considering the terms presented above for a saturated soil system, the total porosity, n, and the field capacity, θ_r, can be expressed, respectively, as follows:

$$n = \frac{V_{fw}+V_{aw}}{V_t} \tag{4.2}$$

and

$$\theta_r = \frac{V_{aw}}{V_t} \tag{4.3}$$

Therefore, the effective porosity is related to the total porosity and the field capacity according to the following expression:

$$n_e = n-\theta_r \tag{4.4}$$

Several aspects of the soil system influence the value of the effective porosity. They include (1) the adhesive water on minerals, (2) the absorbed water in clay mineral lattice, (3) the existence of unconnected pores, and (4) the existence of dead-end pores. The adhesive water in the soil is that part of the water present in the soil that is attached to the surface of the soil grains through the forces of molecular attraction (Marsily 1988). The sum of the volumes of the adhesive and absorbed water plus the water that fills the unconnected and dead-end pores constitute the volume of the adsorbed water, V_{aw}, which is not able to move through the system.

A detailed list of representative porosity values (total porosity and effective porosity) is presented in Table 3.2.

4.2 MEASUREMENT METHODOLOGY

Determination of the effective porosity of soils can be accomplished indirectly by measuring the total porosity and the field capacity, θ_r, and then calculating n_e from the following expression:

$$n_e = n-\theta_r \tag{4.5}$$

The total porosity is also obtained indirectly by measuring the soil densities according to the method already described in Section 3.2. To determine the field capacity of the soils, the soil sample is first saturated with water and is then let to drain completely under the action of gravity until it gets to its irreducible saturation. The value of θ_r can then be obtained according to the methods used for measuring the volumetric water content (see Section 6.2).

4.3 RESRAD DATA INPUT REQUIREMENTS

To use RESRAD, the user is required to define (or to use the default values) of the effective porosity of three distinct materials: (1) the contaminated zone, (2) the saturated zone, and (3) the unsaturated zone. In RESRAD, the porosity values are entered as decimal fractions rather than as percentages. As a default value, RESRAD adopts the value of $n_e = 0.2$ for all three materials.

5 HYDRAULIC CONDUCTIVITY

5.1 DEFINITION

The hydraulic conductivity of a soil is a measure of the ability of the soil to transmit water when submitted to a hydraulic gradient. It is defined by Darcy's law which, for one-dimensional vertical flow, can be written as follows:

$$U = -K\frac{dh}{dz} \tag{5.1}$$

where U is Darcy's velocity (or the average velocity of the soil fluid through a geometric cross-sectional area within the soil), h is the hydraulic head, and z is the vertical distance in the soil. The coefficient of proportionality, K, in Equation 5.1 is called the hydraulic conductivity. The term coefficient of permeability is also sometimes used as a synonym for hydraulic conductivity. On the basis of Equation 5.1, the hydraulic conductivity is defined as the ratio of Darcy's velocity to the applied hydraulic gradient. The dimension of K is the same as that for velocity, that is, length per unit of time (L/T).

The hydraulic conductivity is one of the hydraulic properties of the soil; the other involves the soil fluid retention characteristics. These properties determine the behavior of the soil fluid within the soil system under specified conditions. More specifically, the hydraulic conductivity determines the ability of the soil fluid to flow through the soil matrix system under a specified hydraulic gradient; the soil fluid retention characteristics determine the ability of the soil system to retain the soil fluid under a specified pressure condition.

The hydraulic conductivity depends on the grain size and the structure of the soil matrix, the type of soil fluid, and the relative amount of soil fluid (saturation) present in the soil matrix. The important properties relevant to the solid matrix of the soil include the pore size distribution, the shape of the pores, the tortuosity, the specific surface, and the porosity. In relation to the soil fluid, the important properties include the fluid density, ρ, and the fluid viscosity, μ. For a subsurface system saturated with the soil fluid, the hydraulic conductivity K can be expressed as follows (Bear 1988):

$$K = \frac{k\rho g}{\mu} \tag{5.2}$$

where k, the intrinsic permeability of the soil, depends only on properties of the solid matrix, and ρg/μ, called the fluidity of the liquid, represents the properties of the percolating fluid. In the International System of Units (SI), the hydraulic conductivity K is expressed in terms of meters per second (m/s), the intrinsic permeability k is expressed in m^2, and the fluidity ρg/μ in 1/m·s. Using Equation 5.2, Darcy's law can be rewritten explicitly in terms of its coefficient of proportionality (hydraulic conductivity K):

$$K = \frac{k\rho g}{\mu} = \frac{|U|}{|dh/dz|} \tag{5.3}$$

When the fluid properties of density and viscosity are known, Equation 5.3 can be used to determine experimentally the value of the intrinsic permeability, k, and the hydraulic conductivity, K, as will be shown in Section 5.2.

The values of saturated hydraulic conductivity in soils vary in a wide range of several orders of magnitude, depending on the soil material. Table 5.1 lists representative values of saturated hydraulic conductivity to be expected for a range of unconsolidated and consolidated soil materials. A detailed list of expected representative values of K for several soil materials is also presented in Table 5.2.

Because of the spatial variability usually found in the geological formation of soils, saturated hydraulic conductivity values also show variations throughout the space domain within a subsurface geological formation. Such a geological formation is said to be heterogeneous. If the properties of the geologic formation are invariable in space, the

TABLE 5.1 Saturated Hydraulic Conductivity, K, in Water of Various Soil Materials

Soil Type	Range of K (m/yr)
Unconsolidated Deposits	
Gravel	$1 \times 10^4 - 1 \times 10^7$
Clean sand	$1 \times 10^2 - 1 \times 10^5$
Silty sand	$1 \times 10^0 - 1 \times 10^4$
Silt, loess	$1 \times 10^{-2} - 1 \times 10^2$
Glacial till	$1 \times 10^{-5} - 1 \times 10^1$
Unweathered marine clay	$1 \times 10^{-5} - 1 \times 10^{-2}$
Rocks	
Shale	$1 \times 10^{-6} - 1 \times 10^{-2}$
Unfractured metamorphic and igneous rocks	$1 \times 10^{-7} - 1 \times 10^{-3}$
Sandstone	$1 \times 10^{-3} - 1 \times 10^1$
Limestone and dolomite	$1 \times 10^{-2} - 1 \times 10^1$
Fractured metamorphic and igneous rocks	$1 \times 10^{-1} - 1 \times 10^3$
Permeable basalt	$1 \times 10^0 - 1 \times 10^5$
Karst limestone	$1 \times 10^1 - 1 \times 10^5$

Source: Adapted from Freeze and Cherry 1979.

TABLE 5.2 Representative Values of Saturated Hydraulic Conductivity and Saturated Water Content

Texture	Hydraulic Conductivity, K_{sat} (m/yr)	Saturated Water Content θ_{sat}
Sand	5.55×10^{-3}	0.395
Loamy sand	4.93×10^{-3}	0.410
Sandy loam	1.09×10^{-3}	0.435
Silty loam	2.27×10^{-2}	0.485
Loam	2.19×10^{-2}	0.451
Sandy clay loam	1.99×10^{-2}	0.420
Silty clay loam	5.36×10^{-1}	0.477
Clay loam	7.73×10^{-1}	0.476
Sandy clay	6.84×10^{-1}	0.426
Silty clay	3.21×10^{-1}	0.492
Clay	4.05×10^{-1}	0.482

Source: Clapp and Hornberger 1978.

formation is homogeneous. A geological formation is said to be isotropic if the values of the saturated hydraulic conductivity (K), at any point in the medium, are independent of the direction of measurement. Again, because of the usually stratified nature of the geological formation of unconsolidated sedimentary soil materials, the soils are usually anisotropic. Within an anisotropic geological formation, the vertical component of the saturated hydraulic conductivity is usually smaller (one to two orders of magnitude) than the horizontal component.

5.2 MEASUREMENT METHODOLOGY

The saturated hydraulic conductivity of water in the soil (or the intrinsic permeability of the soil) can be measured by both field and laboratory experiments. Either way, the experimental measurement of K (or k) consists in determining the numerical value for the coefficient in Darcy's equation.

The methodology used for the experimental determination of K (or k) in either laboratory or field experiments is based on the following procedures (Bear 1988):

(1) Assume a flow pattern (such as one-dimensional flow in a porous medium) that can be described analytically by Darcy's law;

$$K = \frac{k\rho g}{\mu} = \frac{|U|}{|dh/dz|} \tag{5.4}$$

(2) Perform an experiment reproducing the chosen flow pattern and measure all measurable quantities in Equation 5.4, including fluid density, viscosity dynamic velocity, flow velocity, and the gradient of the hydraulic head; and

(3) Compute the coefficient K (or k) by substituting the measured quantities into Equation 5.4 above.

Many different laboratory or field experiments can be used to determine the coefficient K (or k). The laboratory tests are carried out on small samples of soil materials collected during core-drilling programs. Because of the small sizes of the soil samples handled in the laboratory, the results of these tests are considered a point representation of the soil properties. If the soil samples used in the laboratory test are truly undisturbed samples, the measured value of K (or k) should then be a true representation of the in situ saturated hydraulic conductivity at that particular sampling point.

In contrast to laboratory methods for measuring soil parameters, field methods, in general (including the saturated hydraulic conductivity), involve a large region of the soil and, consequently, their results should reflect an average value of the property being measured. Selection of a specific method for a particular application will depend on the objectives to be achieved. Because of the difficulty in obtaining a perfectly undisturbed sample of unconsolidated soil, the K value determined by laboratory methods may not be applicable to the field.

Therefore, field methods should be used whenever the objective is to characterize as accurately as possible the physical features of the subsurface system in question. Field methods, however, are usually more expensive than laboratory methods and, consequently, when the question of cost becomes decisive, or when the actual representation of the field conditions is not of fundamental importance and in situ hydraulic conductivity is not available, the laboratory methods may be used to determine the saturated hydraulic conductivity K.

5.2.1 Laboratory Methods

In the laboratory, the value of K is determined by means of an instrument called a permeameter (Bear 1988). In a permeameter, a soil sample is placed in a small cylindrical recipient representing a one-dimensional soil configuration through which the circulating liquid is forced to flow. The size of the cylindrical soil sample used in the permeameter varies from 2 to 10 cm in diameter and from 5 to 25 cm in length. Depending on the flow pattern imposed through the soil sample, the permeameter is classified as either a constant-head permeameter, with a steady state flow regimen, or a falling-head permeameter, with an unsteady state flow regimen.

Important considerations regarding the use of permeameters are related to the soil sampling procedure and the preparation of the test specimen and the circulating liquid. The sampling process usually disturbs the matrix structure of the soil, resulting in a misrepresentation of the actual field conditions. Undisturbed sampling of soils is possible, but it requires the use of specially designed techniques and instruments (Bear 1988; Klute and Dirksen 1986).

Selecting the test fluid is also of fundamental importance for the laboratory determination of the saturated hydraulic coefficient. The objective is to have the test fluid mimic the actual properties of the soil fluid as closely as possible. When an inappropriate test fluid is selected, the test sample can get clogged with entrapped air, bacterial growth, and fines. To avoid such problems, a standard test solution such as a deaerated 0.005 mol calcium sulfate ($CaSO_4$) solution, saturated with thymol (or sterilized with another substance, such as formaldehyde) should be in the permeameter, unless there are specific reasons to choose another solution (Klute and Dirksen 1986).

5.2.1.1 Constant-Head Permeameter

The constant head permeameter is one of the most commonly used instruments for determining the hydraulic conductivity of saturated soils in the laboratory. It operates in accordance with the direct application of Darcy's law to a soil liquid configuration representing a one-dimensional, steady flow of a percolating liquid through a saturated column of soil from a uniform cross-sectional area. In this instrument, a cylindrical soil sample of cross-sectional area A and length L is placed between two porous plates that provide no extra hydraulic resistance to the flow. A constant head difference, H_2-H_1, is then applied across the test sample. By measuring the volume V of the test fluid that flows through the system during time t, the saturated hydraulic conductivity K of the soil can be determined directly from Darcy's equation:

$$K = \frac{VL}{[At(H_2-H_1)]} \tag{5.5}$$

To improve the results, it is recommended that the test should be performed several times under different head permeameters. It is also recommended that the quantity of liquid collected should be sufficient to provide at least three significant figures in the measured volume. In a simple version of the constant-head permeameter, the lower limit of the measurement of K is approximately 1 × 10^{-5} cm/s, which corresponds to the lower limit of the conductivity of sandy clay soils. For lower values of K, it is recommended that either an enhanced version of the constant-head permeameter (i.e., one that has a more sensitive method of measuring the volume flow rate) or the falling-head permeameter be used (Klute and Dirksen 1986).

5.2.1.2 Falling-Head Permeameter

The falling-head parameter is primarily used for determining the value of K (or k) in the laboratory when a high head is desired. Like the constant-head permeameter, it also operates in accordance with the direct application of Darcy's law to a one-dimensional, saturated column of soil from a uniform cross-sectional area. The falling-head permeameter differs from the constant-head permeameter in that the liquid that percolates through the saturated column is kept at an unsteady state flow regimen in which both the head and the discharged volume vary during the test. In the falling-head permeameter, a cylindrical soil sample of cross-sectional area A and length L is placed between two highly conductive plates.

The soil sample column is connected to a standpipe of cross-sectional area a, in which the percolating fluid is introduced into the system. Thus by measuring the change in head in the standpipe, from H_1 to H_2, during a specified interval of time t, the saturated hydraulic conductivity can be determined as follows (Klute and Dirksen 1986):

$$K = \left(\frac{aL}{At}\right) \ln\left(\frac{H_1}{H_2}\right) \tag{5.6}$$

A common problem encountered in using both the constant-head and falling-head permeameter is related to the degree of saturation achieved within the soil samples during the test. Air bubbles are usually trapped within the pore space, and although they tend to disappear slowly by dissolving into the deaerated water, their presence in the system may alter the measured results. Therefore, after measuring K by using these instruments, it is always recommended that the degree of saturation of the sample be verified by measuring the sample's volumetric water content and comparing the result with the total porosity calculated from the particle density.

The lower limit of K, which can be measured in a falling-head permeameter, is about 1×10^{-7} cm/s. This value corresponds approximately to the lower limit of conductivity of silts and coarse clays (Klute and Dirksen 1986).

5.2.2 Field Methods

The several methods developed for in-situ determination of saturated hydraulic conductivity of soils can be separated into two groups: (1) those that are applicable to sites with a shallow water table and (2) those that are applicable to sites with a deep water table. In either group (similar to the laboratory methods), the determination of K is obtained from Darcy's law after measuring the gradient of the hydraulic head at the site and the resulting soil water flux.

The most commonly used methods for determining the saturated hydraulic conductivity of soils in the presence of a shallow water table are the auger-hole and piezometer methods (Amoozegar and Warrick 1986).

5.2.2.1 Auger-Hole Method

The auger-hole method is the field procedure most commonly used for in situ determination of saturated hydraulic conductivity of soils. This method has many possible variations (Amoozegar and Warrick 1986). In its simplest form, it consists of the preparation of a cavity partially penetrating the aquifer, with minimal disturbance of the soil. After preparation of the cavity, the water in the hole is allowed to equilibrate with the groundwater; that is, the level in the hole becomes coincident with the water table level. The actual test starts by removing the entire amount of water from the hole and by measuring the rate of the rise of the water level within the cavity.

Because of the three-dimensional aspect of the flow pattern of the water near the cavity, there is no simple equation for the accurate determination of the conductivity. Yet numerous available semiempirical expressions can be used for the determination of the saturated hydraulic conductivity for different soil configurations. These expressions are functions of the geometrical dimensions of the auger hole and the aquifer and the measured rate in which the water level in the hole changes with time (Amoozegar and Warrick 1986).

The auger-hole method is applicable to an unconfined aquifer with homogeneous soil properties and a shallow water table. In its simplest form, this method provides an estimate of the average horizontal component of the saturated hydraulic conductivity of the soil within the aquifer. Enhanced variations of the method have been developed to account for layered soils and for the determination of either horizontal or vertical components of saturated hydraulic conductivity. Results obtained by the auger-hole method are not reliable for cases in which (1) the water table is above the soil surface, (2) artesian conditions exist, (3) the soil structure is extensively layered, and (4) small strata of high permeability occur.

5.2.2.2 Piezometer Method

The piezometer method, like the auger-hole method, is applicable for the determination of the saturated hydraulic conductivity of soils in an unconfined aquifer with a shallow water table level. Unlike the auger-hole method, however, the piezometer method is appropriately designed for applications in layered soil aquifers and for the determination of either horizontal or vertical components of the saturated hydraulic conductivity.

This method consists of installing a piezometer tube or pipe into an auger hole drilled through the subsurface system without disturbing the soil. The piezometer tube should be long enough to partially penetrate the unconfined aquifer. The walls of the piezometer tube are totally closed except at its lower extremity, where the tube is screened open to form a cylindrical cavity of radius r and height h_c within the aquifer. The water in the piezometer tube is first removed to clean the system and is then allowed to equilibrate with the groundwater level.

Similar to the auger-hole method, the piezometer method is conducted by removing the water from the pipe and then measuring the rate of the rise of the water within the pipe. The saturated hydraulic conductivity is then evaluated as a function of the geometrical dimension of the cavity in the piezometer tube, the dimensions of the aquifer, and the measured rate of rise of the water table in the tube. The value for the conductivity is calculated with the help of a nomograph and tables (Amoozegar and Warrick 1986).

Depending on the relative length (h_c) of the cavity as compared with its radius (r), the piezometer method can be used to determine the horizontal or vertical component of the saturated hydraulic conductivity. Thus, if h_c is large compared to r, the results obtained reflect the horizontal component of K. Otherwise, if h_c is small compared to r, then the vertical component of K is estimated.

The piezometer method is especially suitable for determining the conductivity of individual layers in stratified subsurface systems.

5.2.2.3 Other Methods Used in Shallow Water Table Aquifers

Other methods have been developed for in-situ determination of saturated hydraulic conductivity of soils in unconfined shallow water table aquifers. These methods include (1) pumping and slugs tests, primarily developed for the determination of aquifer properties used in the development of groundwater systems and (2) other tests, such as the two-well, four-well, and multiple-well methods and well-point, pit-bailing, and field monoliths (Amoozegar and Warrick 1986).

In contrast to the auger-hole and piezometer methods whose results reflect an in-situ average of a relatively small region of soil around the created cavity in the soil, the well-pumping tests also provide an in situ representation of the soil hydraulic conductivity but averaged over a larger representative volume of the soil. Well-pumping methods are more laborious and costly than the auger-hole and piezometer methods, and unless there are reasons to conclude otherwise, laboratory methods maybe considered to replace the well-pumping methods for the determination of the saturated hydraulic conductivity of soils. (Further references for these methods can be found in [Freeze and Cherry 1979] and [Amoozegar and Warrick 1986].)

Measuring the saturated hydraulic conductivity (k) of unsaturated soils located above the water table (or in the absence of a water table) by in situ methods is more difficult than measuring k for saturated soils. The important difference is that the original unsaturated soil must be artificially saturated to perform the measurements. An extra large quantity of water may be needed to saturate the medium, which results in a more elaborate and time-consuming measurement. Therefore, in situ methods are not recommended for obtaining input data for RESRAD. (Further references about these methods, including the double-tube method, shallow well press-in method, cylindrical permeameter method, air-entry permeameter method, and infiltration gradient techniques can be found in [Amoozegar and Warrick 1986].)

5.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input a saturated hydraulic conductivity value in units of meters per year (m/yr) for three soil materials: contaminated, unsaturated, and saturated zones.

The vertical infiltration of water within the contaminated zone and through the unsaturated region of the soil, with the subsequent vertical leaching and transport of contaminants into the underlying aquifer, constitutes the important aspects of the problem being modeled. Consequently, in RESRAD, the saturated hydraulic conductivity values related to the contaminated and unsaturated zones of the soil should represent the vertical component of K. For isotropic soil materials, the vertical and horizontal components of K are the same; for anisotropic soils, however, the vertical component of K could be lower than the horizontal component.

The major concern within the saturated zone is related to the horizontal transport of the contaminants that have infiltrated through the unsaturated zone and reached the aquifer. Therefore, the input value for the saturated hydraulic conductivity (K) of the soil material in the saturated zone should reflect the horizontal component of K.

The estimation of the values of K to be used in RESRAD can be performed at different levels of site-specific accuracy, depending on the amount of information available. For generic use of the code, a set of default values of K is defined as 10 m/yr for the contaminated zone and the unsaturated zone and 100 m/yr for the saturated zone. These values represent approximately the condition of an anisotropic soil material of the type silt, loess, or silty sand, in which the vertical component of K is one order of magnitude lower than the horizontal component.

If the geological stratigraphy at the site is known, a better (i.e., more accurate and site-specific) estimation of K can be performed with the help of Table 5.1 or Table 5.2.

For an accurate site-specific estimation of the input data for RESRAD, the values of K should be measured either in the laboratory or in field experiments.

Because of the intrinsic difficulties of the methods available for in-situ measurements of K in unsaturated regions of the soil, it is recommended that laboratory methods be used for the determination of K in the contaminated and unsaturated zones. In these cases, either the constant-head or the falling-head permeameter methods can be used, depending only on the actual values of K being measured. As discussed earlier, the constant-head permeameter is more applicable for large values of K (in the range of 10^0 - 10^6 m/yr), and the falling-head method is more applicable for lower values of K (in the range of 10^{-2} - 10^2 m/yr).

For the determination of K in the saturated zone, either the laboratory methods (i.e., constant-head and falling-head) or the field methods (i.e., auger-hole and piezometer) can be used.

6 VOLUMETRIC WATER CONTENT

6.1 DEFINITION

The volumetric water content, θ, in the soil (also called the volume wetness or volume fraction of soil water) represents the fraction of the total volume of soil that is occupied by the water contained in the soil. Assuming that V_w is the volume of water in the soil and that V_t is the total volume of soil, the volumetric water content, θ, can then be defined as follows:

$$\theta = \frac{V_w}{V_t} = \frac{V_w}{V_s + V_p} \tag{6.1}$$

where V_s and V_p represent, respectively, the volumes of the solid phase and the pore space.

The volumetric water content is often expressed in terms of the total porosity, n, and the water saturation, s, according to the following expression:

$$\theta = ns \tag{6.2}$$

where n is the total porosity (see Section 3.1 for parameter definition) and s, the water saturation, is defined as the ratio of the volume of water, V_w, to the volume of the pore space, V_p. Therefore, considering the definitions of n and s, the expression for the volumetric water content θ can be rewritten as follows:

$$\theta = ns = \left(\frac{V_p}{V_t}\right)\left(\frac{V_w}{V_p}\right) = \frac{V_w}{V_t} \tag{6.3}$$

The possible values for θ range from near zero for dry soils approaching zero saturation, up to the value of the total porosity for fully saturated soils. The lower limit of zero for the volumetric water content is hardly achievable because it is difficult to completely eliminate the water from the soil. In sandy soils, the upper limit of θ, which is equal to the total porosity n, is also hardly achievable because of the difficulty of eliminating all the air bubbles in the soil in order to saturate it completely. Yet, because clayey soils swell upon wetting, the values of θ for these soils can exceed their total porosity.

6.2 MEASUREMENT METHODOLOGY

Determination of the volumetric water content of soils may be accomplished by direct and indirect methods. The direct methods consist essentially of drying and weighing a known volume of a soil sample. The indirect methods are based on the correlation of certain physical and physicochemical properties of the soil with its water content.

Generally, in a direct measurement method, the volumetric water content θ of a soil sample is evaluated on the basis of three measured quantities: (1) W_w, the wet weight of the soil sample; (2) W_d, the oven-dried weight of the sample; and, (3) V_t, the field volume or the total volume of the sample. With these measured quantities available, the volume of water, V_w, in the sample can then be calculated as follows:

$$V_w = \frac{(W_w - W_d)}{\rho_w} \tag{6.4}$$

and the volumetric water content (θ) can be finally evaluated as:

$$\theta = \frac{V_w}{V_t} = \frac{W_w - W_d}{V_t \rho_w} \tag{6.5}$$

where ρ_w is the density of water.

Variations in the direct methods for determining the volumetric water content are related to different ways of collecting the soil samples, measuring the field volume (V_t), and drying the samples. Possible direct methods of collecting the soil samples and measuring V_t have been discussed in Section 2.2 in regard to soil densities.

The definition of a dry state for the soil sample (and the establishment of a method to achieve this state) constitutes the key problem in determining the volumetric water content in soils. As a common practice, such as that described in Section 2.2, the oven-dried weight of the soil sample is measured after drying the sample at 105°C until a near constant weight is reached (Hillel 1980b). As discussed by Gardner (1986), however, this definition for the ovendry method is not precise enough and could create uncertainties and inaccuracies in the measured result. Therefore, if the determinations of water content for a particular site are considered critical, other procedures than the ovendry method should be adopted (Gardner 1986).

The indirect methods of measuring the water content in soils rely on certain physical and physicochemical properties of the soil and their relation with the volumetric water content (θ). Usually these relationships are complicated and require a sophisticated methodology and equipment to express them. The indirect methods of measuring volumetric water content are applicable for in-situ rather than laboratory determinations and involve measuring some property of the soil that is affected by soil water content such as (1) electrical conductivity, (2) neutron scattering, or (3) neutron and gamma-ray absorption (Gardner 1986).

Similar to the discussion of the determination of soil densities, the indirect methods used for measuring volumetric water content present some advantages over the other related laboratory techniques. The main advantages are (1) in-situ evaluation of the water content, (2) minimum disturbance of the soil, (3) relatively short measurement time, (4) applicability to deeper subsoil determinations because of minimum excavation requirement, and (5) nondestructive with the possibility of continuous or repeated measurements at the same spot. The disadvantages of such indirect methods are that they are more sophisticated and require expensive equipment and highly trained operators who must be able to handle the frequent calibration procedures, the electronics, and the sampling equipment. In the case of a system that uses radioactive elements, the operator must be particularly trained in the radiation aspects and radiological protection procedures of the whole operation.

6.3 RESRAD DATA INPUT REQUIREMENTS

To use RESRAD it is necessary to define an input value for the volumetric water content (θ) of the soil of the cover zone and the building foundation material (i.e., concrete). In RESRAD, the dimensionless values of the volumetric water content are entered as decimal fractions rather than as percentages.

For generic use of the model, a set of default values for the volumetric water content is defined internally in the code. The default values are $\theta = 0.05$ for the cover material and $\theta = 0.01$ for the building foundation material (i.e., concrete). Considering the default values for total porosity, 0.4 and 0.1, the volumetric water content values correspond to saturations of 0.125 and 0.1 for the cover material and concrete, respectively.

7 EFFECTIVE RADON DIFFUSION COEFFICIENT

7.1 DEFINITION

The random movement of the radon gas atoms mixed in the air results in a net migration of the radon gas toward the direction of its decreasing concentration in the air. This phenomenon is called molecular or atom diffusion. The diffusion of radon in open air can be described by Fick's law, which states that the flux density of the diffusing substance is linearly proportional to its concentration gradient. Fick's law can be expressed as follows:

$$\bar{J} = - D_o \bar{\nabla} C \tag{7.1}$$

where J is a vector representing the density flux of radon activity in units of activity$\cdot L^{-2}T^{-1}$), $\bar{\nabla}C$ is a vector representing the gradient of radon activity concentration in the air in units of activity$\cdot L^{-4}$), and D_o is the molecular (or atom) diffusivity or the diffusion coefficient of radon in open air in units of $L^2 \cdot T^{-1}$. Therefore, the diffusion coefficient D_o can be defined from Fick's equation and expressed as the ratio of the magnitudes of the vectors J to $\bar{\nabla}C$:

$$D_o = \frac{|\bar{J}|}{|\bar{\nabla}C|} \tag{7.2}$$

For radon diffusion in open air, Fick's law is uniquely expressed and, consequently, the diffusion coefficient of radon in open air, D_o, is also uniquely defined. However, when applied to the conditions of radon diffusion in porous media, such as in soil materials, Fick's equation can be written in different ways, depending on how the variables flux density J and concentration C are defined. Fick's equation can be written four ways when applied to the molecular diffusion phenomenon in porous media depending on whether the bulk or pore volume is used to define the concentration and whether the bulk or pore area is used to define the flux density. These different ways to define the radon diffusion coefficient in soil lead to some confusion in selecting and using these parameters because the symbols and nomenclature used have not been standardized (Nazaroff et al. 1988).

Two distinct ways of defining the diffusion coefficient of radon in porous media have been adopted in the literature: (1) D_e is the effective radon diffusion coefficient and (2) D is the bulk radon diffusion coefficient. However, Culot et al. (1976) and Nazaroff et al. (1988) have noted discrepancies in the literature regarding the way these two coefficients are defined and used in modeling the diffusion of radon through porous media. Therefore, the definitions of D_e and D adopted in this handbook are those suggested by Nazaroff et al. (1988).

Thus, the effective (or interstitial) radon diffusion coefficient, D_e, is defined from Fick's equation as the ratio of the diffusive flux density of radon activity across the pore area, J_e, to the gradient of the radon activity concentration in the pore or interstitial space, $\bar{\nabla} C$. This definition is equivalent to that relating the bulk flux density to the gradient of the bulk concentration of radon activity in the soil. It can be expressed as follows:

$$D_e = \frac{|\bar{J}_e|}{|\bar{\nabla} C|} \tag{7.3}$$

The bulk radon diffusion coefficient, D, is defined as the ratio of the diffusive flux density of radon activity across a geometric or superficial area of the medium, J_b, to the gradient of the radon activity concentration in the pore space, $\bar{\nabla} C$. It can be expressed as follows:

$$D = \frac{|\bar{J}_b|}{|\bar{\nabla} C|} \tag{7.4}$$

The bulk and the effective radon diffusion coefficients in soil, D and D_e, respectively, are correlated by the total soil porosity, n, according to the following expression:

$$D = nD_e \tag{7.5}$$

In general, the diffusion coefficient in porous media is a property of the diffusing species, the pore structure, the type of fluids present in the pores, the adsorption properties of the solid matrix, the fluid saturations, and temperature. For radon diffusion in porous media, the diffusivity for the other isotopes of radon (e.g., radon-220) has been observed to be comparable to that for the isotope radon-222 (Nazaroff et al. 1988).

Several attempts have been made to correlate the radon diffusion coefficients in porous media (D and D_e) to the radon diffusion coefficient in open air (D_o) and the physical properties of the medium such as the total porosity (n). These attempts have not been conclusive. According to experimental work performed by Currie (1960a, 1960b) and quoted by Nazaroff et al. (1988) and Rolston (1986), the coefficients D and D_o can be correlated by an expression of the following form:

$$\frac{D}{D_o} = \gamma n^{\mu} \tag{7.6}$$

where γ and μ represent measures of pore shape of the soil materials. This empirical relationship could fit data from a wide range of dry porous materials in which the values of γ generally lie between 0.8 and 1.0 and the values of μ lie around 1.0. This empirical relationship is not applicable, however, for very wet soil and strongly aggregate soil (Rolston 1986).

The influence of soil moisture content on the effective diffusion coefficient of radon in soil has been investigated by Rogers and Nielson (1991), who proposed the following expression:

$$D_e = D_o n \exp(-6ns - 6s^{14n}) \tag{7.7}$$

where $D_o = 1.1 \times 10^{-5}$ m^2/s is the radon diffusivity in open air, n is the total soil porosity, and s is the water saturation in the soil (or the fraction of the pore space filled with water).

7.2 MEASUREMENT METHODOLOGY

The diffusivity (or the diffusion coefficient) of radon in soils can be measured by both field and laboratory experiments. In either case, the experimental evaluation of the diffusivity consists in determining the numerical value for the coefficient appearing in Fick's equation. Because of the difficulties in implementing field methods, laboratory methods are generally used to determine the radon diffusivity in porous media and particularly in soil materials.

Variations of the laboratory methods for measuring radon diffusivity in porous media have been developed and as yet no standard (or recommended) method exists. All the various laboratory methods are based on the solution of the mass balance equation that represents the diffusion process in a one-dimensional configuration. Depending on the approximation taken on the time domain for the solution of the diffusion equation, these methods can be separated into two distinct groups: (1) the steady state diffusion method and (2) the transient diffusion method (Nielson 1982).

The steady state method used in the laboratory for the determination of the radon diffusivity in soil material, without a source of radon within it, is based on the solution of a one-dimensional diffusion equation in the x-direction expressed as follows:

$$\frac{d^2C}{dx^2} - \frac{\lambda}{D_e}C = 0 \tag{7.8}$$

This steady state equation is obtained by coupling the one-dimensional Fick's equation:

$$J_e = -D_e\frac{dC}{dx} \tag{7.9}$$

with the one-dimensional, steady-state, continuity equation:

$$\frac{dJ_e}{dx} = -\lambda C \tag{7.10}$$

where J_e is the effective flux density of radon activity (pCi)/($m^2 \cdot s$), C is the concentration of radon activity in the pore space (pCi/m^3), and λ is the radon decay constant (1/s).

A steady state diffusion method for determining the effective radon diffusion coefficient (D_e) in uncontaminated (no radon source) soil materials was implemented by Silker and Kalkwarf (Silker 1981 and Silker and Kalkwarf 1983) on the basis of theoretical developments by Cohen (1979). The apparatus used in this method consists of a column of test soil of known depth, d, which is sealed at one end to an air chamber of known volume containing a radon source with a known and constant strength. The other end of the test soil column is kept open. As a boundary condition for this system, it is assumed that in a steady state situation the effective flux density of radon activity at the bottom of the column, J_{eo}, is constant and uniquely dependent on the strength of the radon source and the geometry of the system. Also, the radon activity concentration at the open end of the soil column is assumed to be negligible (i.e., zero).

On the basis of these assumptions and conditions, the effective radon diffusivity, D_e, can then be evaluated by the following equation (Silker 1983):

$$\lambda d \left(\frac{C_o}{J_{eo}}\right) = \left(\frac{d}{L}\right) \left[\frac{1 - e^{-2\frac{d}{L}}}{1 + e^{-2\frac{d}{L}}}\right] \tag{7.11}$$

where C_o is the radon activity concentration within the air chamber, and L is the radon diffusion length (or relaxation length) parameter within the porous medium, which is defined as follows:

$$L = \sqrt{\frac{D_e}{\lambda}} \tag{7.12}$$

The right side of Equation 7.12 is a well defined function of the parameter ratio d/L and is independent of the measured values of C_o and J_{eo}. The left side of the equation is dependent on the measured results. Therefore, by selecting the size (i.e., thickness) of the soil test sample, d, determining the effective flux density J_{eo} on the basis of the strength of the radon source and the column diameter, and making several measurements of C_o, Equation 7.9 can be graphically or numerically solved for the ratio d/L and subsequently to D_e.

Typically, the soil samples used in the determination of D_e have a cylindrical shape with a height to 10 cm and an inner diameter of 14 cm. After equilibration, the steady state radon concentration in the bottom chamber, C_o, is determined by several measurements taken over a 7-to-14 day period. Each measurement consists of withdrawing about 5 cm^3 of gas from a typical 800-cm^3 bottom chamber and determining the radon concentration by using either a scintillation flask technique (such as a Lucas cell) or charcoal absorption and gamma-ray spectrometry (Silker 1983).

7.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input an effective diffusion coefficient value of radon for three materials: (1) the soil of the cover zone, (2) the soil of the contaminated zone, and (3) the building foundation material (i.e., concrete). The dimensions of these input values of D_e are in units of m^2/s. For each porous material considered, the value of D_e is assumed to be the same for both radon isotopes addressed in RESRAD, namely radon-222 and radon-220.

The effective radon diffusivity values in porous media (soils and concrete included) vary over a wide range of several orders of magnitude depending on the porous material and particularly on its degree of water saturation. Table 7.1 lists representative values of

TABLE 7.1 Effective Diffusion Coefficients for Radon in Unconsolidated Soil Materials and Concrete[a]

Porous Material	D_e (m^2s^{-1})	Comments	Reference
Unconsolidated soil material			
Compacted silty sands	$(3.0 \pm 1.3) \times 10^{-6}$	$n = 0.29$-0.36 $s = 0.05$-0.34	Silker 1983
Compacted clayey sands	$(3.2 \pm 1.5) \times 10^{-6}$	$n = 0.32$-0.39 $s = 0.09$-0.55	Silker 1983
Compacted inorganic clays	$(2.5 \pm 1.0) \times 10^{-6}$	$n = 0.32$-0.43 $s = 0.06$-0.34	Silker 1983
Silty sandy clay	2.7×10^{-6}	m = 1.5% dry weight	Strong et al. 1981
	2.5×10^{-7}	m = 10.5% dry weight	
	6.0×10^{-8}	m = 17.3% dry weight	
Uranium mill tailings	$(5.4$-$7.2) \times 10^{-6}$	m = (0.7-1.5)% dry weight	Strong et al. 1981
Loams	8×10^{-7}	Dry	Tanner 1964
Mud	5.7×10^{-10}	$\theta = 37\%$	Tanner 1964
Concrete	$(1.1$-$4.0) \times 10^{-7}$	$n = 0.11$-0.13	Poffijn et al. 1988
	1.2×10^{-8}	$n = 0.25$	Culot et al. 1976
	3.4×10^{-8}	$n = 0.05$	Culot et al. 1976
	3.3×10^{-8}	$n = 0.068$	Zapalac 1983
	$(0.8$-$8.4) \times 10^{-8}$	-	Stranden 1988
Other materials			
Brick	$(0.8$-$3.0) \times 10^{-7}$	-	Stranden 1988
Gypsum	$(1.0$-$4.0) \times 10^{-6}$	-	Stranden 1988

[a] n = total porosity, s = volumetric water saturation, m = percent water content by weight, and θ = percent volumetric water content.

Source: Adapted from Nazaroff et al. 1988.

effective diffusion coefficients of radon obtained by different researchers for a range of unconsolidated soil materials, concrete, and other building materials. Because of the differences in the experimental methodologies adopted by the various researchers, these experimental data are not easily comparable. Nevertheless, they may give an indication of the expected values of D_e in the field.

Typically, the effective diffusivity of radon in unconsolidated soil material with low moisture content is about 10^{-6} m^2/s. The upper limit is represented by the radon diffusivity in open air, D_o, which is about 1.2×10^{-5} m^2/s. At the lower extreme, in a fully saturated soil material the radon diffusion coefficient may be as low as 10^{-10} m^2/s. In RESRAD, a default value of D_e equal to 3.0×10^{-6} m^{-2}/s was adopted for both the cover and contaminated zones. According to the data presented in Table 7.1, this default value of D_e would represent the average effective radon diffusivity in soils with a lower moisture content and composed of silty and clayey sands. The observed range of variation of D_e in concrete, as presented in Table 7.1, goes from 8.0×10^{-9} to 4.0×10^{-7} m^2/s. A default value of D_e equal to 1.0×10^{-7} m^2/s was adopted in the RESRAD model to represent the effective radon diffusivity in concrete.

The estimation of the values of the effective radon diffusion coefficient (D_e) to be used in RESRAD can be performed at different levels of site-specific accuracy, depending on the amount of information available. For generic use of the code, a set of default values of D_e was defined as 3.0×10^{-6} m^2/s for the cover and contaminated zones, and 1.0×10^{-7} m^2/s for the building foundation (i.e., concrete). If the type of soil materials at the site is known, a slightly more accurate estimation of D_e can be performed with the help of Table 7.1. For most applications, this approach will suffice because of the natural variability of D_e within the soil and building materials at any specific site.

In cases in which there are reasons to suspect that the default values of the effective radon diffusion coefficient (D_e) do not reflect the conditions at a specific site and there is no possibility of measuring D_e, the RESRAD code is able to estimate it internally on the basis of the values of the water saturation (calculated from the volumetric water content) and total porosity, according to Equation 7.7. To implement this option, the user should enter any negative number as an input value of D_e to RESRAD.

For an accurate site-specific estimate of the input data to RESRAD, however, the values of D_e should be measured in either the laboratory or field experiments. Whenever necessary and possible, the measurement of D_e in the soil cover zone (it is assumed that it is not contaminated with radon sources) should be performed in the laboratory by using a method such as the Silker and Kalkwarf (1983) technique.

8 RADON EMANATION COEFFICIENT

8.1 DEFINITION

The radon emanation coefficient is the fraction of the total amount of radon produced by radium decay, which escapes from the soil particles and gets into the pores of the medium. It is also called the emanating power, emanating fraction, release ratio, and escape-to-production ratio. The radon emanation coefficient is a dimensionless parameter and is represented as either a fraction or a percentage.

The two most common radioisotopes of radon gas, radon-222 and radon-220, are generated by a radioactive process of alpha decay from two radium isotopes, radium-226 and radium-224, respectively. Because of the conservation of linear momentum in the alpha-decay process, the newly created radon-222 and radon-220 atoms are left with a kinetic (usually called "recoil") energy of about 86 and 103 keV, respectively (Nazaroff et al. 1988).

Thus, after being generated, the radon atoms tend to move away from their original location until their recoil energy is totally transferred to the medium. Consequently, depending on their original location within the solid phase of the soil, the soil pore distribution, and the soil moisture content, the newly created radon atoms may end up within the same soil particle in which they were created, within the adjacent soil particle because of posterior penetration after escaping from the host soil grain, or within the pore of the medium.

Experimental data reported by several investigators indicate that the radon emanation coefficient is strongly influenced by the moisture content of the medium, particularly within the range of low water saturation (Nazaroff et al. 1988). On the basis of results of this kind, it has been hypothesized that the amount of water present in the pore increases the absorption of the recoil energy of the radon atom passing through it, thus enhancing the chance that the atom will terminate its recoil within the water. Partition equilibrium of radon in the water and air phases in the pore will follow afterwards based on Henry's law.

Although temperature may influence the magnitude of the radon emanation coefficient, it has been demonstrated that within the normal range of temperature variability of surface soils, this effect is of minor importance (Nazaroff et al. 1988).

The radon emanation coefficient, f, is one of the characteristic soil parameters that determine the rate of radon emanation into the pores of the soil matrix. The other soil characteristic parameter in relation to radon production is the concentration of radium (radium-226 and/or radium-224) in the soil particles, C_{Ra}. In RESRAD, the source of radon generation in the pore air or the rate of radon generation and emanation into the soil gas phase (pore air), S, in units of pCi/m^3·s, is calculated as follows:

$$S = f \rho_s \, C_{Ra} \, \lambda \left(\frac{1-n}{n} \right) \tag{8.1}$$

where ρ_s is the soil particle density (kg/m^3), C_{Ra} is the mass concentration of radium (radium-226 or radium-224) in the soil particles (pCi/kg), λ is the radon (radon-222 or radon-220) decay constant (1/s), and n is the total porosity of the contaminated soil.

The values of the radon emanation coefficient (f) in soils depend on the radon isotope being considered, the soil material, and the moisture content. Experimental measurements of f in different soils, rocks, and other materials have been reported by many investigators. Table 8.1 presents a summary of these available data. Because of the differences in the experimental methodologies adopted by the various investigators, these data are not easily comparable. The data are also incomplete in that they do not reflect a rigorous and

TABLE 8.1 Measurements of Emanation Coefficients of Radon (Rn-222 and Rn-220) in Unconsolidated Soils and Other Materials

Material	Number of Samples	Isotope	Emanation Coefficient[a]	Moisture Content	Reference
Unconsolidated soils					
Sand	7	Rn-222	0.14 (0.06-0.18)	Unknown	Sisigina 1974
Sandy loam	7	Rn-222	0.21 (0.10-0.36)	Unknown	Sisigina 1974
Silty loam	7	Rn-222	0.24 (0.18-0.40)	Unknown	Sisigina 1974
(Heavy) loam	12	Rn-222	0.20 (0.17-0.23)	Unknown	Sisigina 1974
Clay	5	Rn-222	0.28 (0.18-0.40)	Unknown	Sisigina 1974
Various soils (Danish)	70	Rn-222	0.22 (0.02-0.70)	0-70% dry wt	Damkjaer and Korsbech 1985
Soil	21	Rn-220	0.30 (0.03-0.55)	Unknown	Barreto 1974
Soil	2	Rn-220	(0.09-0.15)	Oven-dried	Megumi and Mamuro 1974
Other materials					
Uranium ore (crushed)	17	Rn-222	0.28 (0.06-0.55)	Moist, saturated	Thamer et al. 1981
Uranium mill tailings	2	Rn-222	0.14 (0.02-0.36)	Vacuum-dried	Thamer et al. 1981
			(0.29-0.31)	Saturated	Strong and Levins 1982
			(0.067-0.072)	Oven-dried	Strong and Levins 1982

[a] Arithmetic mean (range of values).

Source: Adapted from Nazaroff et al. 1988.

u04583

systematic analysis of the radon emanation coefficient for all radon isotopes in a broad range of soil materials and rocks under different degrees of water saturation. Although incomplete, these available data may give an indication of the expected values of f in the field.

8.2 MEASUREMENT METHODOLOGY

The methodology for measuring the radon emanation coefficient (f) of a porous material contaminated with radium consists basically of measuring the radon concentration in the air within a sealed accumulation chamber, in which a sample of the contaminated soil material has been left for a period of time (around four days) until the radon concentration reaches equilibrium. A detailed description of a variation of this method is presented in Strong and Levins (1982). Their experimental apparatus consisted of an ingrowth (accumulation) chamber, a sampling cylinder, a diaphragm pump, a scintillation cell, and supporting electronics for the radiation measurement.

8.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input a value for the radon emanation coefficient (f) that is related to the soil material of the contaminated zone for the two radon isotopes, radon-222 and radon-220. This parameter is dimensionless and its value should be entered as a fraction (rather than as a percentage).

As shown in Table 8.1, the radon emanation coefficient varies from 0.02-0.70 in soils. The values of f for radon-222 are usually higher than those for radon-220 under the same circumstances. In RESRAD, the adopted default values of the radon emanation coefficient (f) for radon isotopes radon-222 and radon-220 are, respectively, 0.25 and 0.15, in the soil of the contaminated zone. These default values represent approximately the conditions in a silty loam soil with a low moisture content (i.e., not dry).

The estimation of the values of f for radon-222 and radon-220 for use in RESRAD can be performed at different levels of site-specific accuracy, depending on the amount of information available. For generic use of the code, a set of default values for f (0.25 for radon-222 and 0.15 for radon-220) was defined to represent approximately the condition of a silty loam soil with a low moisture content in the contaminated zone.

If the type of soil materials at the site is known, a slightly more accurate estimation of f can be performed with the help of the data in Table 8.1. For most applications, this approach will suffice because of the natural variability of f within the soil of the contaminated zone of any specific site.

In cases in which it is absolutely necessary to have an accurate estimate of f and there are reasons to suspect that the data in Table 8.1 do not reflect the conditions at a specific site, the values of f for radon-222 and radon-220 can be determined experimentally in the laboratory by using the previously mentioned method of Strong and Levins (1982).

9 PRECIPITATION RATE

9.1 DEFINITION

The precipitation rate, P_r, is the average volume of water in the form of rain, snow, hail, or sleet that falls per unit of area and per unit of time at the site. It is measured in units of volume per area per time, or LT^{-1}.

Precipitation is one of the primary processes of the hydrologic cycle, that is, the endless movement of water through the various elements of the environment (oceans, atmosphere, land surface water bodies, and subsurface soil systems). Other processes of the hydrologic cycle include evapotranspiration, infiltration, overland flow (runoff), streamflow, and groundwater flow. Thorough descriptions of these processes have been presented in numerous texts in the hydrology literature (Bedient and Huber 1988; Linsley, Kohler, and Paulhas 1982; Chow 1964).

A simplified description of the hydrologic cycle could start with considering the water vapor contained in the atmosphere, which under appropriate conditions, condenses and precipitates over the oceans and the continental land. The portion of the water that falls over the surface land, that is, precipitation, is subsequently dispersed by following different pathways. Thus, from the precipitation, a parcel of water is retained in the vicinity of the place where the precipitation falls and is then transferred back to the atmosphere through evaporation (i.e., the water changes from a liquid at the soil surface to a vapor) and transpiration (i.e., the indirect loss of water vapor from the soil to the atmosphere through plant tissue). The combined effect of evaporation and transpiration is commonly called evapotranspiration. Another parcel of the precipitation water penetrates the subsurface soil system, that is, the process of infiltration, and is added to the groundwater flow system. Finally, the last parcel of precipitation water (the one that is not transferred back to the atmosphere and does not infiltrate the soil) becomes overland flow, also called surface runoff, and feeds local streams, rivers, or lakes. Both the surface and the subsurface flow of water move toward low elevations and eventually reach the oceans. Evaporation, primarily from the oceans and inland surface waters transfers water vapor back to the atmosphere, thus completing the hydrologic cycle.

The concept of the hydrologic cycle is applicable to a large-scale hydrologic system on earth and can be represented mathematically by a water balance (or budget) equation based on the law of the conservation of matter. The same principle can be applied to any hydrologic system of any scale, whether it is a small basin or a large watershed, to generate a water balance equation that, in its simplest form can be expressed as follows:

$$q_{in}-q_{out} = \frac{ds}{dt} \tag{9.1}$$

where q_{in} is the water inflow rate into the system, q_{out} is the outflow rate, and ds/dt is the change in time of the water stored within the system.

To illustrate the application of the water balance concept, consider a hydrologic system represented by irrigated agricultural land and the movement of water through it. According to the law of the conservation of matter, the variation of ΔS (i.e., the change in the volume of water stored in the soil per unit of surface area of the land) during a given time period ΔT must be equal to the difference between the average inflow rate in time and space (i.e., precipitation, P_r, plus irrigation, IR_r, rates) minus the outflow rate (i.e., infiltration, I_r, plus runoff, R_r, and evapotranspiration, ET_r, rates). The water balance equation for this system could then be represented as follows:

$$\frac{\Delta S}{\Delta T} = (P_r+IR_r) - (I_r+R_r+ET_r) \tag{9.2}$$

where all the inflow and outflow rates are expressed in units of LT^{-1}.

The precipitation over a specific hydrologic system is an erratic process with large fluctuations in the time domain. Consequently, because all the inflow and outflow processes mentioned previously are related to the precipitation, they also present large and erratic variations along the time. As a result, the change in ΔS is highly dependent on the period of time (ΔT) being considered. For short periods, the change in the soil-water storage (ΔS) is also an erratic process and can present relatively large values. However, for a long period, such as an entire season or a whole seasonal cycle of one year, the change in the soil-water storage (ΔS), particularly in the upper part of the soils, is likely to be small in relation to the total water balance of the system (Hillel 1980a).

Thus, considering annual averages of the inflow and outflow water rates in this hypothetical hydrologic system of a generic irrigated agricultural land, the respective water budget equation can be reduced to the following:

$$P_r + IR_r = I_r + R_r + ET_r \qquad (9.3)$$

Except for the infiltration rate, I_r, all other terms of Equation 9.3 can be determined either by direct field measurements or by using specific coefficients derived from soil and other environmental characteristics. The experimental methodology for field measurement of the precipitation, runoff, irrigation, and evapotranspiration rates are described in this handbook (see Sections 9.2, 10.2, 11.2, and 12.2, respectively). Direct field measurement of the deep percolation (infiltration) component of the field water balance has not yet proven to be practical (Hillel 1980a) and, therefore, the infiltration rate is often determined from the other measured components of the equation as follows:

$$I_r = (P_r + IR_r) - (R_r + ET_r) \qquad (9.4)$$

The parameter I_r, or the water infiltration rate, represents the amount of water that percolates through the upper layers of the soil and eventually ends up being added to the groundwater flow underneath the hydrologic system. In the RESRAD model, the parameter I_r is used to calculate the radionuclide leaching from the contaminated zone and the final contamination of the groundwater. The infiltration rate is calculated internally in the code as a function of the precipitation (P_r) and irrigation (IR_r) rates and the runoff (C_r) and evapotranspiration (C_e) coefficients. The latter two parameters are defined respectively, as follows:

$$C_r = \frac{R_r}{P_r} \tag{9.5}$$

and

$$C_e = \frac{ET_r}{(1-C_r)P_r+IR_r} \tag{9.6}$$

Detailed discussion of the runoff (C_r) and evapotranspiration (C_e) coefficients and the irrigation rate (IR_r) are presented elsewhere in this handbook (see Sections 10.1, 12.1, and 11.1, respectively).

Thus, from Equations 9.4, 9.5, and 9.6, the infiltration rate, I_r, can be expressed as follows:

$$I_r = (1-C_e)[(1-C_r)P_r+IR_r] \tag{9.7}$$

The mass balance equation (Equation 9.7) is the one used in RESRAD to calculate the water infiltration rate into the soil.

9.2 MEASUREMENT METHODOLOGY

Measurement of the precipitation rate at a site-specific location can be performed with a precipitation gage, which basically consists of a receptacle with vertical walls and an opening at the top with a specified area. The ratio of the volume collected in the receptacle during a specified period of time to the area of the opening at the top of the receptacle gives the point estimate of the precipitation rate at a specific location and time.

In principle, any receptacle with an open collector area of known dimensions, plus a volume measuring device can be used as a precipitation gage. However, because of some operational features of these devices, unless they are of the same shape and dimensions and similarly exposed, precipitation rate measurements are usually not comparable (Linsley, Kohler, and Paulhus 1982).

The standard precipitation gage adopted by the U.S. National Weather Service has a collector (receiver) with an 8-in. (20.3 cm) diameter and can measure the precipitation to the nearest 0.25 mm. Two types of precipitation gages can be used, recording and nonrecording. The recording gage, the most commonly used, records on a strip of paper, paper punch, or data logger every 0.01 in. (0.0254 cm) of precipitation along the time scale. The recorded data is then reported as an average precipitation rate, total volume, or intensity variation.

According to Bedient and Huber (1988), a network of five to ten gages per 260 km^2 (100 mi^2) is usually required in urban areas to define precipitation variability. The maintenance costs of such networks are high and, therefore, for a particular application, it is usually more convenient to rely on data collected from a national network. Precipitation gage networks designed to provide point estimates of precipitation rates in the United States and its territories are maintained by the U.S. National Weather Service and the U.S. Geological Survey.

Data on the point estimates of precipitation rates obtained from these national networks can be used to estimate the average areal precipitation rate over a specific area. The aerially averaged values of the precipitation rate can be derived by three methods

(Bedient and Humber 1988): arithmetic mean, the Thiessen polygon method, and the isohyetal method.

The arithmetic mean of the point precipitation rates provides the simplest and most straightforward way to obtain an estimate of the areal precipitation rate at a particular site. For cases in which the gages are uniformly distributed and the point values have minimal variations, this method provides satisfactory results.

The Thiessen polygon method consists in aerially weighing the point precipitation from each gage. It is the most commonly used method, although not the most accurate.

The isohyetal method consists in drawing contour lines of equal precipitation (isohytes) and aerially weighing the average precipitation between pairs of contour lines crossing over the area of the site being considered. It is the most accurate among the methods for determining aerially averaged values of the precipitation rate but requires an extensive gage network to draw the isohyets accurately.

A distribution of values of average annual precipitation rates over the U.S. continental territory, transcribed from the *Water Atlas of the United States* (Geraghy 1973), is shown in Figure 9.1.

9.3 RESRAD DATA INPUT REQUIREMENTS

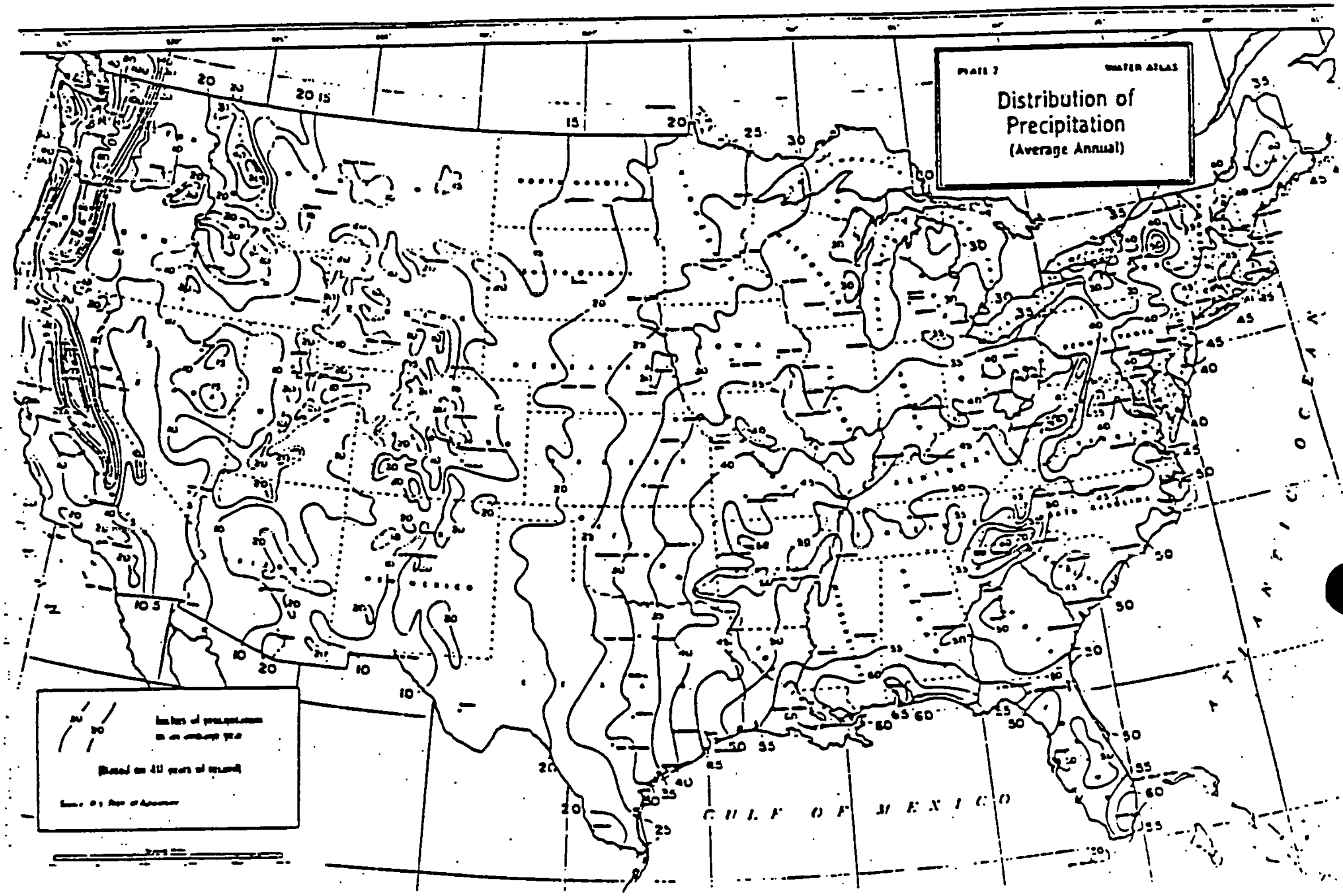
In RESRAD, the user is requested to input a value for the annual average areal precipitation rate (P_r) that is representative of the conditions at the site. The precipitation rate is expressed as an annual average rate in units of m/yr.

The precipitation rate and other input parameters, such as the irrigation rate and the runoff and evapotranspiration coefficients (see Sections 11.1, 10.1, and 12.1, respectively), are used in RESRAD to determine the water infiltration rate, according to Equation 9.7. The water infiltration rate is ultimately used to calculate the radionuclide leaching from the contaminated zone and the subsequent contamination of the underlying groundwater system.

FIGURE 9.1 Distribution of the Average Annual Precipitation Rates in Units of in./yr over the U.S. Continental Territory (Source: Geraghy 1973)

084583

Figure 1: Distribution of the Average Annual Precipitation Rates, in units of (in/yr), over the U.S. continental territory.
Source: [Geraghy, 1973].



For generic use of the code, a default value of the precipitation rate (P_r) equal to 1 m/yr (about 40 in./yr) was adopted in the RESRAD model. This value represents approximately the condition of a relatively humid region. Whenever possible, however, site-specific input data for P_r should be used in the RESRAD calculations.

In cases in which data are available on the annual average point precipitation rates at specific locations in the vicinity of the site, the user can estimate the site-specific areal precipitation rate by using one of three averaging methods described in Section 9.2.

If data on the precipitation rate (P_r) are not being collected at the site or its vicinity, a site-specific estimation of P_r can be obtained from the U.S. National Weather Service or the U.S. Geological Survey network database.

Annual average values of P_r in units of in./yr for the U.S. continental territory, based on 40 years of recording, are presented in the *Water Atlas of the United States* (Geraghy 1973). For most applications, in the absence of data collected specifically at a site, the information provided in this atlas can be used to estimate the site-specific value of P_r at any particular location in the United States.

10 RUNOFF COEFFICIENT

10.1 DEFINITION

The runoff coefficient, C_r, is the fraction of the precipitation that does not infiltrate into the soil and is not transferred back to the atmosphere through evapotranspiration. The runoff coefficient represents the fraction of the precipitation, in excess of the infiltration and evapotranspiration, that becomes surface flow and ends up in either perennial or intermittent surface water bodies. The runoff coefficient is a dimensionless parameter.

In a well-designed and well-operated irrigation system, the flow and the quantity of the irrigation water are controlled by an appropriate drainage system (ditching) and the duration of each application. Consequently, under normal circumstances, the irrigation water does not contribute significantly to the overall average annual runoff. On the basis of these assumptions, the runoff coefficient (C_r) can be defined mathematically by the following expression:

$$C_r = \frac{R_r}{P_r} \tag{10.1}$$

where R_r is the runoff rate and P_r is the precipitation rate. Because R_r is always smaller than (or at the most equal to) P_r, the values of C_r vary within the range of zero to one.

The runoff rate at a specific location is influenced by the morphology of the region, the degree of the slopes, the type of soil material, and the type of soil utilization. Table 10.1 lists values for the runoff coefficient, C_r, under various conditions of soils and soil uses.

10.2 MEASUREMENT METHODOLOGY

A methodology for estimating the runoff coefficient (C_r) is presented in Table 10.1. The value of C_r can be evaluated on the basis of the type of soil and its land utilization at the specific site.

10.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input a value for the average annual runoff coefficient (C_r) that represents the conditions at the site. The runoff coefficient is a dimensionless parameter and its input value should be entered in the form of a decimal fraction rather than as a percentage.

For generic use of the code, a default value of 0.2 was adopted in the RESRAD model for the runoff coefficient. According to the methodology presented in Table 10.1, this default value of C_r represents an agricultural environment of cultivated flat land with a sandy loam type of soil. Whenever possible, however, site-specific information should be used for more accurate use of the code. If site-specific data are not available, Table 10.1 may be used to estimate the average annual runoff coefficient (C_r).

The runoff coefficient and other input parameters such as the precipitation and irrigation rates and the evapotranspiration coefficient (see Sections 9.1, 11.1, and 12.1, respectively) are used in RESRAD to determine the water infiltration rate according to a mass balance equation (Equation 9.7) presented in Section 9.1. The water infiltration rate is ultimately used to calculate the radionuclide leaching from the contaminated zone and the subsequent contamination of the underlying groundwater system.

TABLE 10.1 **Runoff Coefficient Values**

Type of Area	Coefficient	Value
Agricultural Environment[a]		
Flat land with average slopes of 0.3 to 0.9 m/mi	c_1	0.3
Rolling land with average slopes of 4.6 to 6.1 m/mi	c_1	0.2
Hilly land with average slopes of 46 to 76 m/mi	c_1	0.1
Open sandy loam	c_2	0.4
Intermediate combinations of clay and load	c_2	0.2
Tight, impervious clay	c_2	0.1
Woodlands	c_3	0.2
Cultivated lands	c_3	0.1
Urban Environment		
Flat, residential area -- about 30% impervious	C_r	0.4
Moderately steep, residential area -- about 50% impervious	C_r	0.65
Moderately steep, built-up area -- about 70% impervious	C_r	0.8

[a] The runoff coefficient for an agricultural environment is given by $C_r = 1 - c_1 - c_2 - c_3$.

Source: Gilbert et al. 1989.

11 IRRIGATION RATE

11.1 DEFINITION

The irrigation rate, IR_r, is the average volume of water that is added to the soil at the site, per unit of surface area and per unit of time. It is measured in units of volume per area per time, or LT^{-1}. In the RESRAD code, the irrigation rate is expressed as an annual average rate in units of m/yr.

Irrigation is the practice of supplying water artificially to the soil in order to permit agricultural use of the land in an arid region or to compensate for occasional droughts in semidry or semihumid regions. Irrigation is closely dependent on the precipitation rate at the site, in the sense that a well-designed and well-operated irrigation system should optimize the spatial and temporal availability of water in the soil.

As discussed earlier (see Section 9.1), irrigation, together with precipitation, constitute the source of inflow water into a hydrologic system formed by the soil in an agricultural land and the water that circulates through it. The outflow of water in this system is the result of processes such as surface runoff and evapotranspiration and infiltration rates.

The irrigation rate and other input parameters such as the precipitation rate and the runoff and evapotranspiration coefficients (see Sections 9.1, 10.1, and 12.1, respectively) are used in RESRAD to determine the water infiltration rate according to Equation 9.7 in Section 9.1. The water infiltration rate is ultimately used to calculate the radionuclide leaching from the contaminated zone and the subsequent contamination of the underlying groundwater system.

11.2 MEASUREMENT METHODOLOGY

The average annual irrigation rate at a site is determined as a ratio of the total volume of irrigation water added to the field during the year to the surface area of the irrigated land and to the number of hours in a year. This quantity is not measured in the field, per se, but is obtained from the operational activities of the irrigation system.

A well-designed and well-operated irrigation system should be able to supply water to the plants at a rate sufficient to balance their transpiration rate requirements. The objective is to provide water to the soil in a well-distributed manner during the crop season so that the plants can maintain their own hydration without loss of continuity. As long as the water uptake rate from the plants' roots matches the water loss due to the plants' transpiration from their foliage, they can maintain their hydration. As soon as the water intake from the roots becomes lower than the transpiration, however, the plants start losing moisture, resulting in a stressful situation for the development of the crop (Hillel 1980a).

Therefore, the required rate of irrigation at a specific agricultural site is governed by the properties of the soil and the plants, and, fundamentally, by the meteorological conditions at the site. The soil/plant system properties determine the ability of the soil to supply and transmit water to the roots, as well as the ability of the roots to extract water from the soil at a rate needed to overcome transpiration. The meteorological conditions, however, dictate the rate at which the plants are required to transpire and, therefore, the amount of water needed for their survival.

Estimation of the annual irrigation rate at a specific site can be obtained in different ways, depending on the degree of knowledge about the agricultural activities at the site. When information on irrigation systems in operation at the site or at its vicinity is available, the annual irrigation rate can be obtained from operational records. When little information is available on the irrigation procedures at a site, an estimation of the eventual need for the irrigation rate can be obtained on the basis of the measured (or assessed) values of the potential evapotranspiration and precipitation rates and on the basis of an estimated "irrigation efficiency."

Irrigation efficiency is the ratio of the volume of water used consumptively (such as in evapotranspiration) to the total volume of water applied to the field (Hillel 1980a). This definition is similar to the one for the evapotranspiration coefficient, C_e, (see Section 12.1) and can be expressed as follows:

$$\textit{Irrigation Efficiency} = C_e = \frac{ET_r}{(1-C_r)P_r + IR_r} \tag{11.1}$$

Most irrigation projects are inherently inefficient; the average irrigation efficiency is less than 50% (Hillel 1980a). Thus, by assuming a value for the irrigation efficiency (e.g., around 50%) at a specific site with little available data on agricultural activities, and by determining the potential evapotranspiration rate, ET_r, the precipitation rate, P_r, and the runoff coefficient, C_r, the predicted, necessary average annual irrigation rate, IR_r, at the site can be estimated as follows:

$$IR_r = \frac{ET_r}{C_e} - (1-C_r)P_r \tag{11.2}$$

11.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the annual average irrigation rate, IR_r, that represents conditions at the site. The IR_r should be entered in units of meters per year (m/yr).

A default value of IR_r equal to 0.2 m/yr was adopted in the RESRAD model. This value represents approximately the condition of a relatively humid region where only a small amount of irrigation is needed per year. For an arid region, 1 m/yr is considered to be an appropriate generic value for IR_r.

When there is no site-specific information on the annual average irrigation rate, the input value of IR_r at the site can be estimated on the basis of the irrigation efficiency (usually below 50%) and the measurement (or estimation) of another parameter such as the potential evapotranspiration rate, ET_r, the precipitation rate P_r, the runoff coefficient, C_r, and the evapotranspiration coefficient, C_e (i.e, irrigation efficiency), according to the following expression:

$$IR_r = \frac{ET_r}{C_e} - (1 - C_r)P_r \qquad (11.3)$$

12 EVAPOTRANSPIRATION COEFFICIENT

12.1 DEFINITION

Evapotranspiration is one of the processes of the hydrologic cycle and represents the total volume of water that changes phase, that is, from the liquid or solid state to the gaseous state, near the ground surface and is transferred to the atmosphere during a fixed period of time. Consequently, it represents the combination of two separate processes: (1) evaporation (i.e., the change of phase of water near the ground surface and the direct transfer of water vapor from the ground to the atmosphere) and (2) transpiration (i.e., the transfer of water from the ground to the atmosphere through the plants and their foliage).

Evapotranspiration is also called "consumptive use" in the hydrology literature and is defined as the quantity of water used by either cropped or natural vegetation in transpiration or in the building of plant tissue, together with water evaporated from the adjacent soil or from intercepted precipitation, during a fixed period of time (Veihmeyer 1964).

Two parameters need to be defined in relation to the concept of evapotranspiration: (1) the evapotranspiration rate, ET_r, and (2) the evapotranspiration coefficient, C_e.

The evapotranspiration rate, ET_r, is the total volume of water vapor that is transferred to the atmosphere due to the combined effect of evaporation and transpiration, per unit of the ground surface area and per unit of time at the site. It is measured in units of volume per area per time, or LT^{-1}. The evapotranspiration rate is neither required as input data to the RESRAD code, nor is it used implicitly within the model. However, the measured or estimated site-specific value of ET_r is used to estimate the input value of the evapotranspiration coefficient, which is then used in the code. For consistency with other correlated parameters handled in the RESRAD code, the evapotranspiration rate is expressed as an annual average rate in units of m/yr.

The evapotranspiration coefficient, C_e, is the ratio of the total volume of water leaving the ground due to evapotranspiration, ET_r, to the total volume of water available within the root zone of the soil $((1-C_r)P_r + IR_r)$, during a fixed period of time. It can then be expressed as follows:

$$C_e = \frac{ET_r}{(1-C_r)P_r + IR_r} \tag{12.1}$$

where P_r is the precipitation rate (m/yr), IR_r is the irrigation rate (m/yr), and C_r is the runoff coefficient (dimensionless). (All these parameters are defined in this handbook; see Sections 9.1, 11.1, and 10.1, respectively.)

In well-irrigated agricultural land, transpiration predominates over evaporation in composing the total evaporation. Under these circumstances, the evapotranspiration coefficient represents the efficiency by which the water available in the root zone of the soil is actually transferred through the plant system and into the atmosphere. Thus, for cultivated land, the evapotranspiration coefficient (C_e) is also called the "irrigation efficiency." According to Hillel (1980a), most irrigation projects are inherently inefficient, and although irrigation efficiencies of 80% to 90% can be achieved in actual practice with proper water management, the average irrigation efficiency is less than 50%.

The evapotranspiration process is fundamentally governed by the meteorological conditions at the site, as well a by the properties of the soil/plant system. Meteorological parameters such as air temperature, wind speed, atmospheric pressure, air humidity, and exposure to the sun, all have an important role in determining the evapotranspirational demand at a specific location and time of the year. However, it is the amount of water available in the root zone of the soil that limits the occurrence of the evapotranspiration process. Thus, the power of the atmosphere to extract water from the ground surface because of evaporation decreases as the moisture content of the soil decreases. The smaller the moisture content is, the more strongly the water is bound to the porous matrix of the soil because of capillarity, and thus more energy is needed to extract it. Transpiration is also limited by the availability of water at the root zone, the ability of the soil to supply and transmit water toward the root zone, and the ability of the root system to absorb water from the soil in its vicinity. Below a certain value of soil moisture called the wilting point, the roots of the plants are not able to extract water from the soil, and the transpiration process is broken, resulting in dehydration and wilting. Therefore, as a combination of evaporation and transpiration, the actual evapotranspiration at a specific site depends on the external climatic conditions and on the type and density of vegetation covering the ground surface as well as on soil moisture, root distribution, and other soil properties.

The concept of the "potential evapotranspiration rate", ET_{pr}, has been introduced into the hydrology literature to represent the so called "climatic demand" for water, independently of the transient properties of the soil (Hillel 1980a). As such, the potential evapotranspiration rate, ET_{pr} (or the evaporating power of the atmosphere), is defined as the evapotranspiration rate that occurs on the ground of a land area totally covered with vegetation and where sufficient water is continuously available for the needs of the plants. The actual evapotranspiration rate, ET_r, is then a function of the potential evapotranspiration rate, ET_{pr}, and the quantity of water available in the root zone of the soil. Where there is an excess of water in the root zone, the value of ET_r is at its maximum equal to ET_{pr}, and the excess water infiltrates the soil toward the groundwater system. During a water shortage period, however, the value of ET_r becomes lower than ET_{pr}, with no resulting infiltration. A distribution of average annual potential evapotranspiration rates over the

U.S. continental territory, transcribed from the *Water Atlas of the United States* (Geraghy 1975), is shown in Figure 12.1.

The evapotranspiration coefficient and other input parameters such as the precipitation rate, the irrigation rate, and the runoff coefficient are used in RESRAD to determine the water infiltration rate, according to Equation 9.7 in Section 9.1. The water infiltration rate is ultimately used to calculate the radionuclide leaching from the contaminated zone and the subsequent contamination of the underlying groundwater system.

12.2 MEASUREMENT METHODOLOGY

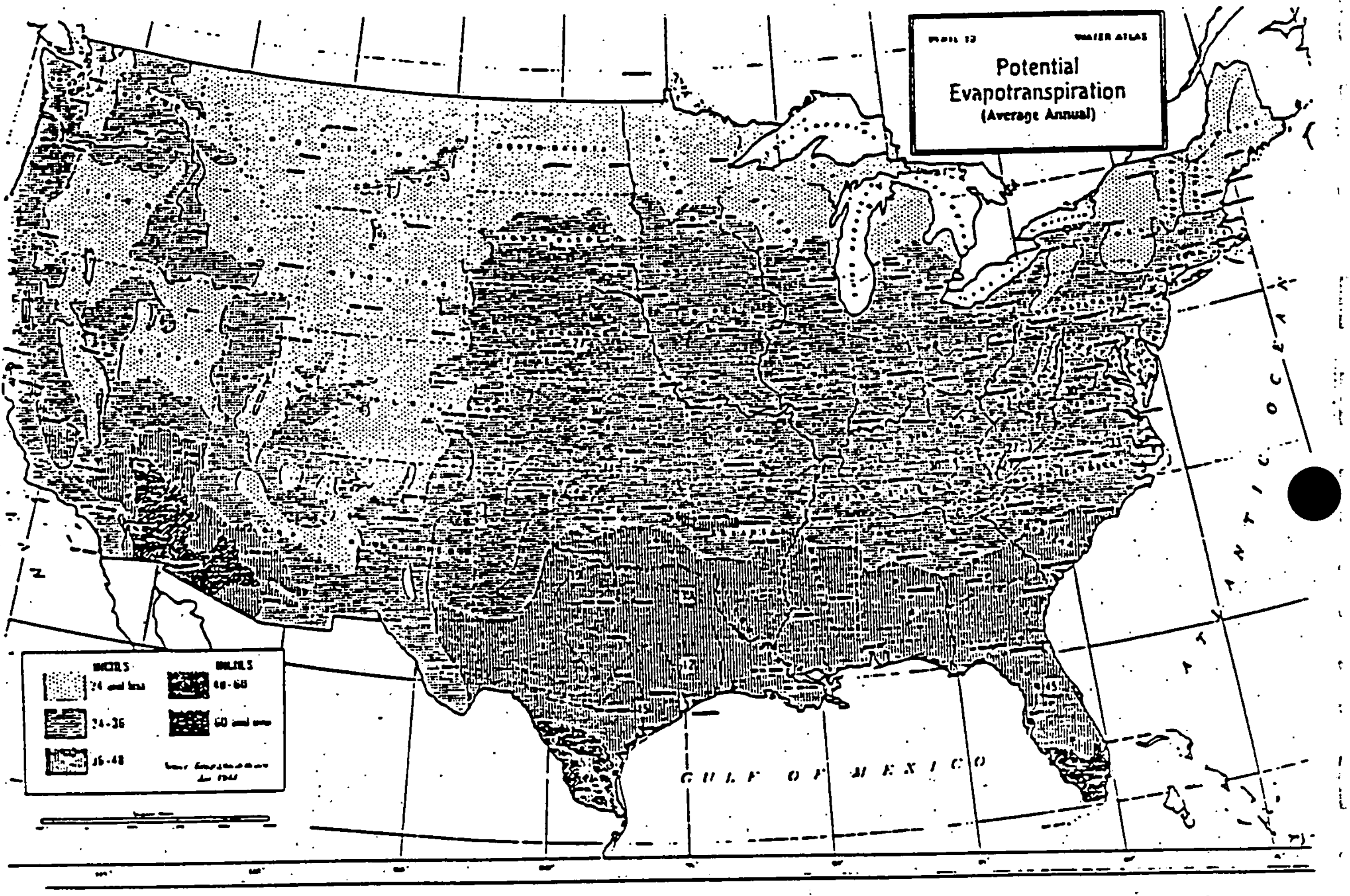
Estimation of the evapotranspiration coefficient, C_e, (to be used as input data to the RESRAD code), should be obtained from measured (or otherwise estimated) values of the evapotranspiration rate, ET_r, the precipitation rate, P_r, the irrigation rate, IR_r, and the runoff coefficient, C_r, according to Equation 12.1.

There are many methods of measuring or estimating the actual (ET_r) and the potential (ET_{pr}) evapotranspiration rate. However, no one method can be used for all purposes (Veihmeyer 1964). Most of the methods used for estimating ET_r can also be used for estimating ET_{pr}, provided that the available water supply is sufficient for the area under observation during the duration of the test. These methods can be classified into three broad categories: (1) the theoretical approach, based on physical principles governing the process; (2) the analytical approach, based on conservation principles, either as a mass or as an energy balance; and (3) the empirical approach, based on experimental results expressing the correlation between measured evapotranspiration and local climatic conditions.

A generic description of various methods used for measuring evapotranspiration can be found in Veihmeyer 1964. The methods available are (1) soil-moisture sampling, (2) lysimeter measurement, (3) inflow-outflow measurements, (4) integration method,

FIGURE 12.1 **Distribution of Average Annual Potential Evapotranspiration Rates in Units of in./yr over the U.S. Continental Territory (Source: Geraghy 1973)**

Figure 1:Distribution of the Average Annual Potential Evapotranspiration Rates, in units of (in/yr), over the U.S. continental territory. Source: [Geraghy, 1973].



(5) energy balance, (6) vapor transfer, and (7) groundwater fluctuations. For example, the lysimeter method consists of using a large barrel (also called a tank or evapotranspirometer) with about a 1-m diameter and a 2-m depth that is filled with soil and buried in the ground so its top matches the ground surface. Individual crops and/or natural vegetation are grown on and around the lysimeter. The evapotranspiration rate can then be determined on the basis of the mass balance by measuring the infiltration flux seeping out of the bottom of the lysimeter and the rainfall rate. The loss of water necessary to maintain satisfactory plant growth represents the evapotranspiration. When operated properly, the lysimeter can provide reasonably reliable values of potential evapotranspiration. However, reliable measurements of actual evapotranspiration (particularly when it is much lower than the potential) are rarely attained because of the difficulty in maintaining comparable soil moisture and vegetation cover conditions on and around the lysimeter (Linsley 1982).

Because of the inherent difficulties of field methods for measuring evapotranspiration, several empirical formulas have been developed to relate the potential evapotranspiration to some readily available climatic data, such as temperature, sunshine, wind velocity, and so forth. A list of typical evapotranspiration equations is presented by Veihmeyer in Table 11.2 of the *Handbook of Applied Hydrology* (Veihmeyer 1964, pp. 11-27).

12.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input a value for the annual average evapotranspiration coefficient, C_e, representative of the conditions at the site. The input value of C_e is given in dimensionless units.

In the process of estimating the value of C_e as an input value for RESRAD, it is assumed that the cultivated land at the site under consideration is maintained with the necessary level of moisture content in the soil for the growth and development of the crop. This condition is achieved either by the natural precipitation rate or by the planned combination of precipitation and irrigation rates. In other words, it is assumed that the required moisture content for potential evapotranspiration based on the annual average is maintained in the soil.

Therefore, the estimation of the input value of C_e for some site-specific conditions is based on a previously measured (or otherwise determined) value of the potential evapotranspiration, ET_{pr}, the precipitation rate, P_r, the irrigation rate, IR_r, and the runoff coefficient, C_r, according to the definition of C_e presented in Equation 12.1.

A default value of C_e equal to 0.6 (dimensionless) was adopted in the RESRAD model. This value represents the condition of 60% efficiency in the irrigation process at a generic site. Under this condition, 60% of the water available in the root zone of the soil is transferred to the atmosphere, and 40% of the water infiltrates the soil and percolates toward the aquifer system. Whenever possible, however, site-specific input data for C_e should be used in the RESRAD calculations.

Field measurements of the average annual evapotranspiration rate, ET_r, usually are expensive and time-consuming. Therefore, if data on ET_r have not been collected at the site or its vicinity, a site-specific estimation of ET_r (and ultimately of C_e) should be obtained from information in the literature. The user can also use the annual average values of potential evapotranspiration for the U.S. continental territory (Figure 12.1), which are presented in the *Water Atlas of the United States* (Geraghy 1973). The information provided in this atlas can be used to estimate the site-specific value of ET_r (and ultimately of C_e) at any particular location in the United States. For most applications, in the absence of site-specific data, this approach should suffice because of the intrinsic uncertainties associated with the model itself and the natural variability of the potential evapotranspiration at any site.

13 SOIL-SPECIFIC EXPONENTIAL b PARAMETER

13.1 DEFINITION

The soil-specific exponential b parameter is one of several hydrological parameters used to calculate radionuclide leaching from the contaminated zone. (See also precipitation rate, irrigation rate, runoff coefficient, evapotranspiration coefficient, hydraulic conductivity, and soil porosity). The soil-specific b parameter is an empirical and dimensionless parameter that is used to evaluate the volumetric water saturation, S, of the soil, according to a soil characteristics function called the conductivity function (i.e., the relationship between the unsaturated hydraulic conductivity, K, and the saturation, S).

It has been suggested that a power function is an acceptable form of representing the conductivity function. As cited by Clapp and Hornberger (1978), Campbell (1974) derived a partly empirical and partly theoretical conductivity function on the basis of the power function model; this function proved to be reasonably accurate over a large number of cases. Campbell suggested the following power expression to represent the working relationship for the conductivity function:

$$k = S^{(2b+3)} \tag{13.1}$$

where k is the relative conductivity (or relative permeability, dimensionless), S is the saturation (dimensionless), and b is the fitting parameter, called the soil-specific exponential parameter, which must be determined experimentally.

The relative permeability, k, at any location in the unsaturated zone, is defined as a ratio of the unsaturated hydraulic conductivity, K, at that point, to the saturated hydraulic conductivity, K_{sat}. Thus, k can be expressed as follows:

$$k = \frac{K}{K_{sat}} \tag{13.2}$$

Substituting the definition of the relative permeability k into Equation 13.1 yields:

$$\frac{K}{K_{sat}} = S^{(2b+3)} \tag{13.3}$$

or

$$S = \left(\frac{K}{K_{sat}}\right)^{\left(\frac{1}{2b+3}\right)} \tag{13.4}$$

In downward water infiltration into the unsaturated upper layer of the soil, the infiltration rate, I_r, (see also precipitation rate) can be approximated by the unsaturated hydraulic conductivity, K (Hillel 1980a). Therefore, substituting I_r for K in Equation 13.4 yields:

$$S = \left(\frac{I_r}{K_{sat}}\right)^{\left(\frac{1}{2b+3}\right)} \tag{13.5}$$

Equation 13.5 is used internally in the RESRAD model to evaluate the volumetric water saturation, S, in all unsaturated regions of the soil system. According to Equation 13.5, under unsaturated infiltration conditions, the saturation S is a function of the infiltration rate I_r, the saturated hydraulic conductivity K_{sat}, and the texture of the soil, as determined by the fitting parameter b. When the medium is fully saturated, I_r becomes equal to K_{sat} and equals unity.

13.2 MEASUREMENT METHODOLOGY

The soil-specific b parameter is an empirical fitting parameter and, therefore, must be determined experimentally. For each type of soil, the best estimate of b can be obtained by adjusting the best-fit values of each soil to an experimentally determined curve of relative permeability versus saturation, according to the power function model proposed above (Equation 13.1).

Determining the conductivity function of a soil sample experimentally by measuring the relative permeability and the saturation is not an easy laboratory task because of many technical and procedural difficulties. Yet some data have been reported in the literature that demonstrate reasonable agreement with the proposed model. For example, Clapp and Hornberger (1978) have reported that Campbell's model for the conductivity function has proved to be acceptable under different conditions of soil saturation over a wide range of b values (0.17 - 13.6) and even for values of saturation, s, near unity (i.e., full saturation). Table 13.1 lists representative values of the soil-specific exponential b parameter for various soil textures.

13.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to define an input value for the soil-specific b parameter for (1) the contaminated zone, (2) the unsaturated zone strata, and (3) the saturated zone. Input for the saturated zone b parameter will only be required if the water table drop rate (see Section 18.1) is greater than zero.

Reported measured data indicate that values of b vary within the range of 0.17 to 13.6 (Clapp and Hornberger 1978). A default value of 6 was adopted in the RESRAD model. This value represents approximately the condition of a silty loam soil material. Whenever possible, however, site-specific input data for b should be used in the RESRAD calculation.

A relatively more accurate value of parameter b for site-specific soil materials can be obtained from the data listed in Table 13.1. For most applications this approach should suffice because of the difficulties in obtaining laboratory determinations of the soil conductivity function.

TABLE 13.1 Representative Values of Soil-Specific Exponential b Parameter

Texture	Soil Specific Exponential Parameter, b
Sand	4.05
Loamy sand	4.38
Sandy loam	4.90
Silty loam	5.30
Loam	5.39
Sandy clay loam	7.12
Silty clay loam	7.75
Clay loam	8.52
Sandy clay	10.40
Silty clay	10.40
Clay	11.40

Source: Clapp and Hornberger 1978.

14 EROSION RATE

14.1 DEFINITION

The erosion rate is the average volume of soil material that is removed from one place to another by running water, waves and currents, wind, or moving ice per unit of ground surface area and per unit of time. The erosion rate represents the average depth of soil that is removed from the ground surface per unit of time at the site and is expressed in units of LT^{-1}.

14.2 MEASUREMENT METHODOLOGY

Erosion rates can be estimated by means of the Universal Soil Loss Equation (USLE), an empirical model that has been developed for predicting the rate of soil loss by sheet and rill erosion. If sufficient site-specific data are available, a site-specific erosion rate can be calculated. Details of the calculation are discussed by Wischmeier and Smith (1978) and Foster (1979). Estimates based on the range of erosion rates for typical sites in humid areas east of the Mississippi River (based on model site calculations for locations in New York, New Jersey, Ohio, and Missouri) can also be used (Knight 1983). For example, for a site with a 2% slope, these model calculations predict a range of 8×10^{-7} to 3×10^{-6} m/yr for natural succession vegetation, 1×10^{-5} to 6×10^{-5} m/yr for permanent pasture, and 9×10^{-5} to 6×10^{-4} m/yr for row-crop agriculture. The rate increases by a factor of about 3 for a 5% slope, 7 for a 10% slope, and 15 for a 15% slope. If these generic values are used for a farm/garden scenario in which the dose contribution from food ingestion pathways is expected to be significant, an erosion rate of 6×10^{-4} m/yr should be assumed for a site with a 2% slope. This would lead to erosion of 0.6 m of soil in 1,000 years. A proportionately higher erosion rate must be used if the slope exceeds 2%. An erosion rate of 6×10^{-5} m/yr, leading to erosion of 0.06 m of soil in 1,000 years, can be used for a site with a 2% slope if it can be reasonably shown that the farm/garden scenario is unreasonable; for example, if the site is, and will likely continue to be, unsuitable for agricultural use.

Erosion rates are more difficult to estimate for arid than for humid sites. Although water erosion is generally more important than wind erosion, the latter can also be significant. Water erosion in the West is more difficult to estimate because it is likely to be due to infrequent heavy rainfalls for which the empirical constants used in the USLE may not be applicable. Long-term erosion rates are generally lower for sites in arid locations than for sites in humid locations; hence, values estimated in the manner described above for humid sites can be used because they can be expected to provide conservative values.

14.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input a value for the annual average erosion rate for the cover zone and the contaminated zone. The dimensions of these input values of the erosion rate are given in units of m/yr.

For generic use of the code, a default value of the annual erosion rate equal to 0.001 m/yr was adopted in the RESRAD model for both the cover and the contaminated zones. This default value should suffice for screening estimates. For a particular site, however, a more accurate site-specific estimation of the erosion rates for both the cover and the contaminated zones should be attempted. The erosion rate of the contaminated zone only becomes significant if and when the cover zone is completely eroded, thus exposing the contaminated zone to the erosive effects of the environmental elements.

A site-specific estimation of the erosion rate for the cover and contaminated zones can be performed by means of the USLE.

15 HYDRAULIC GRADIENT

15.1 DEFINITION

The hydraulic gradient is the change in hydraulic head per unit of distance of the groundwater flow in a given direction. The hydraulic gradient, i, is expressed as follows:

$$i = \frac{h_1 - h_2}{L} \tag{15.1}$$

where h_1 and h_2 represent the hydraulic head at points 1 and 2, respectively, and L is the distance between these two points. Mathematically, the hydraulic gradient is a vector that can be expressed as grad h. The norm of the vector represents the maximum slope of the hydraulic gradient, and its orientation represents the direction along the maximum slope. The hydraulic gradient is a dimensionless parameter customarily represented as a fraction rather than as a percentage.

In an unconfined (water table) aquifer, the horizontal hydraulic gradient of groundwater flow is approximately the slope of the water table. In a confined aquifer, it represents the difference in potentiometric surfaces over a unit distance. The potentiometric surface is the elevation to which water rises in a well that taps a confined aquifer. It is an imaginary surface analogous to a water table. In general, the hydraulic gradient of groundwater flow in a highly permeable geologic material, such as sand or gravel, is far less than that in a geologic material with a low permeability, such as silt and clay.

15.2 MEASUREMENT METHODOLOGY

The hydraulic head at a point in the saturated zone can be measured in the field by installing a piezometric nest at the site. A piezometer is basically a tube or pipe long enough to be introduced through the unsaturated zone down into the saturated zone. Its walls must be completely sealed along all its length but it must be open to the atmosphere at the top and to the water flow at the bottom. The water level measured inside the piezometer, as compared with a defined reference level (such as mean sea level), gives the hydraulic head of the aquifer at the point of measurement.

The distribution of the hydraulic head in a groundwater system is actually three-dimensional. Thus with the installation of three or more piezometers spatially distributed in an aquifer, it is possible to determine the spatial distribution of the hydraulic head at the site. By knowing the distances between the piezometers, the hydraulic gradient of the dominant aquifer flow at the site can be evaluated. A detailed description of piezometer nests can be found in Freeze and Cherry (1979).

15.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input a value for the hydraulic gradient in the dominant groundwater flow direction in the underlying aquifer at the site. This parameter is dimensionless and should be entered as a decimal fraction rather than as a percentage.

For generic use of the code, a default value of 0.02 was adopted for the hydraulic gradient in the RESRAD model. Because the hydraulic gradient varies significantly from one site to another, whenever possible, site-specific information should be used for more accurate use of the code.

Site-specific data on the hydraulic gradient and the general flow pattern of the groundwater system at the site can be obtained by installing a piezometric nest in the area, as suggested above.

16 LENGTH OF CONTAMINATED ZONE PARALLEL TO THE AQUIFER FLOW

16.1 DEFINITION

The length, ℓ, of the contaminated zone parallel to the aquifer flow is the maximum horizontal distance measured in the contaminated zone, from its upgradient edge to the downgradient edge, along the direction of the groundwater flow in the underlying aquifer.

The parameter ℓ is used in RESRAD to evaluate the dilution of the contaminated inflow water (which percolates the contaminated zone vertically and reaches the aquifer underneath) by the uncontaminated inflow groundwater in the Nondispersion Model for a well located near the contaminated zone.

16.2 MEASUREMENT METHODOLOGY

To evaluate the value of parameter ℓ at a specific site, it is first necessary to determine the hydraulic gradient of groundwater flow at the site. As described in Section 15.2, the groundwater flow direction in the aquifer can be determined locally by installing a piezometric nest composed of three or more piezometers spatially distributed throughout the hydrogeological system. With a known groundwater flow direction and the horizontal extension of the contaminated zone, the parameter ℓ can be determined by measuring the largest horizontal length of the contaminated zone parallel to the groundwater flow direction.

16.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value of ℓ, the length of the contaminated zone parallel to the groundwater flow, that represents the conditions at the site. The dimensions of ℓ should be entered in units of meters (m).

A default value of 100 m was adopted in the RESRAD model for parameter ℓ. The default value of 100 m is the square root of the default contaminated zone area of 10,000 m^2.

Whenever possible, however, site-specific information should be used for more accurate use of the code.

17 WATERSHED AREA FOR NEARBY STREAM OR POND

17.1 DEFINITION

A watershed is a region contoured by an imaginary line connecting ridge or summit of high land and drained by or draining into a river, river system, or a body of water such as a lake or pond. The watershed area is the surface area of the draining region above the discharge measuring points. This parameter is expressed in units of L^2. In the RESRAD code, the watershed area parameter represents the area of the region draining into the nearby stream or pond located at the vicinity of the site.

The watershed area parameter is used in the RESRAD model to evaluate the dilution factor for the contamination of the water at the nearby stream or pond as it gets mixed with the inflow of water from the contaminated aquifer. Thus, the evaluation of the dilution factor for the ground/surface water pathway is based on the following assumptions (Gilbert et al. 1989): (1) the nearby body of water is a pond, (2) the inflow and outflow of water in the pond are in equilibrium, (3) the average annual inflow of radioactivity into the pond is equal to the average annual quantity of radioactivity that is leached from the contaminated zone into the groundwater system, and (4) the infiltrating water flow through the contaminated zone is vertically downward. Under these conditions and assumptions, the dilution factor is then defined as the ratio of the average annual volume of water that percolates through the contaminated zone to the average annual total inflow of water into the pond. More specifically, the dilution factor is calculated internally in the code as the ratio of the contaminated zone area (AREA) to the watershed area (WAREA).

17.2 MEASUREMENT METHODOLOGY

The area of the watershed draining toward the pond located at the vicinity of the site can be evaluated by using a small-scale morphologic map of the region.

17.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is requested to input a value for the area of the watershed region draining into the stream or pond located at the vicinity of the site. The dimensions of the watershed area should be entered in units of square meters (m^2).

A default value of one million (1×10^6) m^2 for the watershed area was adopted in the RESRAD model. Whenever possible, however, site-specific information should be used for more accurate use of the code.

Site-specific information on the watershed area can be obtained from small-scale hydrological and morphological maps covering the region under study. In the RESRAD code, the watershed area must be larger than or equal to the area of the contaminated zone. The code will issue a warning if this condition is violated and will not proceed with the calculations until it is corrected.

18 WATER TABLE DROP RATE

18.1 DEFINITION

The water table drop rate is the rate, in length per time (LT^{-1}), at which the depth of the water table is lowered. The level of the water table in a groundwater system fluctuates seasonally due to the erratically temporal variations of the processes involved in the hydrologic cycle (see Section 9.1), as well as extra use of the water from the system. Under normal circumstances, the level of the water table is approximately stationary if averaged over long periods of time such as one year. For unusually high consumptive use of groundwater in the region, however, the water table may experience a significant drop during the annual period. In these cases, the average annual water table drop rate is not zero and results in the creation of or an increase in the unsaturated zone thickness.

18.2 MEASUREMENT METHODOLOGY

The site-specific water table drop rate can be estimated by observing the change of the water level of a monitoring well appropriately installed at the site. It can also be estimated by consulting water table records of past decades.

18.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the average annual water table drop rate that represents conditions at the site. The dimensions of the water table drop rate should be given in units of meters per year (m/yr).

A default value of 0.001 m/yr was adopted in the RESRAD model for the water table drop rate. This value is the same as the default value used for erosion rate Whenever possible, however, site-specific information should be used for more accurate use of the code.

19 WELL-PUMP INTAKE DEPTH

19.1 DEFINITION

The parameter well-pump intake depth is the screened depth of a well within the aquifer (the saturated zone). The well-pump intake depth is measured in units of length (L).

19.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the well-pump intake depth that represents conditions at the site. Its dimensions should be given in units of meters (m).

A default value of 10 m was adopted in the RESRAD model for the well-pump intake depth. For more accurate use of the code, however, site-specific data should be used whenever possible.

20 RADON VERTICAL DIMENSION OF MIXING

20.1 DEFINITION

The radon vertical dimension of mixing is the height of the atmospheric boundary layer near the ground surface, into which the radon gas that emanates from the ground is uniformly mixed in the outdoor air. This parameter is measured in units of length (L).

20.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the radon vertical dimension of mixing that represents conditions at the site. It dimensions should be given in units of meters (m).

A default value of 2 m was adopted in the RESRAD model for the radon vertical dimension of mixing. For more accurate use of the code, however, site-specific data should be used whenever possible.

21 AVERAGE ANNUAL WIND SPEED

21.1 DEFINITION

The average annual wind speed is the overall average of the wind speed, measured near the ground, in a one-year period. This parameter is measured in units of length per time (LT^{-1}).

21.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the average annual wind speed that represents conditions at the site. The dimensions of the wind speed input should be given in units of meters per second (m/s).

A default value of 2 m/s was adopted in the RESRAD model for the average annual wind speed. For more accurate use of the code, however, site-specific data should be used whenever possible.

Site-specific information on the time distribution of the wind speed and direction at the site can be obtained with the installation of a simple meteorological station instrumented with an anemometer (for measuring the wind speed) and wind vanes (for measuring wind direction). Although simple, the installation, operation, and maintenance of such systems are time-consuming and require the attention of trained personnel. A more general estimation of the average wind speed at a site can be obtained from other meteorological information systems in the area (such as at a commercial airport). For most applications, in the absence of site-specific data, this approach should suffice because of the intrinsic uncertainties associated with the natural variability of the wind speed and direction at the site.

22 AVERAGE BUILDING AIR EXCHANGE RATE

22.1 DEFINITION

The building air exchange (or ventilation) rate is the total volume of air contained in the building that is being exchanged with outside air per unit of time. This parameter expresses the rate at which the total air contained within the building is replaced (or renewed) per unit of time and is measured in units of inverse time (T^{-1}). For example, a building with a ventilation rate of 1 h^{-1} has its volume of air replaced once each hour, on the average.

Important factors affecting the ventilation rate include construction and operating features of the building (i.e., age, window and door weatherproofing, existence of unbalanced mechanical ventilation, the use of fireplaces, etc.), as well as environmental conditions (i.e., atmospheric pressure, temperature, and wind speed and direction). The total ventilation rate is based on three factors (Nero 1988): (1) the infiltration of air through small openings and imperfections in the building structure; (2) the exchange of air through windows, doors, or any other large openings that are kept partially or temporarily open; and (3) the mechanically supplied ventilation due to the operation of exhaust fans or other similar systems. Each of these factors varies significantly along time and, consequently, the total ventilation rate in a house is also strongly time dependent. In the United States, the average ventilation rate during the seasons when houses are kept closed lies within the range of 0.1 to 1.0 h^{-1}.

22.2 MEASUREMENT METHODOLOGY

Measurement of the ventilation rate in a building can be accomplished directly by injecting a tracer gas, sulfur hexafluoride (SF_6), into the house and then, after a mixing time, measuring the gas concentration as a function of time by using an infrared analyzer. The ventilation rate is equal to the rate of decay of the tracer concentration (Nero 1988).

Another available passive measurement technique consists of releasing a gaseous tracer from a small source at a constant rate inside the building. A collecting monitor, consisting of a diffusive tube and an absorber, measures the average concentration during the time the system is in operation. The measured concentration is then proportional to the inverse of the ventilation rate. Further references for these ventilation rate measurement techniques, as well as some predictive quantitative models, can be found in Nero 1988 and Nazaroff et al. 1988.

22.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the average building air exchange rate that represents conditions at the site. The dimensions of the ventilation rate should be given in units of h^{-1}.

For generic use of the code, a default value of 0.5 h^{-1} was adopted in the RESRAD model for the average building air exchange rate. For more accurate use of the code, however, site-specific data should be used whenever possible.

23 BUILDING ROOM HEIGHT

23.1 DEFINITION

The building room height expresses the average height of the house. More specifically, it is defined as the ratio of the volume of the total internal space of the building to the internal area of its floor surface. This parameter is measured in units of length (L). For one-pavement houses, the values for the building room height typically lie within the range of 2.2 to 3.0 m.

23.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the building room height that represents conditions at the site. The dimensions of the input value of the room height should be given in units of meters (m).

For generic use of the code, a default value of 2.5 m was adopted in the RESRAD model for the building room height. For more accurate use of the code, however, site-specific data should be used whenever possible.

24 BUILDING INDOOR AREA FACTOR

24.1 DEFINITION

The building indoor area factor is the fraction of the floor area built on the contaminated area.

24.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the default value of 1.0 should be sufficient for most cases. Values greater than 1.0 indicate a contribution from walls extending into the contaminated zone.

If -1 is entered, the code will calculate a time-dependent area factor on the basis of an assumed floor area of 100 m^2 and the amount of wall area extending into the contaminated zone. For example, if the walls extend to a depth of 0.5 m into the contaminated zone, the building indoor area factor is equal to $1+0.5*4/(100^{0.5})$, or 1.2. The building indoor area factor is time dependent because of soil erosion of the contaminated zone.

25 THICKNESS OF UNCONTAMINATED UNSATURATED ZONE

25.1 DEFINITION

The uncontaminated unsaturated zone is the portion of the uncontaminated zone that lies below the bottom of the contaminated zone and above the water table. The RESRAD code has provisions for up to five different horizontal strata within this zone. Each stratum is characterized by six radionuclide-independent parameters: (1) the thickness of the layer, (2) soil density, (3) total porosity, (4) effective porosity, (5) soil-specific b parameter, and (6) hydraulic conductivity.

25.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for each stratum used in the calculation. Entering a nonzero thickness for a stratum activates that stratum and, similarly, changing the thickness to zero deletes the stratum. Default values are supplied by the code for all parameters of an active stratum; however, the use of site-specific data is strongly recommended.

26 BUILDING FOUNDATION THICKNESS

26.1 DEFINITION

The building foundation thickness is the average thickness of the building shell structure in the subsurface of the soil. Typical values lie around 0.15 m.

26.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the building foundation thickness that represents conditions at the site. The dimensions of the input value of the building foundation thickness should be given in units of meters (m).

A default value of 0.15 m was adopted in the RESRAD model for the building foundation thickness. For more accurate use of the code, however, site-specific data should be used whenever possible.

27 FOUNDATION DEPTH BELOW GROUND SURFACE

27.1 DEFINITION

The foundation depth below ground surface is the vertical distance in the soil from the very bottom of the basement floor slab to the ground surface. Typical values lie within the range of 0.0 to 3.0 m.

27.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the foundation depth below the ground surface that represents conditions at the site. The dimensions of the input value for the foundation depth should be given in units of meters (m).

A default value of 0.5 m was adopted in the RESRAD model for the foundation depth. For more accurate use of the code, however, site-specific data should be used whenever possible.

If a negative value is entered, the absolute value will be adjusted (if needed) so that the foundation depth will not extend into the contaminated zone. Thus, because of erosion of the cover and contaminated zones, the foundation depth can be time dependent and less than the (absolute) specified value.

28 FRACTION OF TIME SPENT INDOORS

28.1 DEFINITION

The fraction of time spent indoors is the average fraction of time during which an individual stays inside the house. A typical value lies around 0.5 (dimensionless).

28.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the fraction of time spent indoors that represents conditions at the site. This is a dimensionless parameter and should be entered as a decimal fraction rather than as a percentage.

A default value of 0.5 was adopted in RESRAD for the fraction of time spent indoors. Fore more accurate use of the code, however, site-specific information should be used whenever possible.

29 FRACTION OF TIME SPENT OUTDOORS

29.1 DEFINITION

The fraction of time spent outdoors is the average fraction of time during which an individual stays outdoors on the site. This is a dimensionless parameter and the typical value lies around 0.25.

29.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the fraction of time spent outdoors that represents conditions at the site. It should be entered as a decimal fraction rather than as a percentage.

A default value of 0.25 was adopted in RESRAD for the fraction of time spent outdoors. For more accurate use of the code, however, site-specific information should be used whenever possible.

30 AREA OF CONTAMINATED ZONE

30.1 DEFINITION

A contaminated zone is a compact area that contains the locations of soil samples with radionuclide concentrations clearly exceeding background levels. Background concentrations are determined from measurements in soil samples taken at several nearby off-site locations where contamination is highly unlikely. The concentration of a radionuclide is considered to clearly exceed the background concentration if it is greater than the mean background concentration plus twice the standard deviation of the background measurements. If the concentrations in the samples used for determining the background concentration are below the lower limit of detection (LLD) of the instrument used, the concentration of that radionuclide is considered to exceed background if it exceeds the LLD of the instrument. The sensitivity of the instrument used must comply with current standards for high quality commercial instruments.

To justify the use of two or more contaminated zones, credible evidence must be provided on the basis of radiological survey data that the intervening area between any two contaminated zones is uncontaminated; otherwise, the contaminated zone should be characterized by a single compact area that contains the locations of all soil samples with above-background radionuclide concentrations.

30.2 RESRAD DATA INPUT REQUIREMENTS

The actual area of the contaminated zone should be entered into RESRAD. The area should be specified in square meters (m^2). A default value of 10,000 m^2 is used in the RESRAD code for the area of contaminated zone.

31 COVER DEPTH

31.1 DEFINITION

The cover depth is the distance, in meters (m), from the ground surface to the location of the uppermost soil sample with radionuclide concentrations that are clearly above background. The background concentration of a radionuclide is defined as the mean concentration in soil samples from nearby uncontaminated regions of the same soil type, plus twice the standard deviation of the counting statistics.

31.2 MEASUREMENT METHODOLOGY

Because the actual radionuclide distributions in a contaminated zone are nonuniform, the cover depth measured in each sampling borehole may not be the same. For a contaminated zone with an area greater than 100 m^2, the average cover depth over an integral subarea of 100 m^2 is calculated first. If one or more boreholes in the 100-m^2 subarea have a cover depth less than one-third of the average cover depth, then the average cover depth is replaced by one-third of the minimum value. The cover depth for the entire contaminated zone is then determined to be the same as the minimum average cover depth over the subareas. For a contaminated zone with an area less than 100 m^2, the average cover depth over the contaminated zone or one-third of the minimum cover depth in a borehole (if it is less than one-third of the average value) is taken as the representative value of the cover depth for the contaminated zone.

31.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the user is required to input a value for the cover depth. The default value used for cover depth is 0 (m).

32 DISTRIBUTION COEFFICIENTS

32.1 DEFINITION

The distribution coefficient, K_d, is the ratio between the mass of solute species adsorbed or precipitated on the solids per unit of dry mass of the soil, S, and the solute concentration in the liquids, C. The distribution coefficient represents the partition of the solute in soil matrix and soil water assuming chemical equilibrium is achieved. If a linear relationship between S and C exists, then

$$S = K_d C \tag{32.1}$$

The transfer of radionuclides from the liquid to the solid phase or vice versa may be controlled by mechanisms such as adsorption and precipitation, depending on the radionuclides. The dimensions of the distribution coefficient are unit of length cubed per mass (L^3/M).

The value of K_d varies over a wide range because of uncertainties in the laboratory methods used to determine its value, as well as its dependence on many soil properties. Soil properties affecting the distribution coefficient include the texture of soils (sand, loam, clay or organic soils) (Sheppard and Thibault 1990), the organic matter content of the soils, pH values (Coughtrey et al. 1985), the soil solution ratio (Sheppard et al. 1983), the solution or pore water concentration (Hoeffner 1985; Nikula 1982; Sheppard et al. 1987; Sheppard and Thibulat 1990), and the presence of competing cations (Bond and Smiles 1988; Gee et al. 1983; Nikula 1982; Hoeffner 1985; Uchida and Kamada, 1987; Rouston et al. 1984). Sometimes, the value of the distribution coefficient (K_d) for a specific radionuclide in soils ranges over several orders of magnitude under different conditions.

32.2 MEASUREMENT METHODOLOGY

32.2.1 Experimental Methods

The two most common experimental techniques for the determination of K_d are the batch and column methods. In the batch method, soil and water (or solvent) are shaken with the radionuclides for a period of time until equilibrium between the soil and the water is achieved or assumed. The distribution coefficient K_d is then taken as the ratio of the mass of the radionuclide in the solid phase of the soil to the radionuclide concentration in the water (or solvent).

The column method involves prepacking soil uniformly in a column then adding solvent with dissolved radionuclides and collecting the filtered solvent at the bottom of the column and analyzing its concentration. The distribution coefficient (K_d) of the radionuclide can be determined by comparing the breakthrough curve with the initial radionuclide concentration in the solution.

Whether the batch or column method is used to determine K_d, certain uncertainties are inherited in each method and should be kept in mind. First, because chemical equilibrium may not have achieved in the batch method, and dispersion occurs in the column method, the two techniques may give different results for nonionic elemental forms (Inoue and Morisawa 1976). Second, the distribution coefficient (K_d) is defined as a ratio of two concentrations; therefore, a small error in concentration measurements either in soil or in water may result in a large error in the coefficient. Another source of variation in K_d is the time factor involved in its determination. Normally, a bath run takes several hours. If equilibrium is not reached within this short period of time, error is introduced. Errors caused after a 24-hour experimental period however, are quite insignificant (Rhodes 1957; Relyea and Brown 1978; Wildung and Rhodes 1963).

32.2.2 Theoretical Analysis

Consider a situation in which water containing a dissolved tracer is introduced into a tracer-free soil column with a known dry density and volumetric water content. With the hydrodynamic dispersion (i.e., the mechanical dispersion and molecular diffusion) of radionuclides throughout the column and the adsorption of radionuclides to the soil, the initial sharp-tracer front near the top end of the soil column spreads out downward. A mass balance equation for the radionuclide concentration in the liquid phase can be derived as follows:

$$R\frac{\partial C}{\partial t} = D\frac{\partial C}{\partial x^2} - v\frac{\partial C}{\partial x} \tag{32.2}$$

where R is the retardation factor, D is the coefficient of hydrodynamic dispersion, v is the flow velocity of the water, and C is the radionuclide concentration in the water. The retardation factor R is related to the distribution coefficient K_d of the radionuclide as follows:

$$R = 1 + \frac{\rho_b K_d}{\theta} \tag{32.3}$$

where ρ_b is the dry soil density and θ is the volumetric water content of the soil. Therefore, K_d can be calculated if R is known. On the basis of the inlet and outlet boundary conditions:

$$C(0,t) = C_0 \tag{32.4}$$

and

$$\frac{\partial C}{\partial x}(\infty,t) = 0 \tag{32.5}$$

the solution to Eq. (32.2) is (Lapidus and Amundson 1952):

$$C(x,t) = \frac{C_0}{2}\left\{ erfc\left[\frac{Rx-vt}{2(DRt)^{1/2}}\right] + \exp\left(\frac{vx}{D}\right) erfc\left[\frac{Rx+vt}{2(DRt)^{1/2}}\right]\right\} \tag{32.6}$$

The relative effluent concentration, C_e, expressed in terms of two dimensionless parameters, the column Peclet number (P) and the number of pore volumes (T), is derived as follows:

$$C_e(T) = \frac{1}{2}\, erfc[(\frac{P}{4RT})^{1/2}(R-T)] + \frac{1}{2}\exp(P) erfc[(\frac{P}{4RT})^{1/2}(R+T)] \qquad (32.7)$$

where

$$C_e = C(L,t)/C_0 \qquad (32.8)$$

$$T = vt/L \qquad (32.9)$$

and

$$P = vL/D \qquad (32.10)$$

The average interstitial or pore-water velocity is represented by υ and is approximately equivalent to the ratio between the water flow rate and the volumetric water content, q/θ. The length of the soil column is represented by L. The parameter L, in the case of field measured concentration-time curves, simply refers to the soil depth at which the concentration was observed. The following expression is frequently used to describe displacement experiments (Danckwerts 1953; Rifai et al. 1956):

$$C(x,t) = \frac{C_0}{2}\, erfc[\frac{Rx-vt}{2(DRt)^{1/2}}] \qquad (32.11)$$

This equation provides a close approximation to Equation 32.6 for relatively large values of P. In terms of the Peclet number (P) and the number of pore volumes (T), when applied to the effluent concentration, Equation 32.11 can be written as follows:

$$C_e(T) = \frac{1}{2}\, erfc[(\frac{P}{4RT})^{1/2}(R-T)] \qquad (32.12)$$

Many empirical methods based on the measured relative effluent concentration (C_e) versus the number of pore volumes (T) have been used for the analysis of P and R. These include the trial-and-error, slope, log-normal plot, and least-squares methods (Van Genuchten and Wierenga 1986; Rifai et al. 1956). There are other methods such as the method of moments (Aris 1958; Agneessens et al. 1978; Skopp 1985; Valocchi 1985; Jury and Sposito 1985) and methods to determine the two coefficients (K_d and D) from the location and peak concentration of a short or instantaneous surface-applied tracer pulse (Kerkham and Powers 1972; Saxena et al. 1974; Yu et al. 1984). The reader is referred to the original studies for discussion of their applications.

32.2.3 Determination of the Distribution Coefficient Without Experiments

In addition to the experimental methods for determining the distribution coefficient (K_d), Sheppard and Thibault (1990) proposed an empirical approach to calculate K_d from the soil-to-plant concentration ratio (CR) on the basis of the strong negative correlation between CR and K_d (Sheppard and Sheppard 1989). According to their model, the relationship between K_d and CR can be described as follows:

$$\ln K_d = a + STEX + b\,(\ln CR) \tag{32.13}$$

where a, b, and STEX are constants. The values for the coefficients were a = 4.62 and b = -0.5. The value of STEX depends on soil type; for sandy soil, STEX = -2.51; for loamy soil, STEX = -1.26; for clayey soil, STEX = -0.84; and for organic soil, STEX = 0. The validity of Equation 32.13 is still under investigation, but a value of K_d can be derived when no experimental or literature data are available. Table 32.1 lists the geometric mean values of K_d obtained from the literature or predicted by using concentration ratios (Sheppard and Thibault 1990).

32.3 RESRAD DATA INPUT REQUIREMENTS

The default distribution coefficients used in the RESRAD code and their ranges based on data compiled by Sheppard and Thibault (1990) are listed in Table 32.2.

From Tables 32.1 and 32.2, it can be seen that K_d is quite variant; that is, it assumes different values under different circumstances. Because K_d is one of the important input parameters that have a strong influence on the calculated results in the RESRAD code, a site-specific value, if available, should always be used for risk assessment.

TABLE 32.1 Summary of Geometric Mean K_d Values (cm^3/g) for Each Element by Soil Type

Element	Sand	Loam	Clay	Organic
Actinium	450	1500	2400	5400
Silver	90^a	120^a	180^a	15000^a
Americium	1900^a	9600^a	8400^a	112000^a
Beryllium	250	800	1300	3000
Bismuth	100	450	600	1500
Bromine	15	50	75	180
Carbon	5^a	20	1	70
Calcium	5	30	50	90
Cadmium	80^a	40^a	560^a	900^a
Cerium	500^a	8100^a	20000^a	3300^a
Curium	4000^a	18000^a	6000	6000^a
Cobalt	60^a	1300^a	550^a	1000^a
Chromium	70^a	30^a	1500	270^a
Cesium	280^a	4600^a	1900^a	270^a
Iron	220^a	800^a	165^a	600^a
Hofnium	450	1500	2400	5400
Holmium	250	800	1300	3000
Iodine	1^a	5^a	1^a	25^a
Potassium	15	55	75	200
Manganese	50^a	750^a	180^a	150^a
Molybdenum	10^a	125	90^a	25^a
Niobium	160	550	900	2000
Nickel	400^a	300	650^a	1100^a
Neptunium	5^a	25^a	55^a	1200^a
Phosphorus	5	25	35	90
Protactinium	550	1800	2700	6600
Lead	270^a	16000^a	550	22000^a
Palladium	55	180	250	670
Polonium	150^a	400^a	3000	7300
Plutonium	550^a	1200^a	5100^a	1900^a
Radium	500^a	36000^a	9100^a	2400
Rubidium	55	180	270	670
Rhenium	10	40	60	150
Ruthenium	55^a	1000^a	800^a	6600^a
Antimony	45^a	150	250	550
Selenium	150	500	740	1800
Silicon	35	110	180	400
Samurium	245	800	1300	3000
Tin	130	450	670	1600
Strontium	15^a	20^a	110^a	150^a
Tantalum	220	900	1200	3300
Technetium	0.1^a	0.1^a	1^a	1^a
Tellurium	125	500	720	1900
Thorium	3200^a	3300	5800^a	89000^a
Uranium	35^a	15^a	1600^a	410^a

TABLE 32.1 (Cont'd)

Element	Sand	Loam	Clay	Organic
Yttrium	170	720	1000	2600
Zinc	200[a]	1300[a]	2400[a]	1600[a]
Zirconium	600	2200	3300	7300

[a] Values obtained from the literature; all other values are predicted by using concentration ratios.

Source: Values obtained from Sheppard and Thibault 1990.

TABLE 3.2.2 RESRAD Default K_d Values and Ranges of Geometric Mean Values[a]

Element	RESRAD K_d	K_d Range[a]
Hydrogen	0	not available
Carbon	0	1-70
Sodium	22	not available
Chlorine	0.1	not available
Potassium	not included	15-200
Calcium	50	5-90
Manganese	200	50-750
Iron	1000	165-800
Cobalt	1000	60-1300
Nickel	1000	300-1100
Strontium	30	15-150
Niobium	0	160-2000
Technetium	0	0.1-1
Ruthenium	0	55-66000
Antimony	0	45-550
Iodine	0.1	1-25
Cesium	1000	170-4600
Cerium	1000	500-20000
Samarium	0	245-3000
Europium	0	not available
Lead	100	270-22000
Radium	70	500-36000
Actinium	20	450-5400
Thorium	60000	3200-89000
Protactinium	50	550-6600
Uranium	50	15-1600
Neptunium	0	5-1200
Plutonium	2000	550-5100
Americium	20	1900-112000
Curium	0	4000-18000
Californium	228	not available

[a] Source: Sheppard and Thibault 1990.

33 RADIONUCLIDE CONCENTRATION IN GROUNDWATER

33.1 DEFINITION

This parameter is a measure of the concentration of the principal radionuclide in a well located at the downgradient edge of the contaminated zone. The groundwater concentration and the radionuclide concentration in soil should be measured simultaneously because they are used in RESRAD as a pair to estimate distribution coefficient.

33.2 RESRAD DATA INPUT REQUIREMENTS

This parameter should be entered in units of picocuries per liter (pCi/L). Input values for the radionuclide concentration in groundwater are required only if the value of the elapsed time since placement of waste material parameter is greater than zero. Only principal radionuclides with nonzero concentrations in soils will have nonzero concentrations in groundwater. These nonzero groundwater concentration inputs will invoke the calculation of soil/water distribution coefficients, and the input coefficient values will be supressed by the calculated results.

34 LEACH RATE

34.1 DEFINITION

The leach rate constant is the fraction of the available radionuclide leached out from the contaminated zone per unit of time. Source factors in the RESRAD code are calculated on the basis of the ingrowth and decay processes among radionuclides and the leach rate constant for adjusting radionuclide concentrations in the contaminated zone.

34.2 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the leach rate should be entered in units of yr^{-1}. An input value of 0 for the leach rate constant will invoke the calculation of this parameter via a first-order leaching model with the value of the soil/water distribution coefficient in the contaminated zone. If the input value of this parameter is greater than 0, however, it will also be used to derive the soil/water distribution coefficient of the contaminated zone conversely, again based on the first order leaching model. The input soil/water distribution coefficients are then replaced by the derived value.

Because the leach rate constant and the soil/water distribution coefficients are the two most critical parameters affecting the calculated results of water-related pathways, site-specific values should always be used whenever available. The default leaching rate constant in RESRAD is 0.

35 MASS LOADING FOR INHALATION

35.1 DEFINITION

The mass loading parameter is the concentration of soil particles in the air and is measured in grams per cubic meter (g/m^3). Use of a mass loading factor from empirical data eliminates consideration of details of the resuspension mechanisms; in particular, the effective depth of the disturbed layer can be bypassed.

Average, ambient concentrations of transportable particles range from 3.3×10^{-5} g/m^3 to 2.54×10^{-4} g/m^3 in urban locations and from 9×10^{-6} g/m^3 to 7.9×10^{-5} g/m^3 in nonurban locations (Gilbert et al. 1983). Anspaugh et al. (1974) and Healy and Rodgers (1979) used 1×10^{-4} g/m^3 for predictive purposes and found that this value gave reasonable results. The U.S. Environmental Protection Agency (EPA) (1977) has used the same value to screen calculations.

The mass loading value will fluctuate above its ambient level depending on human activities such as plowing and cultivating dry soil or driving on an unpaved road. The estimated mass loading for construction activities is about 6.0×10^{-4} g/m^3, for exposure to construction traffic on unpaved roads it is 4.0×10^{-4} g/m^3, and for agriculture-generated dust, it is about 3.0×10^{-4} g/m^3 (Oztunali et al. 1981). The maximum respirable dust loading inside the cab of heavy construction equipment during a surface coal mining operation was found to be 1.8×10^{-3} g/m^3 (Oztunali et al. 1981). Estimates of mass loadings have been as high as 1.3 g/m^3 for instantaneous mass loadings during tilling.

35.2 RESRAD DATA INPUT REQUIREMENTS

Gilbert et al. (1989) suggest a mass loading factor of 2.0×10^{-4} g/m^3 for transportable particles at an onsite loading to take into account short periods of high mass loading and sustained periods of normal farmyard activities for which the dust level may be somewhat higher than ambient. This value is the default value used in RESRAD.

36 OCCUPANCY FACTOR FOR INHALATION PATHWAY

36.1 DEFINITION

This factor should be adjusted on the basis of scenario assumptions. The correct input values can be obtained by using the following equation:

$$FO = TF_1 \times 1 + TF_2 \times 0.4 + TF_3 \times 0 \qquad (36.1)$$

where TF_1 is the fraction of time spent outdoors on-site, TF_2 is the fraction of time spent indoors on-site, and TF_3 is the fraction of time spent off-site.

36.2 RESRAD DATA INPUT REQUIREMENTS

The default occupancy factor (FO) used in RESRAD is 0.45, which is based on the assumption that 50% of a person's time is spent indoors (where the dust level is 40% of the outdoor level [Alzona et al. 1979]), 25% is spent outdoors in the contaminated area, and 25% is spent in uncontaminated areas. A different factor might be used on the basis of site-specific activity profiles for the population group being assessed .

37 DEPTH OF ROOTS

37.1 DEFINITION

This parameter is the average root depth of various plants such as fruit trees and vegetables grown in the contaminated zone.

37.2 RESRAD DATA INPUT REQUIREMENTS

The average root depth should be entered in units of meters (m). The default value in the RESRAD code is 0.9 m.

38 SOIL INGESTION RATE

38.1 DEFINITION

This parameter is the accidental ingestion rate of soil material or soil dust. Children tend to mouth or ingest substances that are not considered to be food, especially those under 18 months. When this behavior extends beyond the age of 18 months, the child is said to practice pica (Barltrop 1966, Robischon 1971, Ziai 1983). Many factors such as nutrition, quality of care, and parental relationship (Behrman and Vaughan 1983; Bellinger et al. 1986; Bicknell 1967; Danford 1982; Danford et al. 1982; Forfar and Arneil 1984; Glickman et al. 1981) influence the extent of this behavior. However, it is believed that a child who practices pica is no different from one who does not because pica cannot be consistently predicted (Feldman 1986), even though severe pica usually occurs among grossly disturbed or mentally retarded children.

According to the literature, a wide variety of substances are ingested: soil, clay, sand, dust, grass, leaves, plaster, hair, starch, paint chips, string, soap, wood, powders, chalk, and paper. No quantitative ingestion rates have been suggested because children with known pica behavior have not been studied. "Abnormal" soil ingestion (pica) is believed to be uncommon and may need to be addressed separately.

On the basis of observational data, children are most likely to ingest soil from age 1 to 6 (Walter et al. 1980, Cooper 1957, Charney et al. 1980, Sayre et al. 1974). Beyond age 6 or 7, ingestion of nonfood substances is usually caused by inadvertent ingestion or developmental problems. Paustenbach et al. (1986) summarized the normal amount of soil ingested by children on the basis of the age of the child. Vermeer and Frate (1979) pointed out that the environmental setting is also an important factor for children in rural areas who tend to ingest a higher amount of soil. Hawley (1985) used data from the literature to develop scenarios to estimate ingestion amounts for young children, older children and adults.

He divided each year into two activity periods: May through October when individuals spend more time outdoors and November through April when most of the time, weather conditions eliminate outdoor activities. Hawley's study indicated that the amount ingested by young children (2.5 years old, weighing 13.2 kg), during outdoor activity between May and October (5 d/week) is 250 mg/d. During November through April, the ingestion during indoor activity is 100 mg/d. For children six-years old, weighing approximately 20.8 kg, the ingestion amount is 50 mg/d during outdoor activity from May through October and 3 mg/d year-round for indoor activity. Working in attics or other uncleaned areas of a house can cause adults (weighing 70 kg) to ingest 110 mg/d of soil for an assumed duration of 12 d/yr. For living space activities, the ingestion amount is 0.56 mg/d. For outdoor activities from May through October, the ingestion amount is 480 mg/active day assuming an 8 h/d and 2 d/week.

According to Binder et al. (1986), the average quantity of soil ingested by children is about 108 mg/d (within a range 4 to 708 mg/d). Clausing et al. (1987) estimated that the ingestion rate of children is 105 mg/d, with a range of 23 to 362 mg/d. Binder et al. (1986) and Clausing et al. (1987) have also provided some limited information on the upper limit of the soil ingestion rate on the basis of evidence that the upper range of the ingestion rate for children is around 800 mg/d or more.

An amount has not been estimated for abnormal soil ingestion behavior among children. However, some evidence suggests that a rate of 5 to 10 g/d may not be unreasonable. The EPA used 5 g/d in its risk assessment for TCDD (EPA 1984). The USDA used a value of 10 g/d in conducting exposure assessments related to the use of sludge in gardens and soils.

After reviewing the limited data available, the EPA (1990) decided that the studies of Binder et al. (1986) and Clausing et al. (1987) appear to be the most reliable and suggested that an estimate of 0.2 g/d be used as an average value for young children (under the age of 7). An upper range of soil ingestion is 0.8 g/d. For other age groups (children older than 7 years), 0.1 g/d should be used for the soil ingestion amount. These factors account for ingestion of both outdoor soil and indoor dust.

38.2 MEASUREMENT METHODOLOGY

Several methods have been used to characterize soil ingestion by children. Lepow et al. (1975) measured hand dust by applying preweighed adhesive labels to the hands and weighing the amount of dirt that was removed. They also observed "mouthing" behavior and reported that a child would put his or her fingers into the mouth about 10 times a day. Duggan and Williams (1977) and Day et al. (1975) also measured the amount of dust on children's hands. Binder et al. (1986) studied the ingestion of soil among children 1 to 3 years of age who wear diapers. Both excreta and soil from the play yards were analyzed for materials that were thought to be poorly absorbed in the gut. Clausing et al. (1987) conducted a soil ingestion study by using a tracer element method similar to that of Binder et al. (1986). They also collected fecal samples for six hospitalized, bedridden children to represent a control group.

Presently there is no widely accepted method for determining the relative contribution of each medium (i.e. soil vs. dust) to the daily ingestion amounts and the effect of climatic variations (e.g. snow cover).

38.3 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the soil ingestion rate should be entered in units of grams per year (g/yr). The default value of 36.5 g/yr is used, which accounts for an average soil intake rate of 0.1 g/d, as recommended by EPA, for 365 d/yr.

39 THICKNESS OF CONTAMINATED ZONE

39.1 DEFINITION

This parameter is the distance between the uppermost and lowermost soil samples that have radionuclide concentrations clearly above background. A soil sample is clearly contaminated if the radionuclide concentration is greater than the average background radionuclide concentration plus twice the standard deviation of the background measurements.

39.2 MEASUREMENT METHODOLOGY

To determine the thickness of a contaminated zone with an area greater than 100 m^2, the average contamination thickness of boreholes drilled to take soil samples is calculated over any integral subarea of 100 m^2. If one or more boreholes in the subarea have a contamination thickness exceeding the average thickness by a factor larger than three, then the average value is replaced by one-third of the maximum contamination thickness. The thickness of the contaminated zone is then taken as the maximum average thickness calculated over a 100 m^2 subarea. For a contaminated zone with an area less than 100 m^2, the average contamination thickness over the contaminated zone or one-third of the maximum contamination thickness in a borehole (if the thickness is greater than three times the average value) is taken as the representative value of the contaminated zone thickness.

39.3 RESRAD DATA INPUT REQUIREMENTS

In RESRAD, the thickness of this ideal contaminated zone is entered in units of meters (m). The default value is 1 m.

40 BASIC RADIATION DOSE LIMIT

The radiation dose used in the RESRAD code is the effective dose equivalent from external radiation plus the committed effective dose equivalent from internal radiation (International Commission on Radiological Protection [ICRP] 1984). The radiation dose limit based on radiation protection standards and requirements as specified in U.S. Department of Energy (DOE) Order 5400.5 (DOE 1990) is 100 mrem/yr. This is also the default value adopted in the RESRAD code.

41 SEAFOOD CONSUMPTION RATE

41.1 DEFINITION

National recreational catch data for coastal areas were obtained by the National Marine Fisheries Service (NMFS) in 1985. The NMFS conducted a direct survey of fishermen in the field and an independent telephone survey of households (NMFS 1986). Total fish consumption data were obtained from a one-year survey conducted by NPD Research, Inc., during 1973 and 1974, and funded by the Tuna Research Institute. Questionnaires were answered by 6,980 families representing the U.S. population.

Javitz (1980) used the data obtained by NPD Research, Inc. to calculate the mean and 95th percentile of seafood consumption for seafood consumers in the United States as 14.3 g/d (5.2 kg/yr) and 41.7 g/d (15.2 kg/yr), respectively. The mean average of 14.3 g/d for seafood consumption includes 2.1 g/d (0.8 kg/yr) for nonfish seafood consumption, that is, lobsters, oysters, scallops, shrimps, squids, and so forth. Unfortunately, the original NPD survey data for seafood consumption did not distinguish between recreationally caught and purchased fish; therefore, this difference is not reflected in the calculated mean and 95th percentile values.

Puffer (1981) conducted 1,059 interviews with sport fishermen in the Los Angeles Harbor area. The interviews revealed that sport fishermen keep 67% to 89% of the finfish and 97% of the shellfish that they catch. The median and 90th percentile seafood (fish plus shellfish) consumption rates of sport fishermen are 37 g/d and 225 g/d, respectively.

Another source for the seafood consumption rate of sport fishermen is a survey conducted in Commencement Bay at Tacoma, Washington, by Pierce et al. in 1981. The sample size (304 fishermen) was smaller than that of Puffer (1981), and the sampling frequency was lower. It was found that over half of the fishermen caught and consumed fish weekly. Pierce et al. (1981) concluded that the mean average seafood consumption rate for the surveyed fishermen was 23 g/d (within a range of 12 to 54 g/d), and the 90th percentile was 54 g/d.

Although the surveys conducted by Puffer (1981) and Pierce et al. (1981) are limited to the West Coast, the EPA (1990) considers these studies to be representative of actual annual consumption rates for recreational fishermen. By averaging the results of these two surveys, the EPA (1990) has suggested that the 50th and 90th percentile seafood consumption of fishermen are 30 g/d (11 kg/yr) and 140 g/d (51 kg/yr), respectively.

Because sport fishermen and their families consume much more seafood than other people, the EPA recommends that the consumption rates of fishermen based on Puffer (1981) and Pierce et al.'s (1981) surveys be used as comparative references for any area where there is a large body of water and widespread contamination is possible.

The NRC (1977) used values of 2.2 kg/yr, 5.2 kg/yr, and 6.9 kg/yr for average individual fish consumption for children, teenagers and adults, respectively. Average individual consumption rates of other seafood were 0.33 kg/yr, 0.75 kg/yr, and 1.0 kg/yr for the three different groups. For a worst-case scenario, the fish consumption rates were 6.9 kg/yr, 16 kg/yr, and 21 kg/yr for children, teenagers, and adults, respectively. For other seafood consumption rates, values of 1.7 kg/yr, 3.8 kg/yr, and 5 kg/yr were used

41.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the seafood consumption rate should be entered in units of kg/yr. The default value for the consumption rate is 5.4 kg/yr for fish and 0.9 kg/yr for other seafood.

42 FRUIT, VEGETABLE, AND GRAIN CONSUMPTION RATE

42.1 DEFINITION

According to the latest survey by the National Gardening Association (1987), 38% or a total of 34 million U. S. households participated in vegetable gardening in 1986. The size of the home vegetable garden, however, has decreased from 600 square feet in 1982 to 325 square feet in 1986 (National Gardening Association 1987). The distribution of home gardens varies geographically, with a large percentage located in the Midwest and South, and more in rural areas than in cities and suburbs. Therefore, homegrown fruits and vegetables make up a larger portion of the average consumption rate in rural areas than in cities or suburbs.

The EPA has made recommendations on the consumption rates of homegrown fruits and vegetables on the basis of two sources of information: *Foods Commonly Eaten by Individuals: Amount Per Day and Per Eating Occasion* (Pao et al. 1982) and *Food Consumption: Households in the United States, Seasons and Year 1977-1978* (USDA 1983). The first source used data collected by the USDA in 1977-1978 from home interviews of 37,874 respondents who were asked to recall food consumed one day before the interview, the day of the interview, and the day after the interview, to calculate percentiles of total fruit and vegetable consumption of the U.S. population. The consumption rate of homegrown fruits and vegetables can be calculated by subtracting the data of the "bought" category for all foods from the data of the "all" category in the USDA food consumption survey. Homegrown dark green vegetables make up approximately one-third of the dark green vegetables consumed. This category includes mustard greens, kale, kohlrabi, spinach, and broccoli. Consumption of homegrown corn, cucumbers, green beans, and tomatoes makes a significant contribution to total consumption. The proportion of homegrown fruits consumed is highest for strawberries, peaches, and pears, and lowest for citrus fruits.

According to the EPA (1990), the average consumption rate of vegetables per person is 200 g/d (73 kg/yr); homegrown products account for 25% of the total consumption rate, which is 50 g/d (18 kg/yr). Total average daily fruit intake is 140 g/d (51 kg/yr) per individual. The total homegrown fruit consumption rate is 28 g/d (10 kg/yr), which is 20% of the total intake rate. For a reasonble worst case, it is suggested that 40% of the total intake is homegrown vegetable consumption and 30% of the total intake is homegrown fruit consumption. Table 42.1 summarizes the EPA's recommendations.

The EPA data provided above do not include information about the grain product consumption. In NRC Regulatory Guide 1.109 (NRC 1977), different total consumption amounts of fruits, vegetables, and grains are suggested for different age groups. The average individual consumption for a child is 200 kg/yr, for a teenager it is 240 kg/yr, and for an adult it is 190 kg/yr. Suggested values used for the maximally exposed individual in a worst case scenario are 520 kg/yr, 630 kg/yr, and 520 kg/yr, for a child, teenager, and adult, respectively. The total consumption for the maximum exposure case consists of 22% for fruit consumption, 54% for vegetable consumption, and 24% for grain consumption.

42.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the consumption rate should be entered in units of kg/yr. The default value for the consumption rate is 160 kg/yr for fruit, vegetables, and grain.

TABLE 42.1 **Vegetable and Fruit Intake Rates**

	Average Total Intake		Average Intake Homegrown		Worst-Case Intake Homegrown	
Item	(g/d)	(kg/yr)	(g/d)	(kg/yr)	(g/d)	(kg/yr)
Vegetables	200	73	50	18	80	29
Fruits	140	51	28	10	42	15
Total	340	124	78	28	122	44

43 INHALATION RATE

43.1 DEFINITION

The inhalation rate varies with activity level, age, weight, sex, and general physical condition. Several formulas have been proposed to calculate the inhalation rate for a human at rest from anthropometric data (EPA 1985). However, in general, the formulas are based on measurements from relatively small sample sizes and are limited to calculating the inhalation rate at rest only.

The EPA (1985) has compiled the available data, most of which is from early studies, and has derived an inhalation rate expressed in cubic meters per hour (m^3/h). Inhalation rates were compiled for each age/sex group at rest and at light, moderate, and heavy activity levels. The activity levels were categorized according to criteria developed by the Environmental Criteria and Assessment Office of the EPA for the air quality criteria document for ozone. A male adult with a body weight of 70 kg (EPA 1984) was used as a reference, and activity level categories for the other age/sex groups were extrapolated from the criteria for male adults on the basis of body weight (American Industrial Hygiene Association 1971). Table 43.1 gives a summary of human inhalation rates at different age/sex/activity levels (EPA 1985). Resting is characterized by such activities as watching television, reading, or sleeping. Light activity includes level walking, meal cleanup, care of laundry and clothes, domestic work and other miscellaneous household chores, attending to personal care of needs, photography, hobbies, and conducting minor indoor repairs and home improvements. Moderate activity includes climbing stairs, heavy indoor cleanup, and performing major indoor repairs and alterations (e.g., remodeling). Heavy activity consists of vigorous physical exercise such as weight lifting, dancing, or riding an exercise bike.

TABLE 43.1 Summary of Human Inhalation Rates for Men, Women, and Children by Activity Level (m^{3}yr)

	Resting	Light	Moderate	Heavy
Adult male	0.7	0.8	2.5	4.8
Adult female	0.3	0.5	1.6	2.9
Average adult	0.5	0.6	2.1	3.9
Child, age 6	0.4	0.8	2.0	2.4
Child, age 10	0.4	1.0	3.2	4.2

Assuming 16 hours of light activity and 8 hours of resting, the ICRP (1981) has reported a 23- m^3/d inhalation rate for adult males and a 21 m^3/d rate for adult females, giving an average value of 22 m^3/d (8,030 m^3/yr) for adults.

Data presented by the EPA (1985) suggest lower inhalation rates for light and resting activity levels. Using the same assumption as the ICRP (1981), the daily inhalation rate would be about 14 m^3/d (5,110 m^3/yr). In adition to assuming lower rates for light resting activity levels, the EPA estimated the daily inhalation rate for moderate and heavy activity levels; therefore, it is possible to estimate the total inhalation rate for any combination of activity levels. The EPA's data suggest that the maximum inhalation rate is roughly twice the reported mean rates for all activity levels.

The EPA (1990) made the following recommendations on the basis of the above mentioned data: 20 m^3/d (7,300 m^3/yr) should be used as the average adult daily inhalation rate and 30 m^3/d (11,000 m^3/yr) as the reasonble worst-case inhalation rate, provided the activity patterns are unknown. For exposure scenarios in which the distribution of activity patterns is known, then the values in Table 47.1 should be used, because it provides a more representative rate for calculation.

For an individual performing outdoor activities, a typical activity mix can consist of 37% at a moderate activity level, 28% at both resting and light activity levels, and 7% at a heavy activity level, which results in a 1.4 m^3/h (12,300 m^3/yr) inhalation rate. A reasonable worst-case outdoor inhalation rate can account for 50% of the time at a heavy activity level and 50% at a moderate activity level with an inhalation rate of 3.0 m^3/h (26,300 m^3/yr).

For an individual performing indoor activities, an average assumption would include 48% of the time both at a resting and light activity level, 3% at a moderate activity level, and 1% at a heavy activity level. A reasonable worst-case includes 25% at a resting activity level, 60% at a light activity level, 10% at a moderate activity level, and 5% at a heavy activity level. The first assumption has an average inhalation rate of 0.63 m^3/h (5,500 m^3/yr), and the second one has a reasonble worst-case inhalation rate of 0.89 m^3/h (7,800 m^3/yr).

4.3.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the inhalation rate should be entered in units of m^3/yr. The default value used in the code is 8,400 m^3/yr.

44 LEAFY VEGETABLE CONSUMPTION RATE

44.1 DEFINITION

The leafy vegetable consumption rate is a dietary factor for human food consumption, which includes consumption of vegetables such as spinach and lettuce.

44.2 RESRAD DATA INPUT REQUIREMENTS

The national average value for leafy vegetable consumption is 14 kg/yr, which is also the default value used in RESRAD.

45 LIVESTOCK WATER INTAKE RATE FOR BEEF CATTLE AND MILK COWS

45.1 DEFINITION

According to NRC Regulatory Guide 1.109 (NRC 1977), the water ingestion rate for beef cattle is 50 l/d. The water ingestion rate for milk cows is 14 gal/d (approximately 50 l/d) plus one gallon for every three pounds of milk produced (Great Lakes Basin Commission 1975). If a production rate of 10 gal/d of milk is assumed, then the water ingestion rate for milk cows would be about 160 l/d (Gilbert et al. 1983).

45.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the livestock water intake should be entered in units of l/d. The default values for beef cattle and milk cows are therefore set to 50 l/d and 160 l/d, respectively, if the user does not specify otherwise.

46 MEAT AND POULTRY CONSUMPTION RATE

46.1 DEFINITION

The USDA conducted a national food consumption survey in 1977-1978 (USDA 1983). The average consumption rates for beef and dairy products as adopted by the EPA (1984a,1984b) are based on the results of this survey.

According to USDA studies, 44% of the annual consumption of beef is homegrown. This finding is based on a survey of 900 rural farm households (USDA 1966). Because the total amount of beef consumed averages approximately 100 g/d (36.5 kg/yr), the average consumption of homegrown beef is about 44 g/d (EPA 1990), which corresponds to 16 kg/yr.

As to the reasonable worst-case the EPA (1990) has suggested that a consumption rate of homegrown beef of 75 g/d (27 kg/yr) be used in risk assessments until better data are available.

The average consumption rate of 36.5 kg/yr recommended by the EPA accounts for beef only. The total consumption rate for meat and poultry should be much higher. According to NRC Regulatory Guide 1.109 (NRC 1977), the recommended average value for consumption of meat and poultry is 37 kg/yr for a chlid, 59 kg/yr for teenagers, and 95 kg/yr for adults. Recommended values for use in a maximum exposed case are 41 kg/yr for children, 65 kg/yr for teenagers, and 110 kg/yr for adults.

Gilbert et al. (1983) used a value of 79 lb/yr (36 kg/yr) for meat, 20 lb/yr (9 kg/yr) for poultry, and 15 lb/yr (7 kg/yr) for egg consumption, with a total value of 114 lb/yr (52 kg/yr). The consumption rate of meat used is about the same as that recommended by the EPA (1990). If the same percentage of homegrown beef can be applied to the consumption of poultry and eggs, then the average consumption of homegrown meat, poultry, and eggs would be 23 kg/yr; for a reasonable worst-case scenario, the value would be 39 kg/yr, on the basis of data of Gilbert et al. (1983).

46.2 RESRAD INPUT DATA REQUIREMENTS

In the RESRAD code, the consumption rate for meat and poultry should be entered in units of kg/yr. The default value for consumption rate is 63 kg/yr.

47 OCCUPANCY AND SHIELDING FACTOR (EXTERNAL GAMMA)

47.1 DEFINITION

This factor should be adjusted on the basis of scenario assumptions. The correct input values can be obtained by using the following equation:

$$FO = TF_1 \times 1 + TF_2 \times 0.7 + TF_3 \times 0 \qquad (47.1)$$

where TF_1 is the fraction of time spent outdoors, TF_2 is the fraction of time spent indoors on-site, and TF_3 is the fraction of time spent off-site.

47.2 RESRAD DATA INPUT REQUIREMENTS

A combined occupancy and shielding factor is used in RESRAD for calculating the effective dose in the external gamma radiation pathway. It is assumed that the indoor radiation exposure level is 70% of the outdoor exposure level. Therefore, a scenario in which a person spends 50% of the time indoors on-site, 25% outdoors on-site, and 25% in uncontaminated areas results in an occupancy and shielding factor of 0.6 ($0.25 \cdot 1 + 0.5 \cdot 0.7 + 0.25 \cdot 0$). This is the default value in the RESRAD code. A different factor might be used on the basis of the assumed activity profiles for the population group being assessed.

48 ELAPSED TIME OF WASTE PLACEMENT

48.1 DEFINITION

The elapsed time of waste emplacement parameter is the duration between the dumping of radioactive waste on-site and the performance of a radiological survey. It is possible that on-site radioactive wastes originated from different sources and have different placement times. Under this situation, an average value or a best representative value should be used.

When using RESRAD for risk assessment, the information obtained during radiological survey is input to derive soil guidelines for cleanup criteria. This information includes soil/water distribution coefficients, soil radionuclide concentrations, and so forth. The soil/water distribution coefficients is used for calculating the breakthrough and rise times of the groundwater contamination and for predicting the future radionuclide concentration in groundwater. In this case, the elapsed time of waste placement is zero. Nonzero values of this parameter should be input only when the soil/water distribution coefficients are not available and above background level groundwater radionuclide concentrations were measured in a radiological survey. Under such conditions, the input radionuclide concentration in groundwater, together with the elapsed time of waste placement would be used to derive soil/water distribution coefficients and to predict the future radionuclide concentration in groundwater.

48.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the elapsed time of waste placement should be entered in units of years. The default value of this parameter is set at zero.

49 SHAPE FACTOR (EXTERNAL GAMMA)

49.1 DEFINITION

A shape factor is used to correct for the noncircular-shaped contaminated area on the basis of an ideally circular zone. The shape factor for a circular contaminated area is 1.0. For an irregularly shaped contaminated area, the shape factor may be obtained by enclosing the irregularly shaped contaminated area in a circle, multiplying the area factor of each annulus by the fraction of the annulus area that is contaminated, summing the products, and dividing by the area factor of a circular contaminated zone that is equivalent in area. The area factors of circular contaminated zones with different radii are listed in Table 53.1. The area factor of an annulus is the area factor from an annular zone bounded by the radii tabulated in Table 49.1.

If an irregularly shaped contaminated zone of 191.4 m^2 is shaped like that in Figure 49.1, the shape factor can be calculated by surrounding the area with the appropriate annuli as indicated. The contaminated fractions within each annulus are 1, 1, 0.97, and 0.22, respectively. The area factor of the irregularly shaped contaminated zone can be calculated as follows on the basis of the values in Table 53.1:

$$\Sigma_{i=1,4} \text{ (area factor)}_i \times \text{(contaminated fraction)}_i \tag{49.1}$$

$$= (0.016 \times 1) + (0.4-0.016) \times 1 + (0.55-0.4) \times 0.97 + (0.8-0.55) \times 0.22 \tag{49.2}$$

$$= 0.601 \tag{49.3}$$

Next, the area factor of a circular contaminated equivalent in area must be determined. The radius of a circle with an area of 191.4 m^2 is 7.8 m. By interpolating the data of Table 49.1, the area factor of a circle equivalent in area can be determined:

$$(0.8-0.55) / (13-5.6) * (7.8-5.6) + 0.55 = 0.62 \tag{49.4}$$

TABLE 49.1 Area Factors for External Gamma Radiation from Contaminated Ground

Contaminated Area (m^2)	Radius[a] (m)	Area Factor,[b] FA
1	0.56	0.016
25	2.8	0.4
100	5.6	0.55
500	13	0.8
1,200	20	1.0

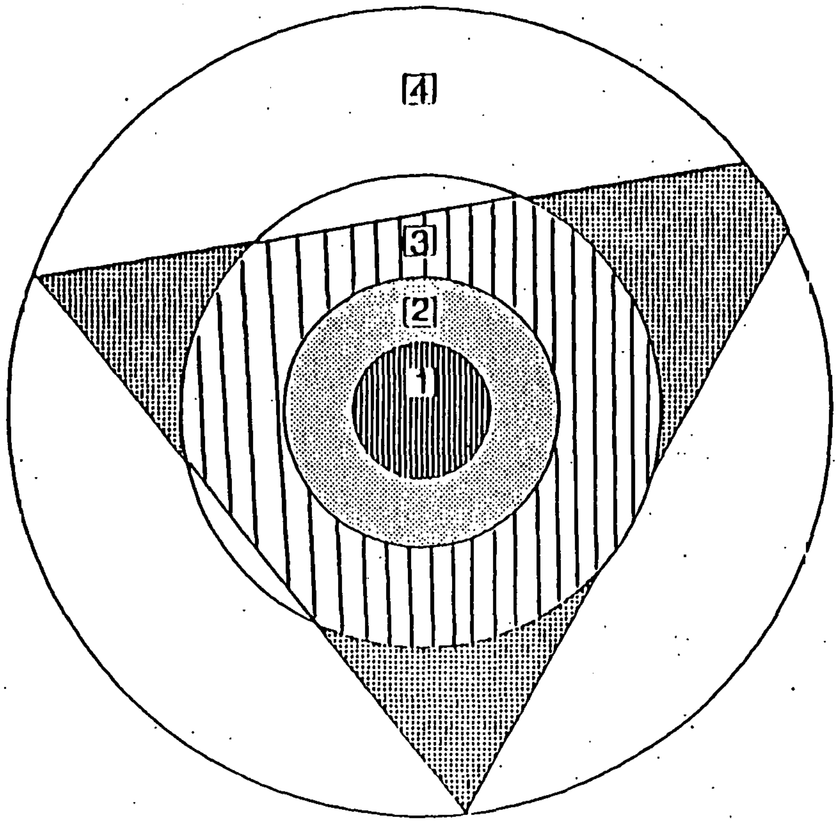
[a] Radius for a circular contaminated area.

[b] Intermediate values may be obtained by linear interpolation.

Source: Napier et al. 1984.

FIGURE 49.1 An Irregularly Shaped Contaminated Zone Enclosed by Four Annuli

annulus	(m)	area factor	fraction
1	0-0.56	0.016	1
2	0.56-2.8	0.4-0.016	1
3	2.8-5.6	0.55-0.4	0.97
4	5.6-13	0.8-0.55	0.22



▥ = contaminated area of annulus 1

▒ = contaminated area of annulus 2

▥ = contaminated area of annulus 3

▒ = contaminated area of annulus 4

U84583

Figure 1. An irregularly shaped contaminated zone enclosed by four annuli.

The final step in obtaining the shape factor of the contaminated zone is to divide the area factor of the contaminated zone by that of the circular zone with an equivalent area:

$$0.601 / 0.62 = 0.97 \tag{49.5}$$

Therefore, the shape factor of an irregularly shaped contaminated zone such as that in Figure 53.1 is determined to be 0.97.

49.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, an ideally circular contaminated zone is the default shape. Therefore, the default value of the shape factor is 1.

If the input shape factor is negative, that is, between 0 and -1, then the SOILD model (Chen 1991) instead of the model mentioned in the original RESRAD report (Gilbert et al., 1989) will be used for calculating the effective dose from external radiation. In this case, the shape factor will not actually be used in SOILD calculations, it serves as an indicator for the choice of options. More detailed information about the fractions of annular areas within the contaminated zone should be provided for use in the SOID calculations.

50 INITIAL CONCENTRATIONS OF PRINCIPAL RADIONUCLIDE

50.1 DEFINITION

A principal radionuclide is a radionuclide with a half-life longer than one-half year. Radionuclides with a half-life of one-half year or less are treated as associated radionuclides. The radionuclides "associated" with a principal radionuclide consist of all decay products down to, but not including, the next principal radionuclide in the chain. It is assumed that all associated radionuclides (except radon daughters) are in secular equilibrium with their principal radionuclide in the contaminated zone and also at the location of human exposure. Only the principal radionuclides in the contaminated zone need input values of radionuclide concentrations.

The single-radionuclide soil guidelines do not depend on the radionuclide concentrations in soil. Even if the radionuclide concentrations are not known, values for these guidelines can be obtained by entering any nonzero radionuclide concentration. The mixture sums (or total dose), however, depend on the radionuclide concentrations; thus, calculated mixture sum values are valid only if the soil radionuclide concentrations are known. When the radionuclide concentrations in soil and groundwater are used with the elapsed time of waste placement to derive the soil/water distribution coefficient, the values of the initial concentrations of the principal radionuclide must be known to obtain accurate results.

For a site-specific case, the distributions of radionuclides are nonuniform. The potential annual individual dose received through a particular pathway is an average of the nonuniform residual radioactivity over an area determined by the scenario activities; for example, the area of daily activities for external radiation or the size of the garden for the plant food pathway. For the purpose of deriving soil guidelines, it is assumed that this area is 100 m^2 for all pathways. The effect of vertical nonuniformities is taken into account by averaging the radionuclide concentrations in a 0.15-m-thick layer over the 100-m^2 area.

The initial concentration of a principal radionuclide is determined by the following procedures. For a contaminated zone with an area greater than 100 m^2, the average radionuclide concentration for any subzone of a 100-m^2 area and 0.15 m thickness is determined. If one or more soil samples within this subzone have radionuclide concentrations greater than three times the average radionuclide concentration, then the average radionuclide concentration of this subzone is replaced by one-third of the maximum measured soil radionuclide concentration. The initial concentration of a principal radionuclide in the contaminated zone is the maximum value of the average subzone radionuclide concentration. For a contaminated zone with an area less than 100 m^2, the initial concentration of a principal radionuclide is the maximum average radionuclide concentration of the 0.15-m-thick subzones.

50.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the initial concentrations of principal radionuclides in the contaminated zone are expressed in units of picocuries per gram (pCi/g). RESRAD treats the contaminated zone as a uniformly contaminated area with a single principal radionuclide at every point.

51 DRINKING WATER INTAKE RATE

51.1 DEFINITION

The EPA uses 2 l/d as the average amount of water consumed by an adult; this includes juices and beverages containing tap water (e.g., coffee). However, this value was established by the U.S. Army in determining the amount of water needed per person in the field and is believed to be an overestimate.

The National Academy of Sciences (NAS) (1977) calculated the average consumption rate of water to be 1.63 l/d per person. It is reasonable to assume that people in physically oriented occupations or living in warmer regions may have an intake rate exceeding this level. Although the consumption rate of 1.63 l/d seems to have a more scientific basis than 2 l/d, the NAS (1977) still adopted the larger volume (i.e. 2 l/d) to represent the intake rate for the majority of people.

Several other drinking water intake rates have been suggested. The National Cancer Institute (NCI) in an investigation of the possible relationship between bladder cancer and drinking water, interviewed approximately 9,000 individuals by using a standardized questionnaire and suggests that the overall average tap water consumption rate is 1.39 l/d (Cantor et al. 1987). According to the NCI's distribution data, 1.3 l/d is the approximate value of the 50th percentile and 2.0 l/d is the approximate value of the 90th percentile.

Gillies and Paulin (1983) suggest an average rate of 1.256 (+ 0.39) l/d and a 90th percentile rate of 1.9 l/d on the basis of a survey conducted in New Zealand. According to the data from the Food and Drug Administration's total diet study, Pennington (1983) reported an average daily fluid consumption rate for water and water-based foods of 1.2 l/d. The ICRP has summarized the intake levels for adults as ranging from about 0.4 l/d to about 2.2 l/d under normal conditions.

The EPA (1984) used data collected by the USDA in its 1977-1978 nationwide food consumption survey to determine daily beverage intake levels by age. The daily beverage intake levels for adults ranged from 1.24 to 1.73 l. The EPA (1990) has suggested that the average adult drinking water consumption rate is 1.4 l/d and the reasonable worst-case value is 2.0 l/d on the basis of the above studies. These values correspond to 510 l/yr and 730 l/yr, respectively, if 365 d/yr is used. Further evidence to support these values is provided by Pennington (1983) and Cantor et al. (1987) who report average total fluid intake rates of 1.7 l/d and 1.87 l/d among adults. Thus, the average water consumption rate should be less than the 2.0 l/d commonly used. Although very little data are available about the intake rate for the reasonable worst case, from the reported value of 1.90 l/d for the 90th percentile by Gillies and Paulin (1983) and 2.0 l/d by Cantor et al. (1987), it is reasonable to assume a worst case value of 2.0 l/d in risk assessment.

51.2 RESRAD DATA INPUT REQUIREMENTS

In the RESRAD code, the drinking water intake should be entered in units of liters per year (l/yr). The default value currently used in RESRAD is 510 L/yr.

52 REFERENCES

Agneessens, J.P., P. Dreze, and L. Sine, 1978, *Modelisation de la migration d'elements dans les sols. II. Determination du coefficient de dispersion et de la porosite efficace*, Pedologie *27*:373-388.

Alzona, J., et al., 1979, *Indoor-Outdoor Relationships for Airborne Particulate Matter of Outdoor Origin*, Atmospheric Environment, *13*:55-60.

American Industrial Hygiene Association, 1971, *Ergonomics Guides: Ergonomics Guide to Assessment of Metabolic and Costs of Physical Work*, American Industrial Hygiene Association Journal, *32*:560-564.

Amoozegar, A., and A. W. Warrick, 1986, *Hydraulic Conductivity of Saturated Soils: Field Methods*, in Methods of Soil Analysis, Part 1, A. Klute, ed., 2nd Ed., American Society of Agronomy, Inc., Madison, Wis., pp. 735-770.

Anspaugh, L.R., J.H. Shinn, and D.W. Wilson, 1974, *Evaluation of the Resuspension Pathway toward Protective Guidelines for Soil Contamination with Radioactivity*, International Atomic Energy Agency/World Health Organization Symposium on Radiological Safety Evaluation of Population Doses and Application of Radiological Safety Standards to Man and the Environment, Portoraz, Yugoslavia, May 20-24.

Aris, R., 1958, *On the Dispersion of Linear Kinematic Waves*, Proc. R. Soc. London, Ser. A, *245*:268-277.

Baes, III, C.F., and R.D. Sharp, 1983, *A Proposal for Estimation of Soil Leaching and Leaching Constants for Use in Assessment Models*, J. Environ. Qual., *12*:17-28.

Baker, D.A., G.R. Hoenes, and J.K. Soldat, 1976, *FOOD — An Interactive Code to Calculate Internal Radiation Doses from Contaminated Food Products*, Pacific Northwest Laboratory Report BNWL-SA-5523, Richland, Wash.

Barltrop, D., 1966, *The Prevalence of Pica*, American Journal of Diseases in Children, *112*:116-123.

Barreto, P.M.C., 1974, "Emanation Characteristics of Terrestrial and Lunar Materials and the Radon-222 Loss Effect on the Uranium-Lead System Discordance," Ph.D. Dissertation, Rice University.

Bear, J., 1988, *Dynamics of Fluids in Porous Media*, American Elsevier Publishing Company, Inc., 1972, Dover Publications, Inc.

Bedient, P.B., and W.C. Huber, 1988, *Hydrology and Floodplain Analysis*, Addison-Wesley Publishing Company.

Behrman, L.E., and V.C. Baughan, III, 1983, *Textbook of Pediatrics*, W.B. Saunders Company, Philadelphia, Pa.

Bellinger, D., et al., 1986, *Correlates of Low-Level Lead Exposure in Urban Children at 2 Years of Age*, Pediatrics 77:826-833.

Bicknell J., 1974, *Lead Poisoning in Childhood*, Update (England), *9*:653-659.

Binder, S., D. Sokal, and D. Maughan, 1986, *Estimating Soil Ingestion: The Use of Tracer Elements in Estimating the Amount of Soil Ingested by Young Children*, Archives of Environmental Health, *41*:341-345.

Blake, G.R., and K.H. Hartge, 1986b, *Particle Density*, in Methods of Soil Analysis, Part 1, A. Klute, ed., 2nd Ed., American Society of Agronomy, Inc., Madison, Wis., pp. 377-382.

Blake, G.R., and K.H. Hartge, 1986a, *Bulk Density*, in Methods of Soil Analysis, Part 1, A. Klute, ed., 2nd Ed., American Society of Agronomy, Inc., Madison, Wis., pp. 363-376.

Bond, W.J., and D.E. Smiles, 1988, *Predicting the Average Movement of Reactive Solutes in Soils*, Soil Use Mgmt., *4*:115-120.

Campbell, G.S. 1974, *A Simple Method for Determining Unsaturated Conductivity from Moisture Retention Data*, Soil Sci., *117*:311-314.

Cantor, K.P., et al., 1987, *Bladder Cancer, Drinking Water Source, and Tap Water Consumption: A Case-Control Study*, Journal of the National Cancer Institute, *79*:1269-1279.

Charney, E., J. Sayer, and M. Coulter, 1980, *Increased Lead Absorption in Inner City Children: Where Does the Lead Come From?* Pediatrics, *65*:226-231.

Cheng, J.-J., C. Yu, and A.J. Zielen, 1991, *RESRAD Parameter Sensitivity Analysis*, ANL/EAIS-3, prepared by Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy, Office of Environmental Restoration and Waste Management, August.

Chow, V.T., 1964, *Handbook of Applied Hydrology*, McGraw-Hill, New York.

Clapp, R.B., and G.M. Hornberger, 1978, *Empirical Equations for Some Soil Hydraulic Properties*, Water Resource Research, *14*:601-604.

Clausing, P., B. Brunekreef, and J.H. Van Wijnen, 1987, *A Method of Estimating Soil Ingestion by Children*, International Archives of Occupational and Environmental Health (W. Germany) *59*:73-82.

Cohen, B.L., 1979, *Methods for Predicting the Effectiveness of Uranium Mill Tailings Covers*, Nucl. Instr. and Methods, *164*:595-599.

Cooper M., 1957, *Pica*, Charles C. Thomas, Springfield, IL.

Coughtrey, P.J., D. Jackson, and M.C. Thorne, 1985, *Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems. A Compendium of Data*, A.A. Balkema, Netherlands.

Culot, M.V.J., H.G. Olson, and K.J. Schiager, 1976, *Effective Diffusion Coefficient of Radon in Concrete Theory and Method for Field Measurements*, Health Physics, *30*:263-270.

Currie, J.A., 1960a, *Gaseous Diffusion in Porous Media. Part 1. A Non-Steady State Method*, British Journal of Applied Physics, *11*:314-317.

Currie, J.A., 1960b, *Gaseous Diffusion in Porous Media. Part 2. Dry Granular Materials*, British Journal of Applied Physics, *11*:318-324.

Damkjaer, A., and U. Korsbech, 1985, *Measurement of the Emanation of Radon-222 from Danish Soils*, Sc. Total Environ., *45*:343.

Danckwerts, P.V., 1953, *Continuous Flow Systems*, Chem. Eng. Sci., *2*:1-13.

Danford, D.E, J.C. Smith, Jr., and A.M. Huber, 1982, *Pica and Mineral Status in the Mentally Retarded*, American Journal of Clinical Nutrition, *35*:958-967.

Danielson, R.E., and P.L. Sutherland, 1986, *Porosity*, in Methods of Soil Analysis. Part 1, A. Klute, ed., 2nd Ed., American Society of Agronomy, Inc., Madison, Wis., pp. 443-462.

Day, J.P., M. Hart, and M.S. Robinson, 1975, *Lead in Urban Street Dust*, Nature, *253*:343-345.

Duggan, M.J., and S. Williams, 1977, *Lead-in-Dust in City Streets*, Science of the Total Environment, *7*:91-97.

Feldman M.D., 1986, *Pica: Current Perspectives*, Psychosomatics (USA) *27*:519-523.

Fletcher, J.F., and W.L. Dotson (compilers), 1971, *HERMES — A Digital Computer Code for Estimating Regional Radiological Effects from the Nuclear Power Industry*, U.S. Atomic Energy Commission Report HEDL-TME-71-168, Hanford Engineering Development Laboratory.

Forfar, J.O., and G.C. Arneil, eds., 1984, *Textbook of Pediatrics*, 3rd ed., Churchill Livingstone, London.

Foster, G.R., 1979, *Sediment Yield from Farm Fields: The Universal Soil Loss Equation and Onfarm 208 Plan Implementation*, in Universal Soil Loss Equation: Past, Present, and Future, Soil Science Society of America, Special Publication No. 8, pp. 17-24.

Freeze, R.A., and J.A. Cherry, 1979, *Groundwater*, Prentice Hall, Inc.

Fries G.F., 1986, *Assessment of Potential Residues in Foods Derived from Animals Exposed to TCDD-Contaminated Soil*, paper presented at Dioxin 87: 6th International Symposium on Chlorinated Dioxins and Related Compounds, Fukuoka, Japan, Sept.

Gardner, W.H., 1986, *Water Content*, in Methods of Soil Analysis, Part 1, A. Klute, ed., 2nd Ed., American Society of Agronomy, Inc., Madison, Wis., pp. 493-544.

Gee, G.W., D. Rai, and R.J. Serne, 1983, *Mobility of Radionuclides in Soil*, in Chemical Mobility and Reactivity in Soil Systems, SSSA Special Publication, *11*:203-227.

Nazaroff, W.W., B.A. Moed, and R.G. Sextro, 1988, *Soil as a Source of Indoor Radon: Generation, Migration, and Entry*, in Radon and Its Decay Products in Indoor Air, W.W. Nazaroff and A.V. Nero, eds., John Wiley & Sons, New York.

Nero, A.V., 1988, *Radon and Its Decay Products in Indoor Air: An Overview*, in Radon and Its Decay Products in Indoor Air, W.W. Nazaroff and A.V. Nero, eds., John Wiley & Sons.

Nielson, K.K, D.C. Rich, and V.C. Rogers, 1982, *Comparison of Radon Diffusion Coefficients Measured by Transient-Diffusion and Steady-State Laboratory Methods*, U.S. Nuclear Regulatory Commission Report NUREG/CR-2875.

Nikula, A., 1982, *Sorption in Typical Finnish Soils and Some Fracture Filling of Bedrock* (in Finnish), Imatra Power Company, Helsinki, Finland.

Oztunali, O.I., et al., 1981, *Data Base for Radioactive Waste Management, Impacts Analyses Methodology Report*, NUREG/CR-1759, Vol.3, prepared by Dames and Moore, White Plains, N.Y., for Office of Nuclear Material Safety and Safeguards, U.S. Nuclear Regulatory Commission, Washington, D.C.

Pao, E.M., et al., 1982, *Foods Commonly Eaten by Individuals: Amount Per Day and Per Eating Occasion*, U.S. Department of Agriculture, Home Economics Report No.44.

Paustenbach, D.J, H.P. Shu, and F.J. Murray, 1986, *A Critical Examination of Assumptions Used in Risk Assessments of Dioxin Contaminated Soil*, Regulatory Toxicology and Pharmacology, *6*:284-307.

Pennington, J.A.T., 1983, *Revision of the Total Diet Study Food List and Diets*, Journal of the American Dietetic Association, *82*:166-173.

Pierce, R.S., D.T. Novieelo, and S.H. Rogers, 1981, *Commencement Bay Seafood Consumption Report*, Preliminary report, Tacoma-Pierce County Health Department, Tacoma, Wash.

Poffijn, A., et al., 1988, *On the Exhalation and Diffusion Characteristics of Concrete*, Radiation Protection Dosimetry, *24*:203-206.

Puffer H., 1981, *Consumption Rates of Potentially Hazardous Marine Fish Caught in the Metropolitan Los Angeles Area*, EPA Grant #R807 120010.

Relyea, J.F., and D.A. Brown, 1978, *Adsorption and Diffusion of Plutonium in Soil*, in Environmental Chemistry and Cycling Processes, D.C. Adriano, and I.L. Brisbin, Jr. eds., Proc. Symposium Held at Augusta, Ga., April 28-May 1, 1976, Technical Information Center, U.S. Dept. of Energy.

Rhodes, D.W., 1957, *Adsorption of Plutonium by Soil*, Soil Sci., *84*:465-471.

Rifai, M.N.E., W.J. Kaufman, and D.K. Todd, 1956, *Dispersion Phenomena in Laminar Flow through Porous Media*, Eng. Res. Ser. No. 93(2), Sanitary Eng. Res. Lab., Univ. of California, Berkeley.

Geraghy, J.J., et al., 1973, *Water Atlas of the United States*, Water Information Center, Port Washington, N.Y.

Gilbert, T.L., et al., 1983, *Pathways Analysis and Radiation Dose Estimates for Radioactive Residues at Formerly Utilized MED/AEC Sites*, ORO-832 (Rev.), prepared by Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations, Oak Ridge, Tenn., March (reprinted with corrections January 1984).

Gilbert, T.L., et al., 1989, *A Manual for Implementing Residual Radioactive Material Guidelines*, ANL/ES-160, DOE/CH/8901, prepared by Argonne National Laboratory for U.S. Department of Energy, June.

Gillies, M.E., and H.V. Paulin, 1983, *Variability of Mineral Intakes from Drinking Water: A Possible Explanation for the Controversy Over the Relationship of Water Quality to Cardiovascular Disease*, International Journal of Epidemiology, *12*:45-50.

Glickman L.T., 1981, *Pica Patterns, Toxocariasis, and Elevated Blood Lead in Children*, American Journal of Tropical Medicine and Hygiene, *30*:77-80.

Great Lakes Basin Commission, 1975, *Great Lakes Basin Commission Framework Study, Appendix 6: Water Supply — Municipal, Industrial, and Rural*, Sponsored by U.S. Environmental Protection Agency, Region V, Public Information Office, Ann Arbor, Mi.

Hauley, J.K., 1985, *Assessment of Health Risk from Exposure to Contaminated Soil*, Risk Analysis, *5*:289.

Healy, J.W., and J.C. Rodgers, 1979, *Limits for the Burial of the Department of Energy Transuranic Waste*, Los Alamos Scientific Laboratory Report LA-UR-79-100, Los Alamos, N.M.

Hillel, D., 1980a, *Applications of Soil Physics*, Academic Press, Inc.

Hillel, D., 1980b, *Fundamentals of Soil Physics*, Academic Press, Inc.

Hoeffner, S.L., 1985, *Radionuclide Sorption on Savannah River Plant Burial Ground Soil-A Summary and Interpretation of Laboratory Data*, Aiken, S.C., Savannah River Laboratory, E.I. du Pont de Nemours and Co., Report DP-1702.

Inoue, Y., and S. Morisawa, 1976, *Distribution Coefficient K_d of Radionuclides between Sample Soil and Water*, Nippon Genshiryoku Gakkai Shi, *18*:524-534.

International Commission on Radiological Protection, 1981, Report of the Task Group on Reference Man, Pergamon Press, N.Y.

International Commission on Radiological Protection, 1984, *A Compilation of the Major Concepts and Quantities in Use by ICRP*, ICRP Publication 42, Pergamon Press, Oxford.

Javitz H., 1980, *Seafood Consumption Data Analysis*, SRI International, final report prepared for U.S. Environmental Protection Agency Office of Water Regulations and Standards, EPA Contract 68-01-3887.

Robischon, P., 1971, *Pica Practice and Other Hand-Mouth Behavior and Children's Developmental Level*, Nursing Research, *20*:4-16.

Rogers, V.C., and K.K Nielson, 1984, *Radon Attenuation Handbook for Uranium Mill Tailings Cover Design*, U.S. Nuclear Regulatory Commission Report NUREG/CR-3533.

Rogers, V.C., and K.K Nielson, 1991, *Multiphase Radon Generation and Transport in Porous Materials*, Health Physics, *60*:807-815.

Rolston, D.E., 1986, *Gas Diffusivity*, in Methods of Soil Analysis, Part 1, A. Klute, ed., 2nd Ed., American Society of Agronomy Inc., Madison, Wis., pp. 1089-1102.

Rouston, R.C., G.S. Barney, and R.M. Smith, 1984, *Radionuclide Sorption on Low-Exchange Capacity Hanford Site Soils*, Commun. Soil Sci. Plant Anal., *15*:375-400.

Saxena, S.K., F.T. Lindstrom, and L. Boersma, 1974, *Experimental Evaluation of Chemical Transport in Water-Saturated Porous Media: 1. Nonsorbing Media*, Soil Sci., *118*:120-126.

Sayre, J.W., et al., 1974, *House and Hand Dust as a Potential Source of Childhood Lead Exposure*, American Journal of Diseases in Children, *127*:167-170.

Serdyukova, S.A. and Y.T. Kapitanov, 1978, *Radon Isotopes and Short-Lived Products of Their Disintegration in Nature*, Indian National Scientific Documentation Centre, New Delhi, Translation T72-51014, 1978, U.S.Department of Commerce, National Technical Information Service; from Russian, Moscow: Atomizdat, 1969.

Sheppard, M.I., and D.H. Thibault, 1990a, *A Four-Year Mobility Study of Selected Trace Elements and Heavy Metals*, J. Env. Qual.

Sheppard, M.I. and D.H. Thibault, 1990b, *Default Soil Solid/Liquid Partition Coefficients, K_ds, for Four Major Soil Types: A Compendium*, Health Physics, *59*:471-482.

Sheppard, S.C., and M.I. Sheppard, 1989, *Impact of Correlations on Stochastic Estimates of Soil Contamination and Plant Uptake*, Health Physics, *57*:653-657.

Sheppard, M.I., D.H. Thibault, and J.H. Mitchell, *Element Leaching and Capillary Rise in Sandy Soil Cores: Experimental Results*, J. Env. Qual., *16*:273-284.

Sheppard, M.I., et al., 1983, *Technetium and Uranium: Sorption by and Plant Uptake from Peat and Sand*, Health Physics, *44*:635-643.

Silker, W.B., 1981, *A Radon Attenuation Test Facility*, U.S. Nuclear Regulatory Commission Report NUREG/CR-2243.

Silker, W.B. and D.R. Kalkwarf, 1983, *Radon Diffusion in Candidate Soils for Covering Uranium Mill Tailings*, U.S. Nuclear Regulatory Commission Report NUREG/CR- 2924, PNL-4434.

Sisigina, T.I., 1974, Assessment of Radon Emanation from the Surface of Extensive Territories, in Nuclear Meteorology, Israeli Program of Scientific Translations, Jerusalem, p. 239.

Skopp, J., 1985, *Analysis of Solute Movement in Structured Soils*, in Proc. ISSS Symp. Water and Solute Movement in Heavy Clay Soil, J. Bouma and P.A.C. Raats, eds., International Institute for Land Reclamation and Improvement, Wagemingen, Netherlands, pp. 220-228.

Stranden, E., 1988, *Building Materials as a Source of Indoor Radon*, in Radon and Its Decay Products in Indoor Air, W.W. Nazaroff and A.V. Nero, eds., John Wiley & Sons, New York.

Strong, K.P., and D.M. Levins, 1982, *Effect of Moisture Content on Radon Emanation from Uranium Ore and Tailings*, Health Physics, *42*:27-32.

Strong, K.P., D.M. Levins, and A.G. Fane, 1981, *Radon Diffusion through Uranium Tailings and Earth Cover*, in Radiation Hazards in Mining: Control, Measurement, and Medical Aspects, M. Gomez, ed., Society of Mining Engineers, New York, p. 713.

Tanner, A.B., 1964, *Radon Migration in the Ground: A Review*, in Natural Radiation Environment, J.A.S. Adams and W.M. Lowder, eds., University of Chicago Press, Chicago, pp. 161.

Thamer, B.J., K.K. Nielson, and K. Felthauser, *The Effects of Moisture on Radon Emanation Including the Effects of Diffusion*, Report BuMines OFR 184-82, PB83-136358, U.S. Dept. of Commerce National Technical Information Service, Springfield, Va.

U.S. Department of Energy, 1989, *Radiation Protection of the Public and the Environment*, Draft DOE Order 5400.

U.S. Department of Agriculture, 1966, *Household Food Consumption Survey, 1965-1966, Report 12, Food Consumption of Households in the United States-Seasons and Year, 1965-1966*.

Uchida, S., and H. Kamada, 1987, *Sorption of Strontium on Soils in Layered and Aerated Zones*, Hoken Butsuri, *22*:179-187.

Valocchi, A.J., 1985, *Validity of the Local Equilibrium Assumption for Modeling Sorbing Solute Transport through Homogeneous Soils*, Water Resour. Res., *21*:808-820.

Van Genuchten M. Th. V and P.J. Wierenga, 1986, *Solute Dispersion Coefficients and Retardation Factors*, in Methods of Soil Analysis: Part I, Physical and Mineralogical Methods, Arnold Klute, ed., 2nd Ed., American Society of Agronomy, Inc. & Soil Science Society of America, Inc.

Veihmeyer, F.J., 1964, *Evapotranspiration* in Handbook of Applied Hydrology, V.T. Chow, ed., McGraw-Hill, New York.

Vermeer, D.E., and D.A. Frate, 1979, *Geophagia in Rural Mississippi: Environmental and Cultural Contexts and Nutritional Implications*, American Journal of Clinical Nutrition, *32*:2129-2135.

Walter, S.D., A.J. Yankel, and I.H. von Lindern, 1980, *Age-Specific Risk Factors for Lead Absorption in Children*, Archives of Environmental Health, *35*:53-58.

Wildung, M.W., and D.W. Rhodes, 1963, *Removal of Radioisotopes from Solution by Earth Materials from Eastern Idaho*, Phillips Petroleum Co., Atomic Energy Division, National Reactor Testing Station, U.S. Atomic Energy Commission, IDO-14624, Nov.

Wischmeier, W.H., and D.D. Smith, 1978, *Predicting Rainfall Erosion Losses - A Guide to Conservation Planning*, Agricultural Handbook No. 537, prepared by Science and Education Administration, Washington, D.C., for U.S. Department of Agriculture.

Yu, C., W.A. Jester, and A.R. Jarrett, 1984, *Simultaneous Determination of Dispersion Coefficients and Retardation Factors for a Low Level Radioactive Waste Burial Site*, Radioact. Waste Manage. Nucl. Fuel Cycle, *4*:401-420.

Zapalac, G.H., 1983, *Time-Dependent Method for Characterizing the Diffusion of Rn-222 in Concrete*, Health Physics, *45*:377-383.

Ziai, M., 1983, *Bedside Pediatrics*, Mohsen Ziai, Rochester, N.Y.

Distribution
Category:
UC-511

ARGONNE NATIONAL LABORATORY
9700 South Cass Avenue, Argonne, Illinois 60439

ANL/ES-160
DOE/CH/8901

A MANUAL FOR IMPLEMENTING RESIDUAL RADIOACTIVE MATERIAL GUIDELINES

A Supplement to U.S. Department of Energy Guidelines for Residual Radioactive Material at Formerly Utilized Sites Remedial Action Program and Surplus Facilities Management Program Sites

by

T.L. Gilbert,* C. Yu, Y.C. Yuan,‡ A.J. Zielen,
M.J. Jusko, and A. Wallo III§

Energy and Environmental Systems Division

June 1989

work sponsored by

U.S. DEPARTMENT OF ENERGY
Assistant Secretary for Nuclear Energy
Office of Remedial Action and Waste Technology

---

*Current address: Chicago Center for Religion and Science, Chicago, Illinois
‡Current address: Dames & Moore, West Valley Demonstration Project, West Valley, New York
§U.S. Department of Energy, Office of Remedial Action and Waste Technology

CONTENTS

CONTENTS (Cont'd)

CONTENTS (Cont'd)

TABLES

TABLES (Cont'd)

FIGURES

FIGURES (Cont'd)

FOREWORD

In the late 1970s and early 1980s, the U.S. Department of Energy (DOE) began work at several sites of the Formerly Utilized Sites Remedial Action Program (FUSRAP) and the Surplus Facilities Management Program (SFMP). These sites became contaminated as a result of previous activities for government nuclear programs. Criteria for cleanup of these sites were proposed on a site-by-site basis by the field offices responsible for the specific remedial actions. The criteria and even the approaches to development of cleanup criteria varied. In reviewing the proposed criteria, DOE Headquarters realized that general guidelines were needed for these activities. The major requirement identified was the need for criteria for residual radioactive material in soil. At the request of DOE Headquarters, the national laboratories made several attempts to develop soil criteria applicable to remedial actions. In 1983, DOE began to consolidate these criteria and associated procedures in order to identify generic soil criteria for all DOE remedial actions.

The DOE Headquarters established a working group comprised of representatives from the concerned headquarters program offices; the Office of Environment, Safety, and Health; DOE operations offices; and the national laboratories involved in development of specific criteria for decontamination efforts. The working group activities were also coordinated with representatives of the U.S. Environmental Protection Agency (EPA) and the U.S. Nuclear Regulatory Commission (NRC). These representatives attended several meetings of the working group in an advisory capacity.

The initial charter of the working group was to develop an acceptable set of generic soil criteria. However, recommendations from the first meeting of the complete group resulted in DOE expanding the charter to include all aspects of cleanup operations. Also, recognizing that soil criteria are significantly affected by many site-specific factors, the working group recommended that a generic procedure for deriving soil criteria should be included in the guidelines rather than generic soil concentration limits. The following is a summary of the working group's findings and recommendations:

1. The guidelines should be consistent with other available standards where they are appropriate; examples cited included

    - EPA standards for Uranium Mill Tailings Remedial Action (UMTRA) (40 CFR Part 192), which were in draft status during the initial part of the working group's activities;

- NRC surface contamination limits (Guidelines for Decontamination of Facilities and Equipment Prior to Release for Unrestricted Use or Termination of Licenses for Byproduct, Source, or Special Nuclear Material, July 1982); and

- DOE Orders.

2. For its residual radioactivity guidelines, DOE should adopt and use the recommendations and system of standards and models provided in Publications 26 and 30 of the International Commission on Radiological Protection (ICRP). This includes use of the concept of effective dose equivalent rather than critical organ dose (ICRP Publication 2) for defining dose limits for the general public and adoption of the lifetime average limit of 100 mrem/yr effective dose equivalent instead of the DOE limit for the general public of 500 mrem/yr in effect at that time.

3. Soil criteria for nuclides other than those covered under the UMTRA standard (i.e., radium and thorium) should be derived for each site according to the dose limit. These criteria should be based on a conservatively assumed plausible-use (realistic) scenario.

4. The as low as reasonably achievable (ALARA) process should be incorporated into the guidelines for all phases of a remedial action.

5. No attempt should be made by DOE to define "de minimis" or "below regulatory concern" levels.

The DOE guidelines for residual radioactive material were developed in accordance with these recommendations and were first issued in February 1985. The second revision was prepared and issued in March 1987. The revisions were generally procedural in nature and were directed toward resolving implementation problems experienced with the earlier version. The guidelines are supported by a separate, but integral, implementation manual (this document) that contains:

- Pathway analysis methodology for deriving soil criteria,

- Associated dose conversion factors,

- Environmental transport factors, and
- Guidance on applying the guidelines, including hot spot criteria and ALARA.

The methodology for deriving soil guidelines is coded in a microcomputer program, RESRAD. The code and this manual (which includes a user's guide for RESRAD) have been developed to support implementation of the DOE guidelines. The manual and code have been used in draft form to allow field testing, and the comments received over this test period have been incorporated into this code and manual.

The computer software for RESRAD has been developed under sponsorship of DOE. Any further distribution by any holder of the RESRAD software package (or other data therein) outside of DOE offices or other DOE contractors, unless otherwise specifically provided for, is prohibited without approval of the National Energy Software Center. Requests from outside DOE for DOE-developed computer software should be directed to the Director, National Energy Software Center, Argonne National Laboratory, 9700 South Cass Avenue, Argonne, Illinois 60439.

The RESRAD code is available on MS-DOS diskettes from the National Energy Software Center.

ACKNOWLEDGMENTS

This document is based on 1985, 1987, and 1988 draft versions. During this work, the authors were assisted by many experts from the scientific community. Special thanks are due to Keith Eckerman (Oak Ridge National Laboratory) for providing internal dose conversion factors (Chapter 3 of the 1985 draft); William Kennedy and Bruce Napier (Battelle Pacific Northwest Laboratory) for providing basic materials for the food chain pathways (Chapter 4 of the 1985 draft); Jack Healy (Los Alamos National Laboratory) and Joe Soldat (Battelle Pacific Northwest Laboratory) for contributions to earlier versions of the discussion of ALARA; and Wayne Hansen (Los Alamos National Laboratory) for providing valuable discussions and counsel for earlier versions of this document. The authors acknowledge the assistance of Carl Welty (U.S. Department of Energy [DOE] Office for Environment, Safety, and Health) and R.E. Baker (Energetics, Inc.) in reviewing the final ALARA section.

Many constructive comments and suggestions were received from those who reviewed the RESRAD code and the manual. The authors wish to thank Roy Eckart (University of Cincinnati), R.D. Freeberg (DOE), G.R. Grandbouche (DOE), Ken Karp (UNC Geotech), R. Murri (UNC Geotech), McLouis Robinet (Argonne National Laboratory), and Lars Soholt (Los Alamos National Laboratory).

Finally, we thank Dee Wyman for editorial assistance through the various drafts of this document.

NOMENCLATURE

The following is a list of the acronyms, initialisms, and abbreviations (including units of measure) used in this document.

ACRONYMS, INITIALISMS, AND ABBREVIATIONS

ALARA	as low as reasonably achievable
AMAD	activity median aerodynamic diameter
ASR	air/soil concentration ratio
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980, as amended
CFR	Code of Federal Regulations
DCF	dose conversion factor
DF	dietary factor
DSR	dose/source ratio
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
ETF	environmental transport factor
FA	area factor
FAR	food/air concentration ratio
FC	cover factor
FCD	cover and depth factor
FD	depth factor
FDW	fraction of drinking water from site
FQR	radionuclide transfer factor for meat and milk
FSR	food/soil concentration ratio
FUSRAP	Formerly Utilization Sites Remedial Action Program
FWR	food/water concentration ratio
GI	gastrointestinal
GM	geometric mean
GSD	geometric standard deviation
ICRP	International Commission on Radiological Protection
LLD	lower limit of detection
MB	mass-balance (model)
NCRP	National Council on Radiation Protection and Measurements
ND	nondispersion (model)
NRC	U.S. Nuclear Regulatory Commission
ORNL	Oak Ridge National Laboratory
QSR	fodder/soil or livestock-water/soil concentration ratio
SARA	Superfund Amendments and Reauthorization Act of 1986
SF	source factor
SFMP	Surplus Facilities Management Program
TDS	total dissolved solids
UMTRA	Uranium Mill Tailings Remedial Action
USLE	Universal Soil Loss Equation

ACRONYMS, INITIALISMS, AND ABBREVIATIONS (Cont'd)

WEF	water exposure factor
WSR	water/soil concentration ratio

UNITS OF MEASURE

cm	centimeter(s)
cm^3	cubic centimeter(s)
Ci	curie(s)
d	day(s)
g	gram(s)
h	hour(s)
ha	hectare(s)
kg	kilogram(s)
L	liter(s)
m	meter(s)
m^2	square meter(s)
m^3	cubic meter(s)
min	minute(s)
mm	millimeter(s)
mrem	millirem(s)
µs	microsecond(s)
ms	millisecond(s)
pCi	picocurie(s)
s	second(s)
yr	year(s)

A MANUAL FOR IMPLEMENTING RESIDUAL RADIOACTIVE MATERIAL GUIDELINES

by

T.L. Gilbert, C. Yu, Y.C. Yuan, A.J. Zielen,
M.J. Jusko, and A. Wallo III

ABSTRACT

This manual presents information for implementing U.S. Department of Energy (DOE) guidelines for residual radioactive material at sites identified by the Formerly Utilized Sites Remedial Action Program (FUSRAP) and the Surplus Facilities Management Program (SFMP). It describes the analysis and models used to derive site-specific guidelines for allowable residual concentrations of radionuclides in soil and the design and use of the RESRAD computer code for calculating guideline values. It also describes procedures for implementing DOE policy for reducing residual radioactivity to levels that are as low as reasonably achievable.

1 INTRODUCTION

This manual presents information for implementing U.S. Department of Energy (DOE) guidelines for residual radioactive material at sites identified by the Formerly Utilized Sites Remedial Action Program (FUSRAP) and the Surplus Facilities Management Program (SFMP). These guidelines (DOE 1987) are referred to herein as the "DOE guidelines" and are reproduced in Appendix A. The manual describes the analysis and models used to derive site-specific guidelines for allowable residual concentrations of radionuclides in soil and the design and use of a computer code for calculating guideline values. It also describes procedures for implementing DOE policy for reducing residual radioactivity to levels that are as low as reasonably achievable (ALARA).

A guideline is defined as a radionuclide concentration or a level of radiation or radioactivity that is acceptable if a site is to be used without radiological restrictions. Guidelines are expressed as (1) concentrations of residual radionuclides in soil, (2) concentrations of airborne radon decay products, (3) levels of external gamma radiation, (4) levels of radioactivity from surface contamination, and (5) concentrations of residual radionuclides in air and water. Soil is defined as unconsolidated earth material, including rubble and debris that may be present in earth material. Generic guidelines for thorium and radium in soil, airborne radon decay products, external gamma radiation, surface contamination, and residual radionuclides in air and water are specified in the DOE guidelines. Soil guidelines for other radionuclides must be derived on a site-specific basis using the DOE residual radioactive material code (RESRAD) described in Chapter 4 of this manual.

To derive site-specific soil guidelines, a basic radiation dose limit of 100 mrem/yr (as specified in the DOE guidelines) is applied to a member of a critical population group. The radiation dose is defined here as the effective dose equivalent from external radiation plus the committed effective dose equivalent from internal radiation (International Commission on Radiological Protection [ICRP] 1984, Section 2.1). The radiation dose limit is based on radiation protection standards and requirements specified in DOE Order 5400.xx (DOE 1989). The critical population group is a relatively small, homogeneous group that is representative of those individuals in the population expected to potentially receive the largest radiation dose. It is assumed, for the purpose of deriving soil guidelines, that the critical population group is a family that establishes residence on a site after the site has been released for use without radiological restrictions. The controlling principles for all guidelines are (1) the annual radiation dose received by a member of the critical population group from the residual radioactive material -- predicted by a realistic but reasonably conservative analysis and averaged over a time interval of 50 years -- should not exceed the basic dose limit of 100 mrem/yr and (2) doses should be kept as low as reasonably achievable. Final cleanup standards (authorized limits), and in some cases guidelines, may be based on a different scenario if the resident family scenario is not appropriate for the specific property or if another plausible-use scenario would result in significantly greater potential for exposure.

All significant exposure pathways for the critical population group must be considered in deriving soil guidelines. These pathways include:

- Direct exposure to external radiation from the contaminated soil material;
- Internal radiation from inhalation of airborne radionuclides; and

- Internal radiation from ingestion of
  - Plant foods grown in the contaminated soil,
  - Meat and milk from livestock fed with contaminated fodder and water,
  - Drinking water from a contaminated well, and
  - Fish from a contaminated pond.

The contribution from inhalation of radon decay products is not included in the current version of RESRAD because generic guidelines have been established for the concentration of radium in soil, which is the only source of radon.

The residues at some FUSRAP and SFMP sites might include material that is hazardous because of its chemical (nonradiological) toxicity. An analysis of the risks from chemical toxicity is outside the scope of this manual. If, however, there is evidence that residues with potential nonradiological hazards are present, a separate analysis should be carried out that is consistent with the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980, as amended by the Superfund Amendments and Reauthorization Act (SARA) of 1986, or other appropriate regulations; the purpose of this analysis would be to identify applicable or relevant and appropriate requirements and to ensure that the public is adequately protected.

Estimating the collective (population) dose is outside the scope of this manual because soil concentration guidelines do not explicitly take into account the collective dose to the general population. The collective dose is, however, an important consideration for applying the DOE policy of reducing the radiation dose received by all individuals (workers, critical groups, and the general population) to ALARA levels. Estimates of the collective dose to the general population may be provided in the documentation produced during the environmental analyses for site remedial action. The methods presented in this manual for deriving the dose to a critical population group of on-site residents could readily be extended to the general population by adding pathways for off-site exposure.

The family-farm scenario used for deriving site-specific soil concentration guidelines is, in most cases, considered credible in the long term. The scenario may be inapplicable in the short term because it is incompatible with currently allowed or feasible use of a site. Soil concentration guidelines derived by the procedures presented here may be unreasonably low for such use. Conversely, there may be situations in which residual radioactivity can be reduced to levels below guideline values at reasonable cost. "Authorized limits" are introduced in the DOE guidelines in order to provide for these situations.

Authorized limits are defined as concentrations of radionuclides and levels of radioactivity that must not be exceeded if the remedial action or decontamination effort is to be considered completed and the site is to be released for use without radiological restrictions. Authorized limits are set equal to guideline values unless (1) variations (supplemental limits or exceptions) specified in Section F of the DOE guidelines apply, in which case an authorized limit may be set above the corresponding guideline value for the specific location or condition to which the exception applies, or (2) it can be clearly established that limits below the guideline values are reasonable and that use of such limits are cost beneficial and comply with appropriate requirements (DOE guidelines, Sections E and F).

In addition to requiring that residual radioactivity be below guideline values, DOE also requires -- as a matter of policy -- that the ALARA process be applied to a site before it is released for use without radiological restrictions. Socioeconomic considerations, as well as technical feasibility, are taken into account in implementing this policy. The ALARA requirements apply to release of FUSRAP and SFMP sites for use with or without restrictions and to management of sites that cannot be released because it is not practicable to reduce the residual radioactivity below authorized limits.

Models for deriving soil concentration guidelines from dose limits are simplified representations of complex processes. It is not feasible to obtain sufficient data to fully or accurately characterize transport and exposure processes. Similarly, it is not possible to predict future conditions with certainty. Hence, there will be uncertainties in the guideline values. The models described in this manual and incorporated into RESRAD have been chosen to be realistic but reasonably conservative, and the calculated doses corresponding to guideline values of the radionuclide concentrations are expected to be reasonably conservative estimates (overestimates) of the actual doses.

The derivation of guideline values for radionuclide concentrations in soil is based on a pathway analysis method known as the concentration factor method (U.S. Nuclear Regulatory Commission [NRC] 1977; ICRP 1979-1982; Till and Meyer 1983; National Council on Radiation Protection and Measurements [NCRP] 1984). With this method, the relation between radionuclide concentrations in soil and the dose to a member of a critical population group is expressed as a pathway sum, which consists of a sum of products of "pathway factors." Pathway factors correspond to pathway segments connecting compartments in the environment between which radionuclides can be transported or radiation transmitted. Most pathway factors are steady-state ratios of concentrations in adjoining compartments. Some are factors for conversion from a radionuclide concentration to a radiation level or radiation dose, and others are use and occupancy factors that affect exposure. Each term in the sum corresponds to a pathway of connected segments. A pathway product or pathway factor can be added, deleted, or replaced without affecting the other pathways or pathway factors. This structuring facilitates the use of alternative models for different conditions or transport processes and the incorporation of additional pathways. Thus, RESRAD can be easily modified or tailored to model any given situation by merely adding or replacing factors or terms in the pathway sum.

This manual covers procedures only for deriving site-specific guidelines for radionuclide concentrations in soil and guidelines for implementing DOE's ALARA process. Problems associated with procedures for collecting and interpreting field measurements of residual radioactivity and protocols for determining whether the guidelines have been met are not covered. Guidance on these topics may be found in the summary protocol for identification, characterization, designation, remedial action, and certification of FUSRAP sites (DOE 1984, 1986a, 1986b) and the procedures manual for remedial action survey and certification activities (Oak Ridge National Laboratory [ORNL] 1982).

The information presented in this manual is organized as follows:

- Pathways and scenarios by which a member of the critical population group can be exposed to radiation -- Chapter 2;
- Derivation and application of soil guidelines -- Chapter 3;
- Description of the RESRAD code and instructions for its use -- Chapter 4;
- ALARA procedures and general guidelines for implementing the ALARA process -- Chapter 5;

- DOE guidelines for residual radioactive material -- Appendix A; and
- Models, formulas, and data used in RESRAD to calculate the pathway factors -- Appendixes B through F.

2 PATHWAY ANALYSIS

The pathway analysis for deriving soil concentration guidelines from a dose limit has four parts: (1) source analysis, (2) environmental transport analysis, (3) dose/exposure analysis, and (4) scenario analysis.

Source analysis addresses the problem of deriving the source terms that determine the rate at which residual radioactivity is released into the environment. This rate is determined by the geometry of the contaminated zone, the concentrations of radionuclides present, the rates of ingrowth and decay of the radionuclides, and the rate of removal by erosion and leaching. Qualitative aspects of these topics are discussed in Section 2.1; quantitative aspects are covered in Chapter 3 and the appendixes.

Environmental transport analysis addresses the problems of (1) identifying environmental pathways by which radionuclides can migrate from the source to a human exposure location and (2) determining the rate of migration along these pathways. The significant environmental pathways are identified and described in Section 2.2. The derivation of environmental transport factors that characterize the rate of migration is covered in Chapter 3 and Appendixes B through E.

Dose/exposure analysis addresses the problem of deriving dose conversion factors for the radiation dose that will be incurred by exposure to ionizing radiation. This problem is discussed in Section 2.3. Appendixes B, C, and D include tabulations of the dose conversion factors used to determine the effective dose equivalent incurred by exposure to external radiation or to internal radiation from inhalation or ingestion of radionuclides.

The parameters that control the rate of radionuclide release into the environment and the severity and duration of human exposure at a given location are determined by patterns of human activity referred to as exposure scenarios. The credible scenarios that are likely to result in the largest exposure to individuals are discussed in Section 2.4.

2.1 SOURCE TERMS

2.1.1 Geometry of the Contaminated Zone

A contaminated zone, defined as a belowground region within which radionuclides are present in above-background concentrations, is the common source term and starting point for all pathways. The derivation of soil guidelines is based on idealized contaminated regions of cylindrical shape, as shown in Figure 2.1, within which radionuclides are assumed to be uniformly distributed. A shape factor, which equals one if the actual contaminated zone is a circular cylinder and less than one if it is irregularly shaped, is used to correct for the difference between dose estimates for the actual and idealized areas. The shape factor is used for the external radiation exposure pathway (see Appendix B). The cover depth (C_d) corresponds to the distance to

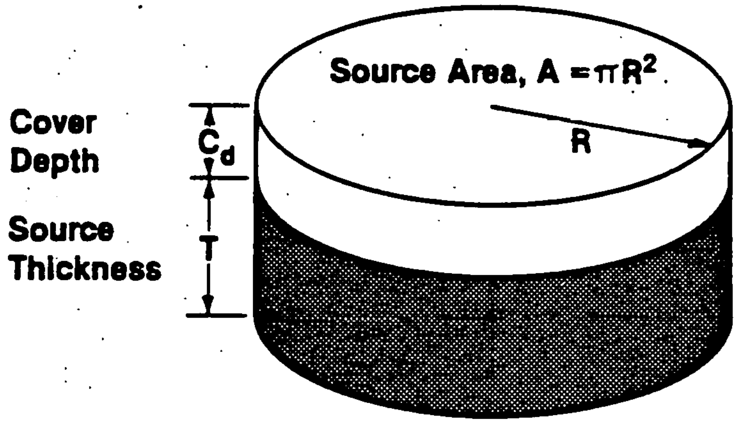


FIGURE 2.1 Geometry of Idealized Contaminated Zone

the uppermost contaminated soil sample, and the cover depth plus contaminated zone thickness ($C_d + T$) corresponds to the distance to the lowest contaminated soil sample.

If radionuclide distributions are approximately uniform throughout the contaminated region, the source geometry is a single cylinder that specifies a homogeneous contaminated zone. If the distributions are not approximately uniform, the source geometry consists of two or more cylinders that specify an inhomogeneous contaminated zone. Limiting radionuclide concentrations, referred to as single-radionuclide soil guidelines, are derived for each cylindrical volume. The manner in which the cylindrical volumes and soil guidelines are determined is described in Sections 3.1 and 3.3.

2.1.2 Time Dependence

The time dependence of the annual dose incurred by a member of the critical population group is controlled by (1) the rate at which radionuclides are leached from the contaminated zone, (2) the rate of ingrowth and decay of the radionuclides, (3) the rate of erosion of the cover and contaminated soil material, and (4) the rate of contaminant transport through the environmental pathways. The first three of these processes occur within, or primarily within, the contaminated zone; the last process occurs outside the contaminated zone. The models used to account for the time dependence of different transport and erosion processes are described in Appendixes B through E; the models used to account for the rate of ingrowth and decay and leaching are described in Appendix F.

2.1.3 Redistribution

A contaminated zone is treated as a single homogeneous or inhomogeneous source of changing thickness and radionuclide concentrations due to leaching, erosion, and ingrowth and decay. Erosion or human activities (such as excavation for a building erected in the contaminated zone) can result in redistribution of the contaminated soil that creates new contaminated zones. Such zones are not explicitly incorporated into RESRAD. If the topography and circumstances are such that redistribution is likely to occur, the new contaminated zone should be taken into account. This can be done by estimating the initial area and thickness of the new contaminated zone and then deriving its contribution to the soil guidelines in the same manner as for the original contaminated zone (see Section 2.3).

Radionuclides can also be redistributed by use of irrigation water that has been contaminated by radionuclides leached from the original contaminated zone. This contribution is taken into account in the derivation of the contribution from food pathways.

2.2 PATHWAYS

2.2.1 Pathway Identification

Potential pathways are identified in Table 2.1. The three major headings correspond to the three exposure pathways by which radionuclides can enter the body. In the first, exposure is by external radiation from radionuclides outside the body. In the second and third, exposure is by internal radiation from radionuclides that are inhaled or ingested. These three types of exposure correspond to the three kinds of dose conversion factors discussed in Section 2.3.

TABLE 2.1 Potential Pathways[a]

---

1 External Radiation
  1.1 Ground
    1.1.1 Volume Source*
    1.1.2 Surface Source*[b]
  1.2 Air
    1.2.1 Dust
    1.2.2 Radon and Radon Decay Products
    1.2.3 Other Gaseous Airborne Radionuclides
  1.3 Water

2 Inhalation
  2.1 Dust*
  2.2 Radon and Radon Decay Products
  2.3 Other Gaseous Airborne Radionuclides

3 Ingestion
  3.1 Food
    3.1.1 Plant Foods (Vegetables, Grains, and Fruits)*
    3.1.2 Meat*
    3.1.3 Milk*
    3.1.4 Aquatic Foods (Fish, Crustacea, and Mollusks)*
  3.2 Water
    3.2.1 Groundwater (Well)*
    3.2.2 Surface Water*
  3.3 Soil

---

[a] An asterisk indicates the pathways used to derive site-specific soil guidelines.

[b] The surface source can be approximated by assuming a very thin layer of contamination (e.g., 0.1 cm).

For each exposure pathway, radionuclides can migrate from a source to a human exposure location by many environmental pathways. The major categories of environmental pathways are listed in Table 2.1. The items listed should be regarded as environmental pathway categories rather than individual pathways because many of the items can correspond to more than one pathway and some of the items can occur as segments in more than one pathway. For example, there are many different plant food pathways, and contaminated groundwater can contribute to the human drinking water pathway and also to several food pathways if contaminated water is used to irrigate crops or water livestock.

Major pathways used to derive site-specific soil guidelines in the RESRAD code are identified in Table 2.1 by an asterisk and shown diagrammatically in Figure 2.2. Minor pathways for on-site exposure (all remaining pathways except the radon inhalation pathway) are not taken into account in deriving soil guidelines because the dose contribution from these pathways is expected to be insignificant. External radiation from a surface layer formed by redeposition of airborne radionuclides carried by the wind from an exposed contaminated zone is expected to be insignificant compared with external radiation from the residual radioactive material in its original location. External radiation from contaminated water is expected to be insignificant compared with internal exposure from radionuclides ingested in drinking water. The external radiation dose from airborne dust is much smaller than the inhalation dose from dust (by a factor of 100 or more for radionuclides in the U-238 series [Gilbert et al. 1983, Tables 5.6-5.8]). The external radiation dose from airborne radon decay products is negligible compared with (1) the internal inhalation dose to the bronchial epithelium (the critical organ for inhalation of radon decay products), (2) the external radiation dose from the parent radium in the soil, or (3) the internal radiation dose from

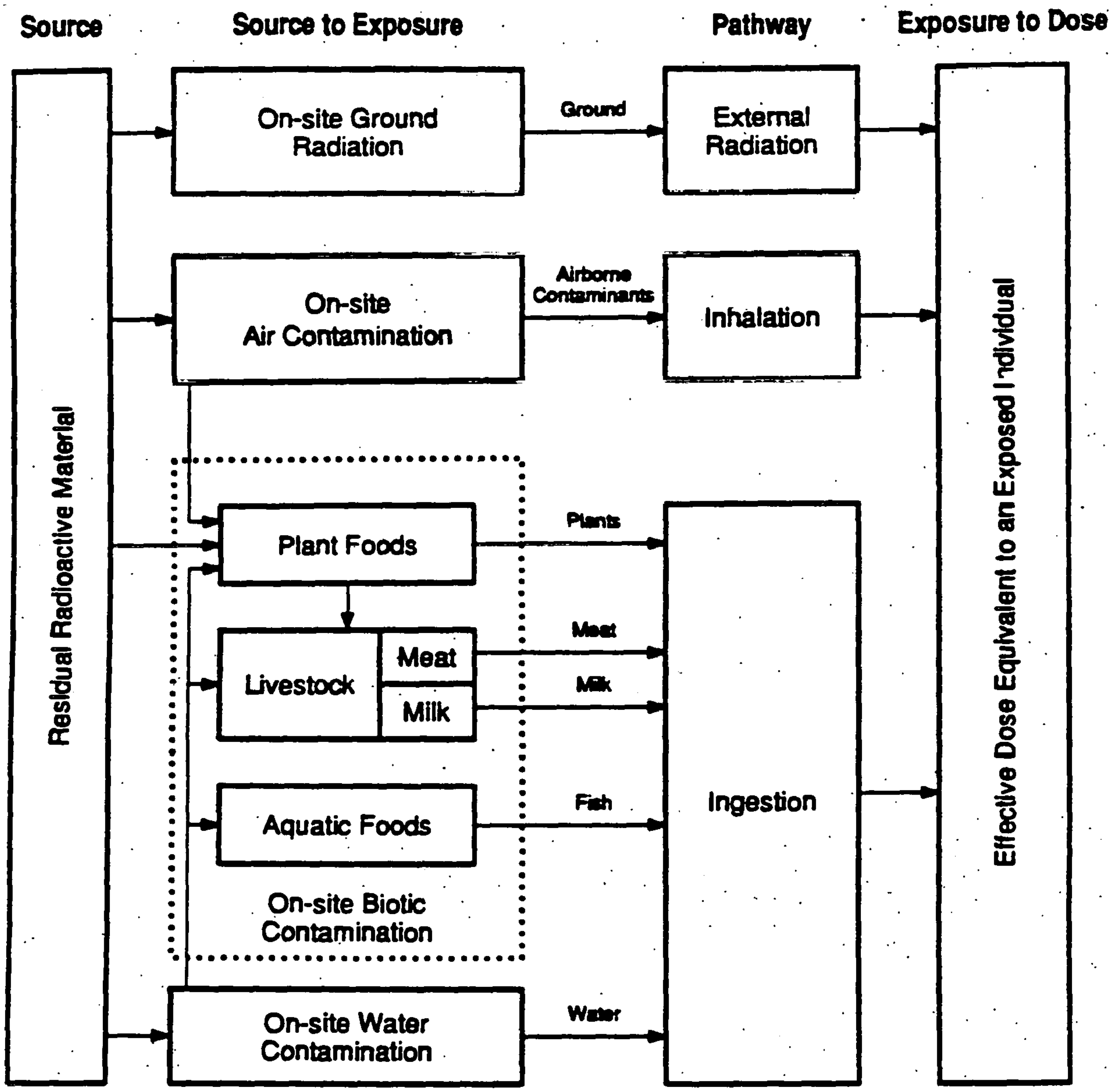


FIGURE 2.2 Exposure Pathway Diagram for Calculating the Dose to an On-site Resident from Residual Radioactive Material at a FUSRAP or SFMP Site (Source: Modified from Gilbert et al. 1983, Figure 2.4)

ingestion of plant foods grown in the radium-contaminated soil (Gilbert et al. 1983). The contributions from gaseous airborne radionuclides other than radon decay products (e.g., from C-14 occurring in CO_2 or from tritium occurring in tritiated water vapor) is insignificant at FUSRAP and SFMP sites. The soil pathway corresponds to direct ingestion of soil (Healy 1977). This pathway is significant only for a child afflicted with pica (a compulsive craving for nonfood objects), which is a rare occurrence limited to a period of about one year in a child's life; thus, it makes only a small contribution to the lifetime dose.

Internal radiation from inhalation of radon decay products is a major -- in many circumstances, the dominant -- dose contributor when radium is present in the soil. The radon inhalation pathway is not included in the current version of RESRAD because generic soil guidelines have been established for radium (see Appendix A). This pathway may be added in a later version to allow guidelines to be derived in a more consistent manner for hot spots and mixtures in which radium occurs.

2.2.2 External Radiation Pathways

Gamma radiation from radionuclides distributed throughout the contaminated zone is the dominant external radiation pathway and the only external radiation pathway taken into account in calculating soil guidelines.* The dose due to external gamma radiation is first calculated for an individual exposed continuously to radiation from an infinite contaminated zone at a distance of one meter from the ground surface. Correction factors are then

---

*The dose contribution from electrons (beta particles) is primarily restricted to the skin; the dose contribution from neutron radiation can be significant for some transuranic radionuclides and should be considered if neutron-emitting radionuclides are known to be present.

applied for the finite area and thickness of the contaminated zone, shielding by a cover of uncontaminated soil (if $C_d > 0$; see Figure 2.1), irregular shape, shielding by the floors and walls of a house, and less-than-continuous occupancy. Quantitative details are presented in Chapter 3 and Appendix B.

2.2.3 Inhalation Pathways

Inhalation exposure results primarily from inhalation of radon decay products and contaminated dust. As noted earlier, the radon pathway is not included in the current version of RESRAD because generic soil guidelines have been established for radium, which is the source of all radon (see Appendix A). The following discussion is, therefore, limited to the dust pathway.

An inhalation pathway consists of two segments: (1) an airborne exposure segment linking the source (contaminated zone) with the airborne radionuclides at an exposure location and (2) an inhalation segment linking the airborne radionuclides with the exposed individual. The inhalation segment is characterized by an occupancy factor (equivalent fraction of time during which an individual inhales contaminated air) and a factor for the inhalation rate. Numerical values for these factors may be obtained by well-established procedures (ICRP 1975; Momeni et al. 1979). The airborne exposure pathway segment is the critical segment. It is characterized by the air/soil concentration ratio, defined as the ratio of the airborne concentration of a radionuclide at a human exposure location to the concentration in the soil. The air/soil concentration ratio for contaminated dust depends on the complex processes by which soil particles become airborne by resuspension and are transported to an exposure location. It is used in the food chain pathways (for the foliar deposition subpathways) as well as in the inhalation pathways.

Modeling the airborne exposure pathway segment consists of two steps: (1) modeling the process by which radionuclides become airborne and (2) modeling the process by which the airborne radionuclides are transported to a human exposure location and diluted before inhalation. The first step gives the ratio of the concentration in air near the source before it is dispersed and diluted to the concentration in the resuspendable layer of dust; the second step gives the ratio of the airborne concentration at the point of exposure to the undiluted airborne concentration at the source. Quantitative details are presented in Appendix C.

2.2.4 Ingestion Pathways

2.2.4.1 Food Pathways

Four food pathway categories are taken into account: plant foods, meat, milk, and aquatic foods. The plant food pathway category is divided into four subcategories corresponding to (1) root uptake from crops grown in the contaminated zone, (2) foliar uptake from contaminated dust deposited on the foliage, (3) root uptake from contaminated irrigation water, and (4) foliar uptake from contaminated irrigation water. The plant food pathway subcategories are applicable to livestock fodder; hence, they are applicable to the pathways through which meat and milk become contaminated by ingestion of contaminated fodder. There is a fifth subcategory for the meat and milk pathways corresponding to contamination by ingestion of contaminated livestock water. The aquatic food pathway is for ingestion of fish, crustacea, and mollusks from a nearby pond that has been contaminated by radionuclides that have been leached from the contaminated zone.

The food pathways are activated by scenarios in which crops are grown in or close to the contaminated zone. Plant crops grown in the contaminated zone will be the dominant and most frequent contributor, especially if the crops are irrigated with contaminated water. Contributions from the meat and milk pathways, which involve an additional pathway segment for transfer of radionuclides from fodder or water to the meat or milk, will generally be smaller but not insignificant. Vegetable gardens are common in urban and suburban areas as well as rural areas, whereas raising livestock is generally limited to rural areas. The aquatic food pathway will occur only in areas where the topography and soil characteristics are favorable for building a pond.

The food pathways may be classified as water-independent or water-dependent. The direct root uptake and foliar dust deposition pathways for plant foods, meat, and milk are water-independent (where water refers to the water that has been contaminated by radionuclides leached from the contaminated zone). Water-dependent pathways are the irrigation water pathway for plant foods, meat, and milk; the livestock water pathway for meat and milk; and the aquatic food pathway. In regions where natural rainfall is the only source of water used in raising crops, the only water-dependent pathways will be the livestock water branch of the meat and milk pathways and the aquatic food pathway. If ditch irrigation is used, the contribution from root uptake of contaminated irrigation water by plant foods and fodder will be added. If overhead irrigation is used, the contribution from the subpathways for foliar and root uptake from irrigation water will be added.

The water-independent pathways are assumed to contribute to the dose as soon as a family establishes a residence and garden on the site. The time dependence of these pathways is determined by the time dependence of the cover and contaminated zone thickness and the radionuclide concentrations in the contaminated zone. The contribution from water-dependent pathways will be delayed until radionuclides transported by groundwater reach a point of water withdrawal (i.e., well or pond). The time dependence of these pathways is determined by the time dependence of the radionuclide concentrations in the contaminated water, as determined by the hydrological model used for the groundwater pathway segment. A fraction of a radionuclide will have been leached from the root zone before the radionuclide first reaches a point of water withdrawal in above-background concentrations (the breakthrough time); hence, the contributions to the dose from the water-independent and water-dependent pathways will occur at different times.

After the breakthrough time, the contaminated irrigation water will create a new contaminated zone as it percolates down through the soil. The contribution of this secondary contaminated zone to pathways other than the food pathway is not taken into account because it is assumed to be small compared with the contributions of the food pathway.

Radionuclide transport through the food pathways is determined by the quantities of different foods consumed (dietary factors), the fraction of the diet from foods that are contaminated by radionuclides from the contaminated zone (which is determined by the fraction raised locally and the area of the contaminated zone), the cover depth and contaminated zone thickness relative to the root zone of the plants, the various transfer factors from root or foliage to plants and from fodder or water to meat or milk, and the concentrations of radionuclides in water that has percolated through the contaminated zone. The factors used to take these effects into account are discussed in Chapter 3 and Appendixes D and E.

2.2.4.2 Water Pathway Segments

A water pathway segment connects the contaminated zone with a point of water withdrawal for drinking or irrigation or with a pond where aquatic foods are raised for human consumption. It is characterized by a water/soil concentration ratio for each radionuclide, defined as the ratio of the radionuclide concentration in the water at the point of withdrawal or use to the radionuclide concentration in the contaminated zone. Irrigation and drinking water are assumed to be taken from a well or pond. The well is assumed to be either in the center of or at the downgradient edge of the contaminated zone (see Appendix E). The pond water is assumed to be contaminated by water that seeps to the surface after percolating through the contaminated zone. Figure 2.3 schematically represents the process by which natural precipitation or irrigation water infiltrates the contaminated zone and transports radionuclides through the unsaturated (vadose) zone and saturated zone (aquifer) to a well or point of seepage into surface water.

Groundwater Pathway Segment. Two models are used for calculating the water/soil concentration ratio for the groundwater pathway segment: a mass-balance (MB) model and a nondispersion (ND) model.* The MB model assumes that all of the radionuclides released annually from the contaminated zone are withdrawn through a well located at the center of the contaminated zone. The ND model assumes that the dispersivity is nil, the vadose zone and aquifer are homogeneous, the well is located at the downgradient edge of the contaminated zone, and water withdrawal introduces only a minor perturbation in the water

---

*The mass-balance model is an adaptation of a model proposed by staff of the NRC (Neuder 1986).

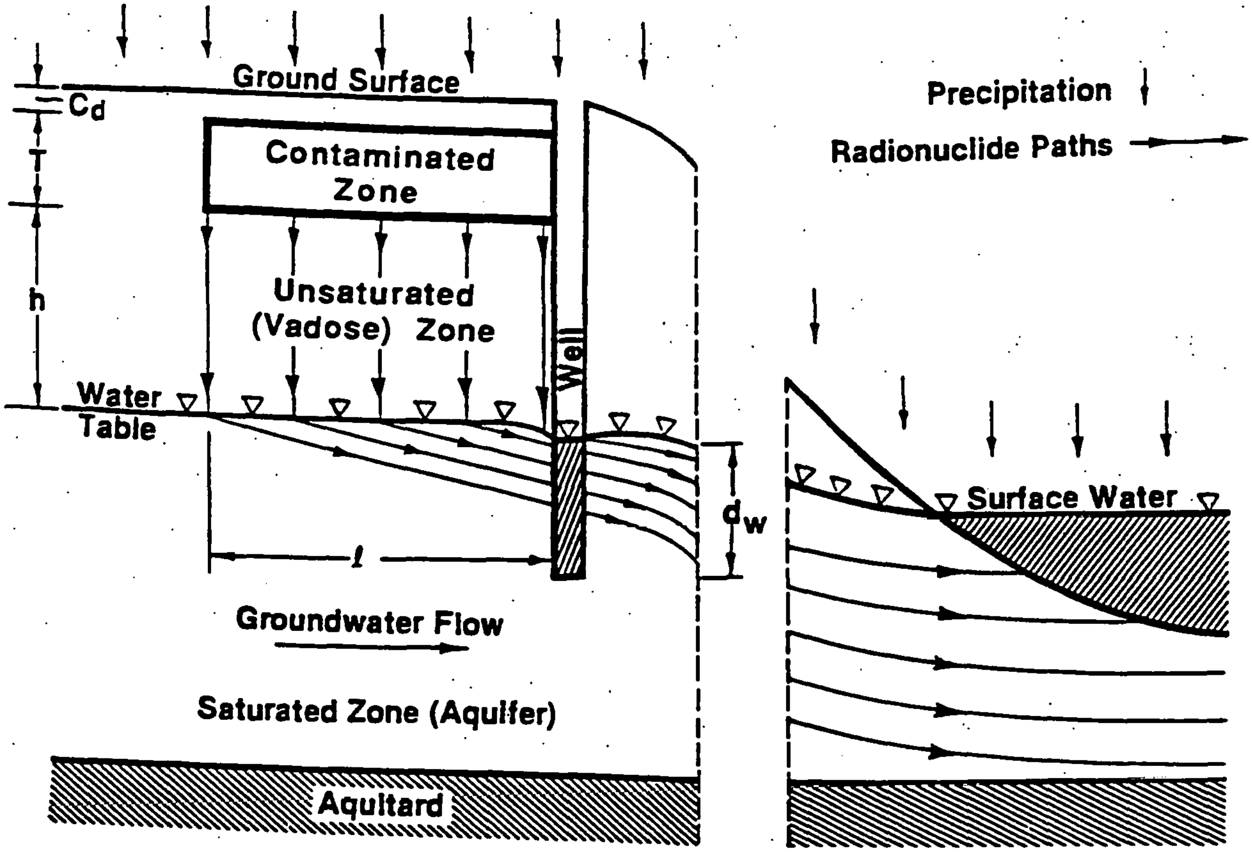


FIGURE 2.3 Schematic Representation of the Water Pathway Segments

flow. These assumptions lead to a flow pattern from which the dilution factor can be estimated by geometric considerations. The MB model is used for smaller contaminated areas (e.g., 1,000 m^2 or less), and the ND model is used for larger contaminated areas. The MB and ND models are discussed in more detail in Appendix E.

The groundwater pathway segment is analyzed in terms of parameters that remain valid for more sophisticated models. These parameters consist of the breakthrough time, rise time, and dilution factor. The breakthrough time is the time from a radionuclide's release into infiltrating water at the contaminated zone until its detection at the point of withdrawal or use. The rise time is the time from initial detection until the concentration reaches a maximum value when the release rate from the contaminated zone is constant after initiation. (A linear rise is assumed; a straight-line approximation can be used when the actual rise has a more complicated time dependence.) The dilution factor is the steady-state ratio of a radionuclide concentration at the point of withdrawal or use to the concentration of the same radionuclide in infiltrating water as it leaves the contaminated zone. Formulas for calculating the breakthrough times, rise times, and dilution factors for the MB and ND models are given in Chapter 3 and Appendix E.

The groundwater pathway models implemented in the RESRAD code apply only to situations for which the hydrological strata can reasonably be approximated by a sequence of uniform, horizontal strata. For sites having more complicated strata such as fracture zones, simple models may be used to provide reasonable estimates if a set of effective hydrogeologic flow parameters are used. However, the accuracy of the results predicted is determined by the accuracy of the effective input parameters used (Yu et al. 1986).

Surface Water Pathway Segment. The surface water is assumed to be a pond for which (1) the water inflow and outflow are in steady-state equilibrium and (2) the annual inflow of radioactivity into the pond equals the annual quantity of radioactivity leached from the contaminated zone. The dilution factor is assumed to be the ratio of the annual volume of water that infiltrates the contaminated zone to the annual total inflow of water into the pond. If, as also assumed, the infiltrating water flow is vertically downward through the contaminated zone, the surface water dilution factor is given by the ratio of the area of the contaminated zone to the area of the watershed that supplies the pond. Breakthrough time and rise time are assumed to be the same as for an on-site well; no credit is taken for the additional time for radionuclides to be transported from the edge of the contaminated zone to the point of seepage. This simplified model will give a conservative estimate of the water/soil concentration ratio for a pond.

2.2.4.3 Drinking Water Pathway

The concentration factor that characterizes the drinking water pathway is obtained by multiplying the water/soil concentration ratio by the annual quantity of contaminated drinking water consumed by an individual. Both well water and surface water can be used for drinking. The fraction of well water balanced from surface water is used to calculate the total contribution from groundwater and surface water. The default value for this fraction is unity, i.e., 100% of the drinking water is drawn from a well.

2.3 DOSE CONVERSION FACTORS

The distribution of dose within the body depends upon the type of radiation involved and the location of the radionuclides emitting the radiation, i.e., within or external to the body. The ICRP has provided a useful quantity that, on a risk scale, can be applied to any radiation pattern (ICRP 1977, 1978, 1979-1982). This quantity -- the effective dose equivalent -- is the weighted sum of the dose equivalent in various organs. The weighting factors for the organs are proportional to the potential risk associated with the irradiation of those organs. All dose factors used for deriving soil concentration guidelines are based on the effective dose equivalent.

Internal radiation doses to organs and tissues of the body are frequently estimated with factors representing the committed dose equivalent for a unit intake of a radionuclide via inhalation or ingestion. In the case of external radiation, the dose rate in organs depends on the concentration of the radionuclide in the environment. These relationships are expressed as dose/exposure ratios called "dose conversion factors." A dose conversion factor is the ratio of the committed effective dose equivalent (for internal exposure) or the effective dose equivalent rate (for external exposure) to the quantity of a radionuclide inhaled or ingested (for internal exposure) or the concentration of a radionuclide in the air, water, or ground (for external exposure).

2.3.1 Ingestion and Inhalation

The radiation dose from inhalation and ingestion of radionuclides has been systematically evaluated by the ICRP in its Publication 30 (ICRP 1979-1982). This effort was undertaken to compute secondary limits for occupational exposures corresponding to the primary radiation protection guidance of Publication 26 (ICRP 1977). Dose equivalents in organs or tissues of the body are calculated with models that (1) describe the entrance of materials into the body (respiratory and gastrointestinal [GI] tract) and the deposition and subsequent retention of the radionuclides in body organs (referred to as metabolic models), and (2) estimate the energy deposition in tissues of the body (ICRP 1979-1982). In applying the results of these calculations, some insight into the details of models for the lung and GI tract is required because these models serve as the interface between humans and the environment.

Inhalation of radionuclides attached to airborne particles is a potential route for intake of radionuclides into the body. The ICRP model of the respiratory tract divides the tract into three regions: nasopharyngeal, tracheobronchial, and pulmonary (ICRP 1966). These regions are connected with one another as well as with body fluids and the GI tract. The fraction of inhaled activity deposited in these regions is a function of the size of the airborne particles. The activity median aerodynamic diameter (AMAD) is the measure most widely used to characterize the aerosol. The rate at which the deposited material is removed from the regions is considered to be independent of particle size; the rate is, however, related to the chemical form of the particles. Chemical compounds are assigned to one of three classes to characterize their removal rate from the lung. The inhalation classes are denoted as D, W, and Y — corresponding to clearance or removal times from the pulmonary region of the lung on the order of days, weeks, or years, respectively. Details are provided in the ICRP task group's report (ICRP 1966) and subsequent ICRP publications (ICRP 1968, 1979-1982).

A portion of the material initially deposited in the lung enters the GI tract and, of course, ingested radionuclides enter the GI tract directly. The ICRP model of the GI tract divides the tract into four compartments: stomach, small intestine, upper large intestine, and lower large intestine (Eve 1966). Absorption of materials into body fluids is generally considered to occur within the small intestine. The fraction of the ingested material absorbed into body fluids from the tract is denoted as f_1. The numerical value of f_1 depends on the chemical form of the ingested material (ICRP 1979-1982).

Radionuclides that enter body fluids from either the lung or the GI tract may be deposited within the various organs of the body. Metabolic processes and radioactive decay reduce the radioactivity in the body. These removal processes are modeled in a rather simple manner in which the organs are represented by a number of mathematical compartments from which the removal rate is directly proportional to the amount of radionuclide present (ICRP 1979-1982). This approach leads to functions involving a sum of exponentials to describe the time course of activity within the body. An exception to this representation occurs for radium, for which the more-detailed metabolic model of ICRP Publication 20 (ICRP 1972) is used.

In calculating the dose from intake of radionuclides into the body, the occurrence of radioactive decay products must be considered. When an atom undergoes radioactive decay, the new atom formed may also be radioactive and thus contribute to the dose. Although these decay products may be treated as independent radionuclides in external exposures, they must be considered in conjunction with the parent in the evaluation of radionuclides inhaled or ingested because their distribution among the organs of the body depends on the metabolism of the parent (ICRP 1979-1982). Thus, the contributions to dose from decay products formed within the body are included in the dose conversion factor for any radionuclide with radioactive decay products.

Factors representing the committed effective dose equivalent per unit intake of radionuclides are given in Appendix C (Table C.1) for inhalation and Appendix D (Table D.1) for ingestion. These values were taken from a DOE report (DOE 1988a); similar values are given in an EPA report (Eckerman et al. 1988). In many instances, several chemical forms have been considered; that is, more than one clearance class (inhalation) or f_1 (ingestion) value is given for the radionuclide. If the user of this manual has information indicating the chemical forms expected in the particular environment being considered, then the appropriate value should be selected after consulting ICRP Publication 30 (ICRP 1979-1982). If no information on the relevant chemical forms is available, then the most conservative value (i.e., the highest estimate of dose) should be used.

Dose conversion factors identified with "+D" in Tables C.1 and D.1 are the sums of dose conversion factors for a principal radionuclide (a radionuclide with a half-life greater than one year) and its associated decay chain (all short-lived decay products of a principal radionuclide down to, but not including, the next principal radionuclide or the final nonradioactive nuclide in the chain). These aggregated dose conversion factors correspond to ingestion or inhalation of the principal radionuclide together with its associated decay product radionuclides, which are assumed to be in secular equilibrium at the time of intake (see Section 3.1 and Table 3.1).

2.3.2 External Radiation

Organs of the body may be irradiated by radiation emitted from radionuclides present in the environment. Gamma radiation is the only external radiation taken into account in calculating soil guidelines (see Section 2.2.2). Gamma radiation is of particular concern because this radiation is sufficiently penetrating that the dose at a given location depends on the spatial distribution over considerable distances. In addition, the dose distribution pattern within the body is rather uniform. The actual distribution of dose in the body depends somewhat on the distribution pattern of the radionuclide in the environment. An idealized distribution pattern that is often assumed is a uniform distribution of the radionuclide within an infinite or semi-infinite region. By considering such idealized distributions, dose conversion factors relating the effective dose equivalent rate to the radionuclide concentration can be tabulated. For other situations, the details of radionuclide distribution must be included in the numerical calculation of dose.

Kocher (1983) has published extensive tabulations of dose-rate factors for radionuclides distributed in air, in water, and on the ground surface. The methods of Kocher and Sjoreen (1985) were used in this manual to derive dose-rate factors of radionuclides uniformly distributed in the soil volume. Dose conversion factors for exposure from radionuclides in or on the ground are presented in Appendix B (Table B.1). The dose-rate factors for surface contamination were taken from a DOE report (DOE 1988b).

The dose conversion factors for ground contamination are for exposure at a point one meter above the ground. The volume contamination is assumed to be uniformly distributed to infinite depth and infinite lateral extent. Contributions from all associated radionuclides, which are assumed to be in secular equilibrium with the parent principal radionuclide, are included in the dose conversion factors for principal radionuclides. (Note: "infinite" dimensions reflect consideration of radiation transport and may in fact correspond to small physical dimensions. For example, one meter of soil is generally infinite with respect to the dose rate from a photon-emitting radionuclide.)

Dose conversion factors for radionuclides distributed in the ground depend on the bulk density of the soil (ρ_b). Factors for ρ_b = 1.0 g/cm^3 and ρ_b= 1.8 g/cm^3 are listed in Table B.1. Values for other soil densities may be obtained by interpolation.

2.4 EXPOSURE SCENARIOS

Many parameters that determine the quantity of radionuclides or radiation to which an individual is exposed are determined by exposure scenarios, i.e., patterns of human activity that can affect release of radioactivity from the contaminated zone and the amount of exposure received at the exposure location. Soil guidelines are based on a family-farm exposure scenario. This scenario activates all environmental pathways for on-site or near-site exposure and is expected to result in the highest predicted lifetime dose.

The exposure of construction workers or scavengers is unlikely to last longer than a few months and would generally be limited to working hours. The lifetime exposure for construction workers and scavengers is, therefore, unlikely to exceed the lifetime dose for a permanent on-site resident. (The basic dose limit of 100 mrem/yr is based on the lifetime dose [ICRP 1977]. A dose limit of 500 mrem/yr is allowable for some years if the dose averaged over a lifetime does not exceed the basic dose limit of 100 mrem/yr [DOE 1989].) Exposure of workers in on-site industrial or commercial buildings can also occur, but this exposure will generally be less than that of residents because the exposure will be limited to working hours and will not include contributions from ingestion of foods grown on-site.

Soil guidelines are based on on-site exposure because on-site residents will receive a radiation dose that is at least as large as the dose to off-site residents and generally larger. The radiation dose for off-site residents will decrease with increasing distance from the site. The external radiation dose will decrease rapidly with distance from the site, and secondary off-site sources -- such as surface deposits of airborne contaminated soil or water contaminated by radionuclides leached from the soil -- will have lower radionuclide concentrations. The contributions from inhalation pathways will decrease with distance from the site for the same reasons. The contribution from the groundwater pathway will be largest for drinking water obtained from a well at the boundary of the contaminated region on the downgradient side that draws water from the unconfined aquifer. This contribution can be the same for on-site and near-site residents but will decrease for wells at greater distances from the boundary. The situation is more complicated for food chain pathways because reconcentration can occur along these pathways. However, the predominant contribution is from on-site

Other scenarios can be taken into account by adjusting the scenario parameters in formulas for calculating transport of radionuclides through the pathways (see Chapters 3 and 4 and Appendixes B through E).

In the family-farm scenario, a family is assumed to move onto the site after it has been released for use without radiological restrictions, build a home, and raise crops and livestock for family consumption. Members of the family can incur a radiation dose by (1) direct radiation from radionuclides in the soil, (2) inhalation of resuspended dust (if the contaminated area is exposed at the ground surface), (3) ingestion of food from crops grown in the contaminated soil, (4) ingestion of milk from livestock raised in the contaminated area, (5) ingestion of meat from livestock raised in the contaminated area, (6) ingestion of fish from a nearby pond contaminated by water percolating through the contaminated zone, and (7) ingestion of water from a well contaminated by water percolating through the contaminated zone. The basis for the choice of a family-farm scenario for the critical population group is summarized below.

Permanent residents, rather than individuals exposed by activities not associated with residential living, have been chosen as the critical population group because the exposure for permanent residents is more likely to be long term and will generally involve exposure by more pathways. The nonresident group most likely to receive significant exposure consists of construction workers. An individual involved in recreational activities (e.g., a baseball field built on the site) will receive a much smaller dose than a permanent resident because the former will spend less time on-site. Scavenging can also occur, although this is less likely considering the lack of economic value of the contaminated material. The exposure of scavengers can reasonably be assumed to be comparable to that of construction workers.

crops and domestic animals, and this contribution will be greatest for on-site residents who raise food for their own consumption.

Exposure scenarios used for establishing soil guidelines should be bounding in the sense that they correspond to actions, events, and processes that will result in the largest exposure likely to occur to individuals and groups. However, they must also be credible, which implies that the probability of occurrence should be above some threshold value. The basis for specifying a credible bounding scenario is ill-defined because a threshold probability for distinguishing between a credible and a noncredible scenario has not been established, and it is usually not possible to assign a meaningful probability of occurrence for a scenario (unless the scenario is physically impossible, in which case a zero probability can be assigned). A family-farm scenario, in which a family constructs a home on the contaminated site and raises an appreciable fraction of its food on this site, is considered to be a credible bounding scenario for the purpose of this manual.* Even though such a scenario may be unlikely in the foreseeable future for a FUSRAP or SFMP site located in an industrial or urban area, it cannot be excluded as noncredible at some time several hundred years in the future.

The assignment of appropriate values to the scenario parameters is based on existing patterns of human activity that can be expected to persist for an indefinite time. For most scenario parameters, this criterion enables a straightforward determination of parameter values on the basis of data for

---

*This scenario is referred to as the intruder-agriculture scenario when used for analyzing the long-term radiological impacts of a low-level radioactive waste disposal site (NRC 1981). This terminology is inappropriate for FUSRAP and SFMP sites because a resident cannot be regarded as an intruder after a site has been released for use without radiological restrictions.

current conditions. Scenario parameters and parameter values required for the pathway analysis are given in Appendixes B through E. The basis for the choice of key parameter values that require special consideration are discussed below.

One scenario parameter is the fraction of the family diet that consists of contaminated foodstuffs. This will be determined by the fraction of the diet that is raised on-site and the fraction of the on-site crops grown in contaminated soil. The fraction raised on-site will be determined by the living and farming style and the area of land available to a family for gardening.

Farms or residential gardens that provide a significant fraction of the family diet are fairly common, and it is not uncommon for families in rural areas to raise livestock to provide most of their meat and milk; however, a fully self-sufficient farm is a rarity. For a "self-sufficient" scenario to apply, the family would have to raise all of its vegetables on-site -- that is, grow its own potatoes, get all fruit from an on-site fruit 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. A scenario that is considered to be bounding and credible is for a family to have a garden that provides half of the total plant food diet if the area available for gardens and orchards is 0.1 ha or larger. This value is based on the following estimates for the area required for a year's supply of food for an individual: 50 m^2 for a year's supply of leafy and other aboveground vegetables, 200 m^2 for root vegetables and grains, and a somewhat larger area for fruit trees. It is inferred from these figures that 0.1 ha would be sufficient for half of the plant food diet for a family of four. It is also assumed, as noted above, that half the diet would be food purchased from a market even if sufficient land were available for a larger garden.

The area required for livestock, including summer pasture and forage for winter use, is somewhat larger. A single pig requires a food supply from an area of about 2,000 m^2, and a single milk cow needs a forage supply that may require an area of 1 ha or more. It is assumed that an area of 2 ha would be needed to provide sufficient meat and milk, including all forage, for a family of four. Thus, if the contaminated area exceeds 2 ha, the scenario diet factors specify that all of the meat and milk and 50% of the plant foods in the family diet are potentially contaminated. The fraction of the meat and milk diet that is contaminated is assumed to decrease linearly from 1 to 0 as the area decreases from 2 to 0 ha. The fraction of the plant food diet that is contaminated is assumed to decrease linearly from 0.5 to 0 as the area decreases from 0.1 to 0 ha.

If the contaminated material is initially underneath a protective cover of uncontaminated material, one should take into account the dose that might be incurred from radioactive material that is redistributed during excavation for the basement of a residence to form an exposed surface layer. For the purpose of estimating the extent of the contaminated zone caused by redistribution of the excavated radioactive material, it may be assumed that the basement has an area of 200 m^2 and requires excavation to a depth of 3 m. It may also be assumed that the excavated soil is mixed before spreading and is redistributed on the surface to a depth of 0.3 m over an area of 2,000 m^2. Thus, if the contaminated layer was T meters thick and did not extend below 3 m, after excavation there would be a contaminated layer exposed at the surface with a radionuclide concentration that was less than the concentration in the excavated contaminated layer by a factor T/3. No credit for this reduction in concentration should be taken if the undisturbed contaminated layer is exposed at the surface or if the redistributed layer would result in a predicted potential dose that was less than the potential dose predicted for the undisturbed layer.

3 GUIDELINES FOR RADIONUCLIDE CONCENTRATIONS IN SOIL

The basic criterion for releasing a site for use without radiological restrictions is specified as a dose limit (see Appendix A). This dose limit is converted to soil guidelines -- specified as radionuclide concentrations -- by means of dose/source ratios (DSRs) that are expressed in terms of three primary factors: dose conversion factors (DCFs), environmental transport factors (ETFs), and source factors (SFs). The definitions and use of these factors for deriving soil guidelines are described in this chapter. Tables, detailed models, and formulas for calculating the factors are given in Appendixes B through E.

3.1 RADIOLOGICAL RELEASE CRITERIA

The basic dose limit that must be satisfied before a site can be released for use without radiological restrictions is

$$H_E(t) \le H_{EL}, \qquad t_r \le t \le t_h \tag{3.1}$$

where*

$H_E(t)$ = average annual effective dose equivalent received by a member of the critical population group at time t following the radiological survey of the site (mrem/yr),

---

*Units are specified in parentheses following the variable description. Numerical values are given for input variables to which default values are assigned. Default values are assigned to all input variables except radionuclide concentrations. Site-specific values should be substituted for default values unless it is not feasible to determine a site-specific value and the mixture sum is not sensitive to the value used.

H_{EL} = basic dose limit (100 mrem/yr),

t_r = time at which the site is released for use without radiological restrictions following the radiological survey (1 year), and

t_h = time horizon (1,000 years).

The time at which a radiological survey is done is the time of origin, or time 0. The time at which a site is released after the radiological survey is the release time, or time t_r. The default value used in the RESRAD code for t_r is 1 year. A time horizon of 1,000 years is used for FUSRAP and SFMP sites (see Appendix A).

If the radionuclides are uniformly distributed within the contaminated zone, Equation 3.1 may be transformed into the inequality

$$M(t) = \sum_i S_i(0)/G_i(t) \le 1, \qquad t_r \le t \le t_h \tag{3.2}$$

where

$M(t) \equiv H_E(t)/H_{EL}$ = fraction of the basic dose limit received by an average member of the critical population group at time t following the radiological survey (dimensionless),

$S_i(0)$ = initial concentration of the i^{th} principal radionuclide in a uniformly contaminated zone at time 0 (pCi/g), and

$G_i(t)$ = single-radionuclide soil concentration guideline for the i^{th} principal radionuclide in a uniformly contaminated zone at time t (pCi/g).

M(t) is referred to as the mixture sum.* Principal radionuclides are radionuclides with half-lives greater than 1 year. 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 1 year. It is assumed that a principal radionuclide is in secular equilibrium with its associated radionuclides at the point of exposure. Principal and associated radionuclides included in the current version of RESRAD are listed in Table 3.1.

A contaminated zone is defined as a belowground volume within which the radionuclide concentrations in soil samples clearly exceed the background concentrations. Background concentrations are determined from measurements in soil samples taken at several nearby off-site locations where contamination is very unlikely. The concentration of a radionuclide is considered to clearly exceed the background concentration if it is larger than the mean background concentration plus twice the standard deviation of the background measurements. If the concentrations in the samples used for determining the background concentration are below the lower limit of detection (LLD) of the instrument used, the concentration of that radionuclide is considered to exceed background if it exceeds the LLD of the instrument. The sensitivity of the instrument used must comply with current standards for high-quality commercial instruments.

A distinction should be made between a uniformly contaminated zone and a homogeneously contaminated zone. In a uniformly contaminated zone, radionuclide concentrations are exactly the same at every point. A uniformly

---

*The term "sum of fractions" is commonly used in regulatory references.

TABLE 3.1 Principal and Associated Radionuclides

Principal Radionuclide[a]			Terminal Nuclide or Radionuclide[c]	
Species	Half-life (yr)	Associated Decay Chain[b]	Species	Half-life (yr)
H-3	12.33	-	He-3	*
C-14	5730	-	N-14	*
Fe-55	2.685	-	Mn-55	*
Co-60	5.271	-	Ni-60	*
Ni-59	7.5×10^4	-	Co-59	*
Ni-63	100	-	Cu-63	*
Sr-90	28.8	Y-90 (64.1 h)	Zr-90	*
Nb-94	2.0×10^4	-	Mo-94	*
Tc-99	2.14×10^5	-	Ru-99	*
I-129	1.6×10^7	-	Xe-129	*
Cs-135	3×10^6	-	Ba-135	*
Cs-137	30.17	Ba-137m (2.551 min)	Ba-137	*
Sm-151	90	-	Eu-151	*
Eu-152	13	-	Gd-152 (27%) Sm-152 (73%)	1.1×10^{14} *
Eu-154	8.5	-	Gd-154	*
Pb-210	22.3	Bi-210 (5.01 d) Po-210 (138.38 d)	Pb-206	*
Ra-226	1.60×10^3	Rn-222 (3.8235 d) Po-218 (3.05 min) Pb-214 (26.8 min) Bi-214 (19.7 min) Po-214 (164 μs)	Pb-210	22.3
Ra-228	5.76	Ac-228 (6.13 h)	Th-228	1.9131
Ac-227	21.733	[Th-227 (98.6%, 18.71 d) Fr-223 (1.4%, 21.8 min)] Ra-223 (11.435 d) Rn-219 (3.96 s) Po-215 (1.78 ms) Pb-211 (36.1 min) Bi-211 (2.15 min) Tl-207 (4.77 min)	Pb-207	*
Th-228	1.9131	Ra-224 (3.66 d) Rn-220 (55.6 s) Po-216 (0.15 s) Pb-212 (10.64 h) Bi-212 (60.60 min) [Po-212 (64%, 0.30 μs) Tl-208 (36%, 3.053 min)]	Pb-208	*

TABLE 3.1 (Cont'd)

Principal Radionuclide[a]			Terminal Nuclide or Radionuclide[c]	
Species	Half-life (yr)	Associated Decay Chain[b]	Species	Half-life (yr)
Th-229	7.3×10^3	Ra-225 (14.8 d) Ac-225 (10.0 d) Fr-221 (4.8 min) At-217 (32.3 ms) Bi-213 (45.6 min) [Po-213 (97.8%, 4 μs) Tl-209 (2.2%, 2.2 min)] Pb-209 (3.25 h)	Bi-209	*
Th-230	7.7×10^4	-	Ra-226	1.60×10^3
Th-232	1.41×10^{10}	-	Ra-228	5.76
Pa-231	3.28×10^4	-	Ac-227	21.6
U-232	72	-	Th-228	1.9131
U-233	1.592×10^5	-	Th-229	7.3×10^3
U-234	2.45×10^5	-	Th-230	8.0×10^4
U-235	7.038×10^8	Th-231 (25.52 h)	Pa-231	3.28×10^4
U-236	2.342×10^6	-	Th-232	1.41×10^{10}
U-238	4.468×10^9	Th-234 (24.10 d) Pa-234m (1.175 min)	U-234	2.45×10^5
Np-237	2.14×10^6	Pa-233 (27.0 d)	U-233	1.592×10^5
Pu-238	87.74	-	U-234	2.45×10^5
Pu-239	2.41×10^4	-	U-235	7.038×10^8
Pu-240	6.57×10^3	-	U-236	2.342×10^6
Pu-241	14.4	-	Am-241	433
Pu-242	3.76×10^5	-	U-238	4.468×10^9
Am-241	433	-	Np-237	2.14×10^6
Am-243	7.37×10^3	Np-239 (2.35 d)	Pu-239	2.41×10^4
Cm-243	28.5	-	Pu-239	2.41×10^4
Cm-244	18.11	-	Pu-240	6.57×10^3

---

[a] Radionuclides with half-lives greater than 1 year that might be present at FUSRAP or SFMP sites.

[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. Stable nuclides are indicated by an asterisk (*) in place of the half-life.

contaminated zone is a mathematical construct used to calculate soil guidelines. A homogeneously contaminated zone is a volume within which deviations from uniformity of the actual radionuclide concentrations are considered small enough to allow the volume to be treated as if it were uniformly contaminated.

Actual radionuclide distributions are nonuniform. The potential annual individual dose received through a particular pathway is an areal average of the nonuniform residual radioactivity over an area determined by the scenario activities, for example, the area of daily activities for external radiation or the size of the garden for plant food pathways. For the purpose of deriving soil guidelines, it is assumed that this area is 100 m^2 for all pathways. The effect of vertical nonuniformities is taken into account by averaging over the radionuclide concentrations in a 0.15-m-thick layer. Thus, the criterion for releasing a site for use without radiological restrictions is as follows: for any 100-m^2 area and 0.15-m-thick layer within the contaminated zone,

$$\bar{M}(t) \equiv \sum_{i} \bar{S}_i(0)/G_i(t) \le 1, \qquad t_r \le t \le t_h \tag{3.3}$$

where

$\bar{M}(t)$ = average mixture sum at time t (dimensionless),

$\bar{S}_i(0)$ = initial concentration of the i^{th} principal radionuclide averaged over a 100-m^2 area and 0.15-m-thick layer (pCi/g), and

$G_i(t)$ = single-radionuclide soil concentration guideline for the i^{th} principal radionuclide in a uniformly contaminated zone at time t (pCi/g).

Equation 3.3 is the homogeneous release criterion. It is directly applicable to homogeneous contamination, defined as contamination in a volume within which radionuclide concentrations in individual soil samples do not exceed the average concentrations by a factor larger than three. For a contaminated site, if the number and distribution of soil samples required by characterization and certification protocols result in only one soil sample from a contaminated region that is 100 m^2 in area and 0.15 m thick, then the contamination within the volume is assumed to be homogeneous and the average radionuclide concentrations are assumed to equal the concentrations in that sample. If no soil samples are taken from the volume, the radionuclide concentrations are assumed to be averages of concentrations in soil samples from the nearest neighboring volumes from which samples are taken. (See Ahrends [1987], DOE [1984], and ORNL [1982] for sampling protocols.)

For field applications, the homogeneous criterion as specified by Equation 3.3 may be replaced by the criterion

$$\bar{M} = \sum_i \bar{S}_i(0)/G_i(t_m) \le 1 \qquad (3.4)$$

where

$\bar{S}_i(0)$ = as defined for Equation 3.3, and

$G_i(t_m)$ = single-radionuclide soil guideline for the i^{th} principal radionuclide in a homogeneous contaminated zone at the time $t = t_m$ within the interval $t_r \le t \le t_h$ that the time-dependent value $G_i(t)$ is minimum (pCi/g).

$G_i(t_m)$ should be obtained from RESRAD prior to the field actions. Equations 3.3 and 3.4 are equivalent when the minima in $G_i(t)$ for different principal radionuclides all occur at the same time. Equation 3.4 is more restrictive than Equation 3.3 because the minima in $G_i(t)$ usually occur at different times for different principal radionuclides.

For inhomogeneous contamination, defined as contamination in a volume within which the above-background concentration of a radionuclide at one or more sampling locations exceeds three times the average concentration, the average concentration $\bar{S}_i(0)$ in Equation 3.3 may be conservatively replaced by $\hat{S}_i(0)/3$, where $\hat{S}_i(0)$ is the maximum concentration in a soil sample taken from within the area.

Alternatively, when sufficient data are available and conditions warrant, one may apply less conservative, inhomogeneous contamination criteria in place of homogeneous contamination criteria. The purpose of the inhomogeneous contamination criteria is to provide a more realistic guideline for cleanups in cases where the conservatism inherent in the homogeneous assumptions would cause excessive expenditures in comparison to the benefits associated with the more conservative approach. The inhomogeneous contamination criteria, which are discussed in Section 3.3.1, should not be confused with the field criteria or hot spot criteria, which are described in Section 3.3.2.

The purpose of the hot spot criteria is to ensure that applying the homogeneous criteria, in which the concentrations of residual radioactive material are averaged over a 100-m^2 area, does not result in the release of small areas that, due to averaging, contain unacceptably high concentrations of residual radioactive material. Although the inhomogeneous criteria may, under appropriate conditions, be used in place of the general guidelines, the hot spot criteria must be used along with the general guidelines or authorized limits resulting from the homogeneous criteria.

A single-radionuclide soil concentration guideline for a uniformly contaminated zone is defined as

$$G_i(t) \equiv H_{EL}/DSR_i(t) \tag{3.5}$$

where

H_{EL} = basic dose limit (100 mrem/yr),

$DSR_i(t) = \sum_p DSR_{ip}(t)$ = dose/soil-concentration ratio for the ith principal radionuclide in the contaminated zone at time t [(mrem/yr)/(pCi/g)], and

$DSR_{ip}(t)$ = dose/soil-concentration ratio for the ith principal radionuclide and pth environmental pathway [(mrem/yr)/(pCi/g)].

The dose/soil-concentration ratios for individual principal radionuclides and pathways are defined as

$$DSR_{ip}(t) \equiv H_{E,ip}(t)/S_i(0) \tag{3.6}$$

where

$H_{E,ip}(t)$ = average annual effective dose equivalent received at time t by a member of the critical population group from the ith principal radionuclide transported through the pth environmental pathway together with its associated decay products (mrem/yr), and

$S_i(0)$ = initial concentration of the ith principal radionuclide in a uniformly contaminated zone (pCi/g).

The total annual effective dose equivalent to an individual is

$$H_E(t) = \sum_i \sum_p H_{E,ip}(t) \tag{3.7}$$

Substituting Equations 3.5–3.7 into Equation 3.1 and rearranging terms leads to Equation 3.2.

3.2 DOSE/SOURCE CONCENTRATION RATIOS FOR UNIFORM CONTAMINATION

The dose/source (D/S) concentration ratios are calculated by first expressing them as products of dose conversion factors [dose/exposure-parameter (D/E) ratios], environmental transport factors [exposure-parameter/source-concentration (E/S) ratios], and source factors (ratios of the concentration of a radionuclide at a given time to the initial concentration). The factored expression for a D/S ratio is

$$DSR_{ip}(t) = \sum_j DCF_{j,x(p)} \times ETF_{jp}(t) \times SF_{ij}(t) \tag{3.8}$$

where

$DCF_{j,x(p)}$ = dose conversion factor for the j^{th} principal radionuclide and $x(p)^{th}$ exposure pathway [(mrem/yr)/(pCi/cm^3) for external radiation from the contaminated zone; mrem/pCi for internal radiation from ingestion or inhalation of radionuclides],

$ETF_{jp}(t)$ = environmental transport factor for the j^{th} principal radionuclide and p^{th} environmental pathway at time t (g/cm^3 for external radiation from the contaminated zone; g/yr for internal radiation from ingestion or inhalation of radionuclides),

$SF_{ij}(t)$ = factor for ingrowth and decay and leaching of the j^{th} principal radionuclide at time t from the i^{th} principal radionuclide present initially (dimensionless),

$x(p)$ = index label for exposure pathways, which is a function of the environmental pathway p [for p = 1 (external radiation from the ground), x(p) = 1; for p = 2 (inhalation), x(p) = 2; and for p = 3, 4, 5, 6, and 7 (ingestion of plants, meat, milk, fish, and water, respectively), x(p) = 3],

p = index label for environmental pathways, and

i,j = index labels for principal radionuclides.

3.2.1 Dose Conversion Factors

A dose conversion factor is the ratio

$$DCF_{ix} = H_{E,ix}/\dot{E}_{ix} \tag{3.9}$$

where

$H_{E,ix}$ = annual effective dose equivalent resulting from exposure to external radiation (x = 1) from the i^{th} principal radionuclide and its associated radionuclides (mrem/yr) or the annual committed effective dose equivalent resulting from exposure for 50 years to internal radiation from the amount of the i^{th}

principal radionuclide and its associated radionuclides inhaled ($x = 2$) or ingested ($x = 3$) in 1 year (mrem/pCi) and

E_{ix} = exposure parameter for the i^{th} principal radionuclide [concentration of the i^{th} principal radionuclide in a standard source (for external radiation pathways) or the annual quantity of the i^{th} principal radionuclide inhaled or ingested (for internal radiation pathways)] (pCi/cm^3 for external radiation [$x = 1$] from the contaminated zone; pCi/yr for internal radiation from inhalation [$x = 2$] or ingestion [$x = 3$]).

An internal dose conversion factor for any radionuclide includes the contribution from ingrowth following ingestion or inhalation. The internal dose conversion factor for a principal radionuclide includes, in addition, the contribution from inhalation or ingestion of associated radionuclides along with the principal radionuclide. The additional contribution is significant only for associated radionuclides with half-lives that are not small compared to the biological half-life. The dose conversion factor for external radiation for a principal radionuclide is defined here to include the dose from all of its associated radionuclides, which are assumed to be in secular equilibrium.

Dose conversion factors for external radiation from the contaminated zone, internal radiation from inhalation, and internal radiation from ingestion are given in Tables B.1, C.1, and D.1, respectively, of Appendixes B, C, and D.

3.2.2 Environmental Transport Factors

An environmental transport factor is the time-dependent ratio

$$ETF_{ip}(t) = E_{ip}(t)/S_i(t) \tag{3.10}$$

where

$E_{ip}(t)$ = exposure parameter value at time t for the ith principal radionuclide (or radiation therefrom) transported through the pth environmental pathway (pCi/cm^3 for external radiation from the contaminated zone; pCi/yr for internal radiation),

p = index label for environmental pathways, and

$S_i(t)$ = average concentration of the ith principal radionuclide in a uniformly contaminated zone at time t (pCi/g).

The exposure parameter for external radiation from the contaminated zone is the concentration of the ith principal radionuclide in the ground, adjusted for occupancy and the size and depth of the contaminated zone by means of multiplying factors. The exposure parameter for internal radiation pathways is the annual quantity of the ith principal radionuclide that is inhaled or ingested after migrating through the pth environmental pathway. There are only two internal exposure pathways (inhalation and ingestion); several environmental pathways can contribute to each.

Models and formulas for calculating environmental transport factors are given in Appendixes B through E.

3.2.3 Source Factors

A source factor is the time-dependent ratio

$$SF_{ij}(t) = S_{ij}(t)/S_i(0) \tag{3.11}$$

where

$S_{ij}(t)$ = concentration at time t of the j^{th} principal radionuclide remaining in the contaminated zone after leaching and ingrowth from the i^{th} principal radionuclide, if $j \neq i$ (pCi/g); or the concentration at time t of the i^{th} principal radionuclide remaining in the contaminated zone after leaching, excluding contributions from ingrowth from other radionuclides, if $j = i$ (pCi/g), and

$S_i(0)$ = initial concentration of the i^{th} principal radionuclide in the contaminated zone (pCi/g).

Formulas for calculating source factors are given in Appendix F.

3.3 GUIDELINES FOR INHOMOGENEOUS CONTAMINATION

Two separate formulations of inhomogeneous release criteria are developed: (1) the inhomogeneous criteria, intended for use during project planning and review as an alternative to the homogeneous criteria, and (2) the hot spot criteria, intended for field use in conjunction with the homogeneous criteria or for use when determination of compliance must be made immediately.

3.3.1 Inhomogeneous Criteria for Project Planning and Review

A contaminated zone is inhomogeneous if it contains a contaminated region within which the concentration of a radionuclide exceeds three times the average for the contaminated zone. Inhomogeneous release criteria, described below, are generally more realistic and hence less restrictive than the homogeneous release criteria. In general, inhomogeneous contamination should be remediated in order to meet homogeneous contamination limits. However, the inhomogeneous contamination criteria may be used in cases where it is inappropriate to use the homogeneous contamination criteria. For example, homogeneous criteria might be inappropriate if the contamination exists under a roadway, railroad right-of-way, or building. Inhomogeneous criteria can also be used if an area of elevated contamination is found during the verification survey after equipment and crews have left the site. An area of elevated contamination must always be remediated before a site can be released for use without radiological restrictions if that area does not comply with inhomogeneous criteria.

The mixture-sum release criterion for any zone of inhomogeneous contamination is

$$M^*(t) + \bar{M}(t) \leq 1, \qquad t_r \leq t \leq t_h \tag{3.12}$$

where

$M^*(t)$ = mixture sum for area of elevated contamination (dimensionless),
$\bar{M}(t)$ = mixture sum for averaging zone (dimensionless), and
t_r and t_h are as defined for Equation 3.1.

The averaging zone for inhomogeneous contamination is an area of 100 m^2 that encloses the elevated contamination.

The two mixture sums are defined as follows:

$$M^*(t) = \sum_i [S_i^*(0) - \bar{S}_i(0)]/G_i^*(t) \qquad (3.13)$$

$$\bar{M}(t) = \sum_i \bar{S}_i(0)/G_i(t) \qquad (3.14)$$

where

$S_i^*(0)$ = initial concentration of the i^{th} principal radionuclide in the area of elevated contamination (pCi/g),

$G_i^*(t)$ = single-radionuclide, time-dependent soil guideline for the i^{th} principal radionuclide in the area of elevated contamination (pCi/g),

$\bar{S}_i(0)$ = average initial concentration of the i^{th} principal radionuclide in the averaging zone over a 100-m^2 area and a 0.15-m-thick layer, excluding the area of elevated contamination (pCi/g), and

$G_i(t)$ = single-radionuclide, time-dependent soil guideline for the i^{th} principal radionuclide in a uniformly contaminated zone (pCi/g).

If the area of elevated contamination is inhomogeneous, one of two approaches may be used: (1) the area may be treated as if it were homogeneous with a concentration $\hat{S}_i(0)/3$, where $\hat{S}_i(0)$ is the peak concentration of the i^{th} principal radionuclide in a sample from within the area of elevated contamination, or (2) the area may be divided into subzones using the same procedure that is used for a zone of inhomogeneous contamination.

When inhomogeneous release criteria are used, Equation 3.12 must be satisfied for every area of inhomogeneous contamination and, in addition, Equation 3.3 must be satisfied for any region within the homogeneous portion of the contaminated zone.

3.3.2 Hot Spot Criteria for Field Application

Hot spots are small areas that have levels of residual radioactive material that are considerably above the levels in the surrounding area. The derivation of remedial action criteria generally assumes homogeneous contamination of large areas (several hundred square meters or more), and the derived concentration guide is stated in terms of concentrations averaged over an area of 100 m^2. Because of this averaging process, within these 100-m^2 areas hot spots can exist that contain concentrations of radionuclides that are significantly higher than the authorized limit. Therefore, the presence of hot spots could potentially pose a greater risk of exposure to individuals using the site than the risk associated with homogeneous contamination. In order to ensure that individuals are adequately protected and to ensure that the ALARA process is satisfied, the following hot spot criteria must be applied along with the general criterion for homogeneous contamination. The hot spot criterion for field application is

$$M^{**} \equiv \sum_i S_i^{*}/G_i^{**} \leq 1 \qquad (3.15)$$

where

M^{**} = hot spot mixture sum for field use (dimensionless),

S_i^{*} = measured concentration of the i^{th} principal radionuclide in the hot spot (pCi/g), and

G_i^{**} = single-radionuclide soil guideline for the i^{th} principal radionuclide in the hot spot (pCi/g).

The measured hot spot concentrations S_i^* are the peak concentrations if the hot spot area is 1 m^2 or less or the average concentrations if the hot spot area is larger than 1 m^2.

The formula for single-radionuclide, hot spot soil guidelines is

$$G_i^{**} = G_i(t_m) \times (100/A)^{1/2} \tag{3.16}$$

where

$G_i(t_m)$ = as defined for Equation 3.4,
A = area of hot spot (m^2), and
$(100/A)^{1/2}$ = hot spot multiplication factor.

Equations 3.15 and 3.16 apply to hot spots with areas of 25 m^2 or less. For larger hot spot areas, the homogeneous release criterion is sufficient. An area of $A = 1$ m^2 is used in Equation 3.15 if the actual hot spot area is less than 1 m^2. The average radionuclide concentrations for any 100-m^2 area must always comply with the homogeneous release criterion, irrespective of hot spot criteria.

For general field applications, it is recommended that the ranges of hot spot multiplication factors provided in Table 3.2 be used. The hot spot guideline for radionuclide i is calculated for each specific site by Equation 3.16. The term $G_i(t_m)$ in Equation 3.16 can be substituted by G_i, the authorized limit at a specific site for the i^{th} principal radionuclide.

TABLE 3.2 Ranges for Hot Spot Multiplication Factors

Range	Factor (multiple of authorized limit)
<1 m^2	10[a]
$1 - <3$ m^2	6
$3 - <10$ m^2	3
$10 - 25$ m^2	2

[a] Areas less than 1 m^2 are to be averaged over a 1 m^2 area, and that average shall not exceed 10 times the authorized limit.

The authorized limit is considered adequate to protect the public for areas larger than 25 m^2; hence, no special hot spot limits are required for areas larger than 25 m^2. Averaging of hot spots less than or equal to 25 m^2 shall be done only over the local hot spot area.

Every reasonable effort shall be made to identify and remove any source that has a radionuclide concentration exceeding 30 times the authorized limit, irrespective of area.

4 USER'S GUIDE FOR RESRAD

The analytical method for deriving guidelines for allowable concentrations of residual radioactive material in soil is described in Chapters 2 and 3 and Appendixes B through E. This method has been implemented in a computer code named RESRAD. RESRAD is designed for use on an IBM or IBM-compatible personal computer, with internal help files for information on input and output data. Instructions for installing and using the code are presented in Sections 4.1 through 4.8.

4.1 INSTALLING

An IBM (or IBM-compatible) microcomputer with a hard disk drive and approximately 400K of memory is required. A DOS 3.1 or equivalent operating system is needed; a mathematics coprocessor is optional but highly recommended. A printer capable of 132-column output is desirable. Most PC printers have a condensed print option (17 characters per inch) that is satisfactory. A color monitor and a mouse are optional but recommended. However, a mouse may not work with all monitor and graphics cards. No problems have occurred with Microsoft Bus Mouse, Logitech Bus Mouse, or Mouse Systems PS/2 Mouse. Any hardware problems encountered should be reported to the contact address on the last RESRAD screen.

The code is available either on two 360K 5.25-inch diskettes or on a single 720K 3.5-inch diskette. One 5.25-inch diskette is marked "RESRAD SOURCE CODES." This diskette contains the source code for all RESRAD programs and other supplementary files; it is not necessary for the operation of RESRAD and need not be copied to the hard disk. On the 3.5-inch diskettes, a \SOURCE subdirectory contains the identical material. The steps for installing the code are as follows:

1. Assuming the hard disk is the C disk, move to the C disk drive with the DOS commands

```
C:
CD\
```

2. Issue the DOS command

```
PROMPT $P$G
```

   This adds the current subdirectory to the DOS prompt.

3. Create a subdirectory called RESRAD on the hard disk (if an earlier version of RESRAD exists on the hard disk, this version will be replaced by the new version) and move to the new subdirectory using the commands

```
MD\RESRAD
CD\RESRAD
```

   The prompt "C:\RESRAD>" should appear on the monitor.

4. Insert either the 3.5-inch RESRAD distribution diskette or the 5.25-inch diskette labeled "RESRAD SYSTEM DISK" into the A disk drive and issue the command

```
COPY A:*.*
```

   to copy all necessary files to the hard disk.

5. The RESRAD system is stored in a special, compacted format. To complete the installation, enter the command

```
INSTALL
```

6. At this point, all of the executable modules, forms, data, and files needed to run RESRAD will be on the hard disk. However, successful execution of the code will require a CONFIG.SYS file in the root directory of the hard disk that contains at least the following:

```
FILES=20
BUFFERS=16
SHELL=C:\COMMAND.COM /P/E:256
```

A CONFIG.SYS consisting of the above lines will be found in the \RESRAD directory. The existing CONFIG.SYS may be replaced by typing

```
COPY C:\RESRAD\CONFIG.SYS C:\
```

or an existing CONFIG.SYS may be modified with a suitable editor such as EDLIN. Note that the 256 in the SHELL command sets the size of the DOS environment-string table. If an out-of-environment-space error occurs, increase the size of the E parameter. A system reboot (Ctrl-Alt-Del) is needed whenever changes are made to the CONFIG.SYS file.

4.2 RESRAD EXECUTION

The prompts (in boldface) and responses for starting RESRAD are

```
C:\> CD\RESRAD
C:\RESRAD> RESRAD [ARGUMENT 1] [ARGUMENT 2] [ARGUMENT 3]
```

Up to three optional command line arguments may be included. The same information can be supplied (or superseded) later via regular RESRAD form input;

however, command line input, if different from the system defaults, is more efficient. The optional arguments are as follows.

ARGUMENT 1: The name of the initial (template) site-specific data file. The default file for this purpose is RESRAD.DEF. It is supplied on the distribution diskette and contains the default values for all RESRAD parameters. It also serves as a sample test problem. In general, users are advised to specify an initial data file that more closely matches the specific site.

ARGUMENT 2: The name of the final site-specific data file. This is the file that actually serves as input data for RESRAD. In the menu phase of RESRAD (see Section 4.4), any of the parameters supplied by the initial site-specific data file (ARGUMENT 1) may be modified. All changes are saved in the final site-specific data file (ARGUMENT 2). If there are no command line arguments, the default for ARGUMENT 2 is SAMPLE.DAT. If only one argument is supplied, ARGUMENT 2 = ARGUMENT 1 is the default (except if RESRAD.DEF is used, in which case the default for ARGUMENT 2 again is SAMPLE.DAT). *Warning: if the initial site-specific data file is to serve as a template for additional runs, a different name must be supplied for the final site-specific data file.*

ARGUMENT 3: The destination of RESRAD hard copy output. The default is LPT1 for normal printer output. Or the user may specify a disk file name to save all or selected portions of RESRAD output. In the case of an existing disk file, the user can select to either overwrite the file or append to the end of the file.

All of the above arguments may include disk drive and path names up to a maximum of 36 characters each. As a special feature, the DOS wildcard characters "*" or "?" may be included in the file name(s). RESRAD will produce an appropriate directory listing(s) before starting normal output, and the user can then input the desired file name or try another wildcard directory listing. An example using all three possible arguments is

```
C:\RESRAD> RESRAD MYSITE.DAT MYSITE1.DAT A:MYSITE1.PRN
```

In the above example, MYSITE.DAT must exist in the default \RESRAD directory, MYSITE1.DAT will be produced (or overwritten) in the \RESRAD directory, and all hard copy output will be "printed" in MYSITE1.PRN on the A drive.

One final feature is available for starting RESRAD. The standard IBM-PC font, which uses ASCII characters 128-255, is assumed to be available on the printer. If strange characters appear on the printed output, use RESRAD1 instead of RESRAD to start the code, i.e.,

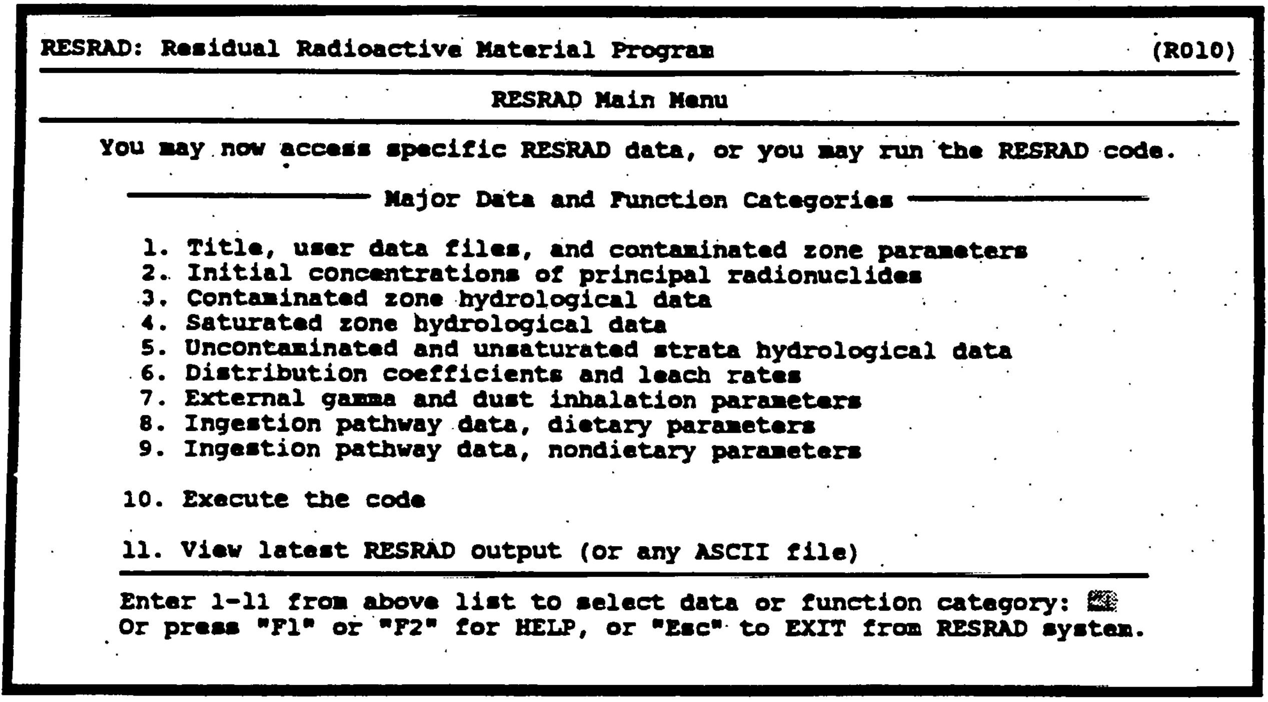
```
C:\RESRAD> RESRAD1 [ARGUMENT 1] [ARGUMENT 2] [ARGUMENT 3]
```

4.3 PRELIMINARY OUTPUT

The initial RESRAD output is a banner page with a brief code definition. This is followed by a printer setup series that allows the user, if desired, to set the printer for output of 132 columns per page. The final preliminary output will occur only if a mouse is available on the PC. In this case, the user is provided with RESRAD-specific instructions on use of the mouse. Although optional, a mouse provides three unique RESRAD advantages: direct access to any menu item, instant review of a parameter's default value, and automatic restoration of a parameter's default value.

4.4 MAIN MENU AND HELP FILES

The true starting point for using RESRAD is the main menu (R010), reproduced in Figure 4.1. This appears as soon as the preliminary output (Section 4.3) is completed. The main menu is also the return point after completion of RESRAD's calculation phase. By selecting the major data and function categories, all input data and output reports are accessible from the main menu. To terminate RESRAD, press "Esc" while the main menu is on screen.



```
RESRAD: Residual Radioactive Material Program                        (R010)
                           RESRAD Main Menu
   You may now access specific RESRAD data, or you may run the RESRAD code.
        ────────────── Major Data and Function Categories ──────────────
          1. Title, user data files, and contaminated zone parameters
          2. Initial concentrations of principal radionuclides
          3. Contaminated zone hydrological data
          4. Saturated zone hydrological data
          5. Uncontaminated and unsaturated strata hydrological data
          6. Distribution coefficients and leach rates
          7. External gamma and dust inhalation parameters
          8. Ingestion pathway data, dietary parameters
          9. Ingestion pathway data, nondietary parameters

         10. Execute the code

         11. View latest RESRAD output (or any ASCII file)
        Enter 1-11 from above list to select data or function category:
        Or press "F1" or "F2" for HELP, or "Esc" to EXIT from RESRAD system.
```

FIGURE 4.1 Main Menu for RESRAD

Instructions for interactive use of the code are provided in help files that can be displayed on the screen by pressing the "F1" or "F2" function key. The "F1" help screen contains information for positioning the cursor, scrolling displays, and restoring, deleting, or entering data. This help screen is the same for all RESRAD input forms. Descriptions of the input and output data are given in the "F2" help files. In addition, the last line of every form is a help message for the current active field (cursor position).

Menu items are selected in the main menu (R010) by entering the appropriate item number and pressing "Enter." Items 1-9 display input forms on the screen, item 10 calculates results for the current set of parameters, and item 11 displays the output (summary or detailed report) from the last RESRAD run or the user may view any selected ASCII file.

Input data needed by RESRAD are accessible in a series of input forms. Default input values are provided for all input data except radionuclide concentrations. After an input form has been called for display, its input data can be reviewed and changed from the terminal. Pressing the "F10" function key will record all displayed entries and return control to the main menu. The changes made after an input form has been modified (but before it has been recorded) can be canceled by pressing the escape ("Esc") key. Recorded data are retained until changed again.

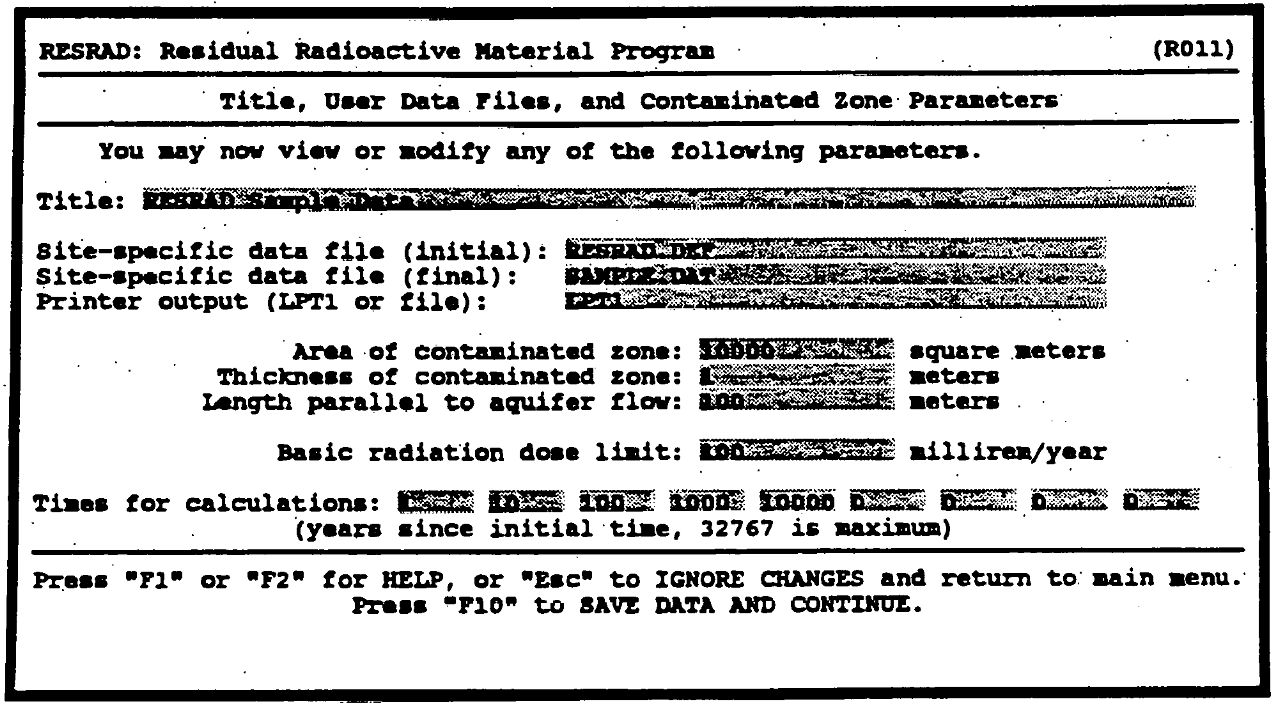
When RESRAD is executed, output from the preceding run is replaced by output from the current run. Thus, output from each run must be reviewed and recorded before executing the next run. However, all input data and all or selected portions of the output can be saved in user-specified disk files. A saved data file can then serve as the default input for a new run, after review and modifications via the input forms.

The output is divided into two parts: (1) the summary report, (SUMMARY.REP), for determining compliance with soil guidelines and examining the relative contributions to the mixture sum from different pathways at different times, and (2) the detailed report, (DETAILED.REP), for examining the effect of different parameters and tracking down the cause of anomalies. Output forms can be viewed page by page and can be printed (or saved) in total or in consecutive page groups.

4.5 INPUT FORMS

4.5.1 Title, User Data Files, and Contaminated Zone Parameters

Input form R011 is used to access the title, user-specified data files, and contaminated zone parameters. Form R011 with default data is shown in Figure 4.2.



```
RESRAD: Residual Radioactive Material Program                              (R011)
                Title, User Data Files, and Contaminated Zone Parameters

   You may now view or modify any of the following parameters.

Title: RESRAD Sample Data

Site-specific data file (initial): RESRAD.DEF
Site-specific data file (final):   SAMPLE.DAT
Printer output (LPT1 or file):     LPT1

                 Area of contaminated zone: 10000      square meters
            Thickness of contaminated zone: 1          meters
           Length parallel to aquifer flow: 100        meters

                Basic radiation dose limit: 100        millirem/year

Times for calculations: 1     10    100   1000  10000 0     0     0     0
                   (years since initial time, 32767 is maximum)

Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                     Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.2 Input Form R011 for Title, User Data Files, and Contaminated Zone Parameters

The TITLE used to identify the run can be up to 70 alphabetic or numerical characters. It is part of the header for every page of RESRAD output.

The SITE-SPECIFIC DATA FILE (INITIAL) is the name of the data file used for initial site-specific RESRAD input. The default file for this purpose is RESRAD.DEF, as supplied with the system diskette. However, the user will probably have a data file closer to the current need. If so, the file name can be specified here, and the new data will be input as soon as "F10" is pressed. Disk drive and path can be included in the file name (36-character maximum).

If the file name contains the wildcard character * or ?, RESRAD will supply a directory listing instead of attempting to read the file. Thus, A:*.DAT would list all files on the A disk with an extension of DAT. The user can then input the desired file or try another wildcard directory listing.

The SITE-SPECIFIC DATA FILE (FINAL) is the name of a file that contains the actual data used for a run. This "parameter" allows the user an easy method for saving the data used for each RESRAD calculation. The file name, again a 36-character maximum that can include drive and path, is included in the header of each page of RESRAD output. The final data file is updated every time a form is exited by pressing the "F10" function key.

The PRINTER OUTPUT (LPT1 OR FILE) is the destination of RESRAD hard copy output, which is obtained using option 11 from the main menu. The default is LPT1 for normal printer output. Or the user may specify a disk file name, again a 36-character maximum, for the "printer" output. If an existing file is specified, the new output can either overwrite or be appended to the end of the file. The user may also include the wildcard * or ? in the file name, which will produce a directory listing when "F10" is pressed.

The site-specific data file names and the printer output file can also be specified as arguments on the RESRAD command line (see Section 4.2). This is actually the preferred method for the initial data file because otherwise a new data set must be read in. The printer output file can also be changed at any time via main menu option 11 and "F7" (see Section 4.7.3).

After each RESRAD execution (main menu option 10) and return to the main menu, the initial site-specific data file is replaced with the final site-specific data file of the just-completed run. The final site-specific data file and printer output file remain unchanged. This allows the parameter input of the previous run to serve as the template for the next execution. The user, of course, may alter these automatic selections at any time.

The AREA OF CONTAMINATED ZONE is a compact area, specified in square meters (m^2), that contains the locations of all soil samples with radionuclide concentrations that are clearly above background and are separated from the locations of other above-background soil samples by a distance of at least 100 m. The concentration of a radionuclide is clearly above background if it exceeds the background level measurements by at least two standard deviations (see Section 3.1). Credible evidence that the intervening distance between any two contaminated zones is uncontaminated, based on radiological survey data, must be provided in order to justify the use of two or more contaminated zones; otherwise the contaminated zone should be characterized by a single compact area that contains the locations of all soil samples with radionuclide concentrations that are above background.

The THICKNESS OF CONTAMINATED ZONE is the distance, in meters (m), between the uppermost and lowermost soil samples with radionuclide concentrations that are clearly above background.

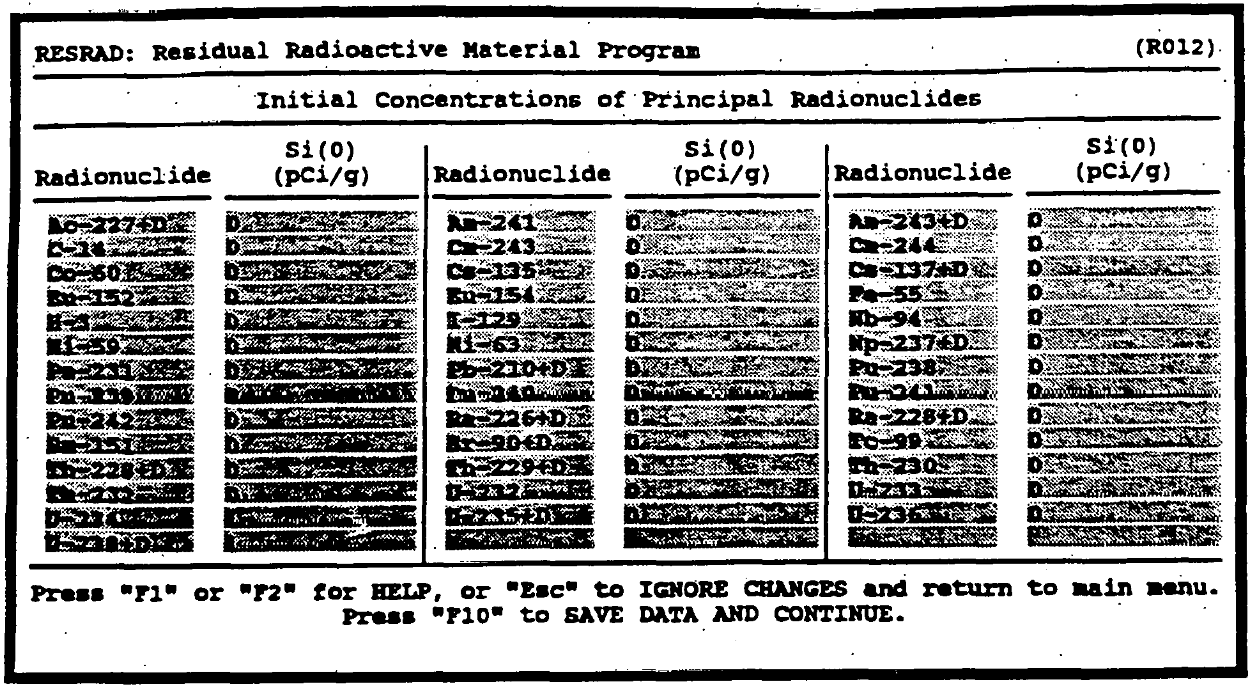
The LENGTH PARALLEL TO AQUIFER FLOW is the distance, in meters (m), between two parallel lines perpendicular to the direction of aquifer flow, one at the upgradient edge of the contaminated zone and the other at the downgradient edge.

The BASIC RADIATION DOSE LIMIT is the annual radiation dose limit, in millirems per year (mrem/yr), used to derive all site-specific soil guidelines.

The TIMES FOR CALCULATIONS are the times, in years, following the radiological survey for which the single radionuclide soil guidelines and mixture sums will be calculated. The default values are 1, 10, 100, 1,000, and 10,000 years. The guidelines require that the mixture sum be less than one at all times out to the time horizon; hence, exploratory calculations at other times should be carried out to ensure that no mixture sum maxima lie between the default times. A time horizon of 1,000 years is used for FUSRAP (see Appendix A); however, calculations can be carried out to greater periods to identify any delayed contributions from the groundwater or other pathways. RESRAD always calculates values for year 0. All years must be input in ascending order. If fewer than 10 time periods are selected, a zero must be input for the time period immediately following the last one desired.

4.5.2 Initial Concentrations of Principal Radionuclides

Input form R012, shown in Figure 4.3, is used to access the initial concentrations of principal radionuclides. The input fields on this form are broken into three parts, each with two columns. All radionuclides for which guidelines can be derived with the current version of RESRAD are listed in the first column of each of the three parts. Soil guidelines are calculated only for those radionuclides for which nonzero concentrations are entered.



RESRAD: Residual Radioactive Material Program (R012)

Initial Concentrations of Principal Radionuclides

Radionuclide	Si(0) (pCi/g)	Radionuclide	Si(0) (pCi/g)	Radionuclide	Si(0) (pCi/g)
Ac-227+D	0	Am-241	0	Am-243+D	0
C-14	0	Cm-243	0	Cm-244	0
Co-60	0	Cs-135	0	Cs-137+D	0
Eu-152	0	Eu-154	0	Fe-55	0
H-3	0	I-129	0	Nb-94	0
Ni-59	0	Ni-63	0	Np-237+D	0
Pa-231	0	Pb-210+D	0	Pu-238	0
Pu-239	[illegible]	Pu-240	0	Pu-241	0
Pu-242	0	Ra-226+D	0	Ra-228+D	0
Sm-151	0	Sr-90+D	0	Tc-99	0
Th-228+D	0	Th-229+D	0	Th-230	0
Th-232	0	U-232	0	U-233	0
U-234	[illegible]	U-235+D	0	U-236	0
U-238+D	[illegible]				

Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
Press "F10" to SAVE DATA AND CONTINUE.

FIGURE 4.3 Input Form R012 for Initial Concentrations of Principal Radionuclides

A principal radionuclide is a radionuclide with a half-life longer than one year. An associated radionuclide is a decay product with a half-life of one year or less. The radionuclides "associated" with a principal radionuclide consist of all decay products down to, but not including, the next principal radionuclide in the chain. It is assumed that all associated radionuclides are in secular equilibrium with their principal radionuclide in the contaminated zone and also at the location of human exposure.

The single-radionuclide soil guidelines do not depend on the soil concentrations; even if the concentrations are not known, values for these quantities can be obtained by entering any nonzero concentration. The mixture sums, however, depend on the concentrations; hence, calculated mixture sum values are valid only if the concentrations are known.

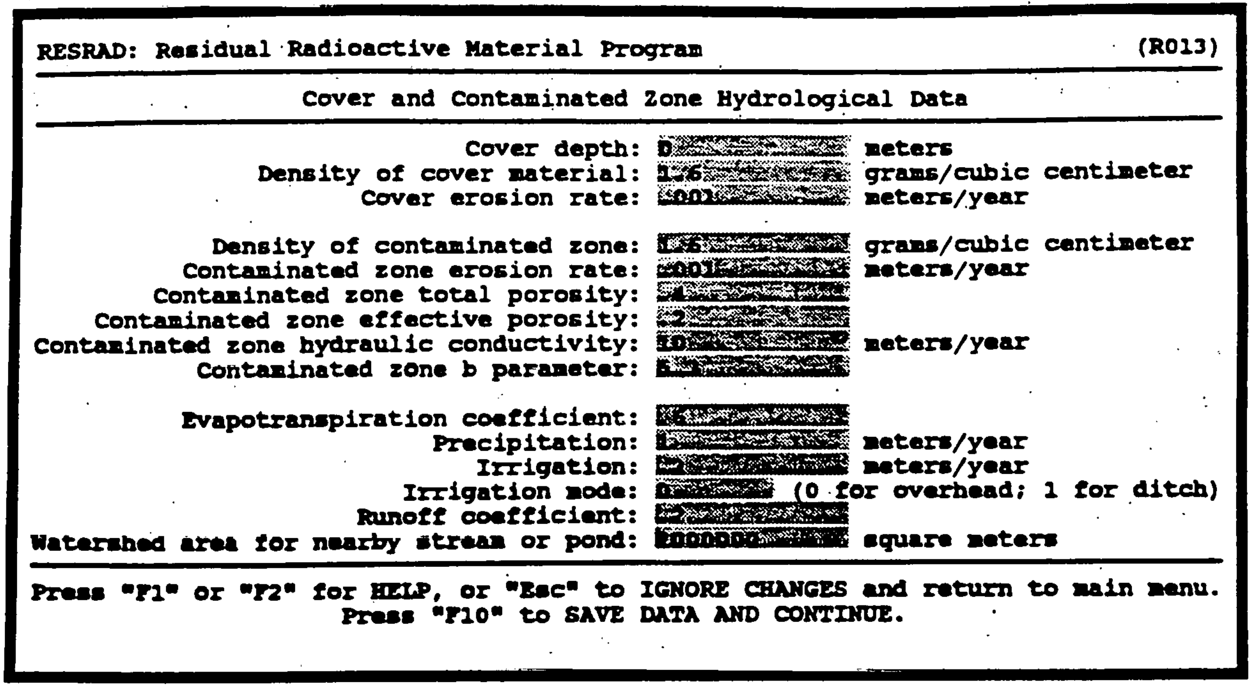
If the contaminated zone is treated as a single homogeneous contaminated zone, the mixture sum should be calculated using the radionuclide concentrations averaged over a compact 100-m^2 area or one-third of the peak radionuclide concentrations within the same area — whichever gives the largest mixture sum. A 100-m^2 area that gives the largest mixture sum should be used if the area of the contaminated zone is larger than 100 m^2. The entire contaminated zone area should be used if this area is less than 100 m^2 but greater than 1 m^2. If the contaminated zone area is less than 1 m^2, peak radionuclide concentrations should be used.

If the contaminated zone is treated as an inhomogeneous contaminated zone, then the contaminated zone may be divided into subzones within each of which the peak concentration does not exceed the average concentration by a factor larger than three. The mixture sums for these subzones are then calculated as if each subzone were an isolated, homogeneous contaminated zone. The total of the mixture sums for subzones within a 100-m^2 area must be less than 1 for any 100-m^2 area within the contaminated zone.

The current version of RESRAD will calculate the mixture sum only for a single homogeneous zone or subzone. For an inhomogeneous contaminated zone, the subzone contributions must be summed separately according to the procedures described in Section 3.3.

4.5.3 Cover and Contaminated Zone Hydrological Data

Input form R013, shown in Figure 4.4, is used to access cover and contaminated zone hydrological data. The definitions of these data and information for determining appropriate site-specific values are given in Appendix E. The DENSITY OF COVER MATERIAL and COVER EROSION RATE appear on the form only if the cover depth is not zero.



```
RESRAD: Residual Radioactive Material Program                          (R013)
                Cover and Contaminated Zone Hydrological Data

                              Cover depth: 0            meters
                Density of cover material: 1.6          grams/cubic centimeter
                       Cover erosion rate: .001         meters/year

                Density of contaminated zone: 1.6       grams/cubic centimeter
              Contaminated zone erosion rate: .001      meters/year
            Contaminated zone total porosity: .4
        Contaminated zone effective porosity: .2
     Contaminated zone hydraulic conductivity: 10       meters/year
                Contaminated zone b parameter: 5.3

              Evapotranspiration coefficient: .5
                              Precipitation: 1          meters/year
                                 Irrigation: .2         meters/year
                             Irrigation mode: 0   (0 for overhead; 1 for ditch)
                          Runoff coefficient: .2
   Watershed area for nearby stream or pond: 1000000    square meters

Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                   Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.4 Input Form R013 for Cover and Contaminated Zone Hydrological Data

The COVER DEPTH is the distance, in meters (m), from the ground surface to the location of the uppermost soil sample with radionuclide concentrations that are clearly above background.

The bulk density of dry soil, specified in grams per cubic centimeter (g/cm^3), should be used for the DENSITY OF COVER MATERIAL and the DENSITY OF CONTAMINATED ZONE. A default value of 1.6 g/cm^3 can be used for preliminary estimates.

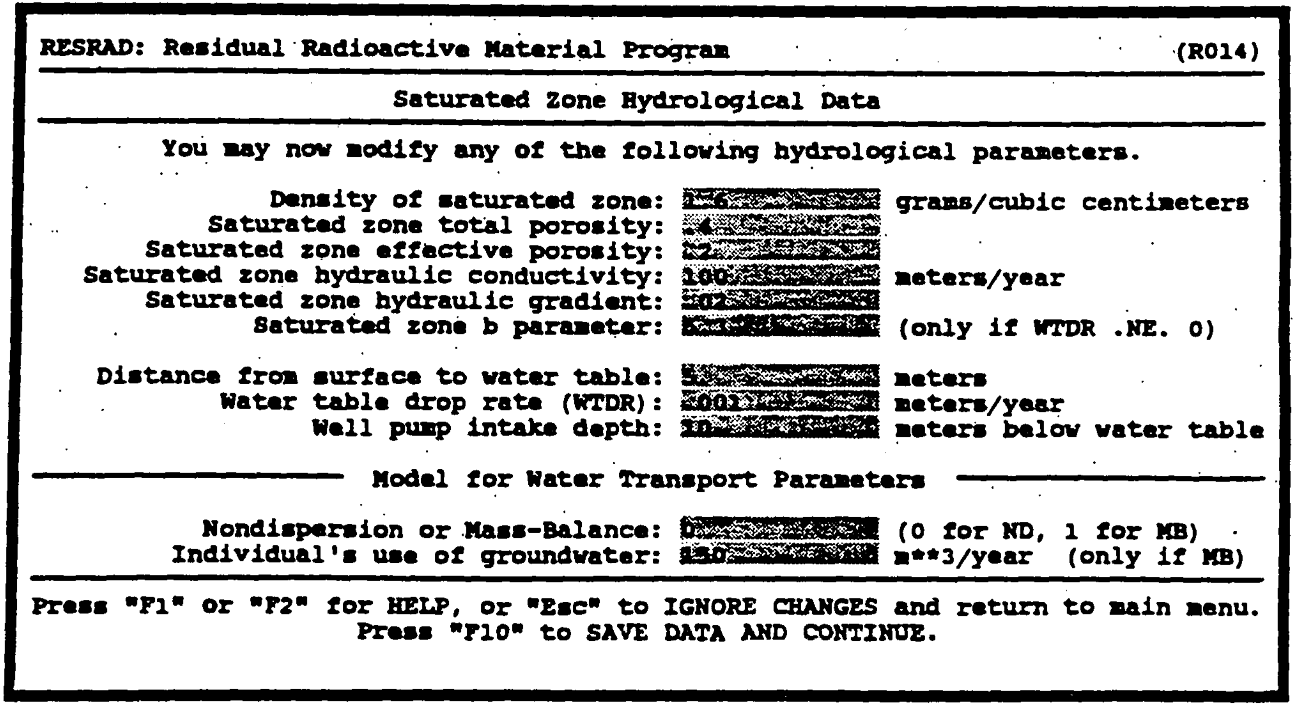
The COVER EROSION RATE and the CONTAMINATED ZONE EROSION RATE are the rates, in meters per year (m/yr), at which soil is removed by erosion. They can be estimated by means of the Universal Soil Loss Equation, as described in Appendix B. A default value of 0.001 m/yr is used for screening estimates. The contaminated zone erosion rate is only significant if and when the cover depth becomes zero. All other parameters are discussed in Appendix E.

Typical parameter values for various soil materials are presented in Tables E.1 through E.7.

4.5.4 Saturated Zone Hydrological Data

Input form R014, shown in Figure 4.5, is used to access saturated zone hydrological data. The definitions and information for determining appropriate values are given in Appendix E. Parameters with special considerations are listed below.

The DISTANCE FROM SURFACE TO WATER TABLE must equal the sum of the cover depth (menu R011), contaminated zone thickness (menu R011), and thicknesses of the unsaturated zone strata below the contaminated zone (menu R015). Immediately prior to the start of calculations (option 10 on the main menu), a consistency check is made to determine that the sum of the above parameters does in fact equal the specified distance to the water table. If the values



```
RESRAD: Residual Radioactive Material Program                          (R014)
-------------------------------------------------------------------------------
                      Saturated Zone Hydrological Data
-------------------------------------------------------------------------------
          You may now modify any of the following hydrological parameters.

                    Density of saturated zone: 1.6        grams/cubic centimeters
               Saturated zone total porosity: .4
           Saturated zone effective porosity: .2
      Saturated zone hydraulic conductivity: 100.         meters/year
          Saturated zone hydraulic gradient: .02
                 Saturated zone b parameter: [illegible]  (only if WTDR .NE. 0)

         Distance from surface to water table: 5          meters
                  Water table drop rate (WTDR): .001      meters/year
                       Well pump intake depth: 10         meters below water table

--------------------- Model for Water Transport Parameters ---------------------

              Nondispersion or Mass-Balance: 0            (0 for ND, 1 for MB)
            Individual's use of groundwater: 250          m**3/year  (only if MB)
-------------------------------------------------------------------------------
Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                      Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.5 Input Form R014 for Saturated Zone Hydrological Data are not in balance, the user is prompted for corrections. The code will not execute until the imbalance is removed.

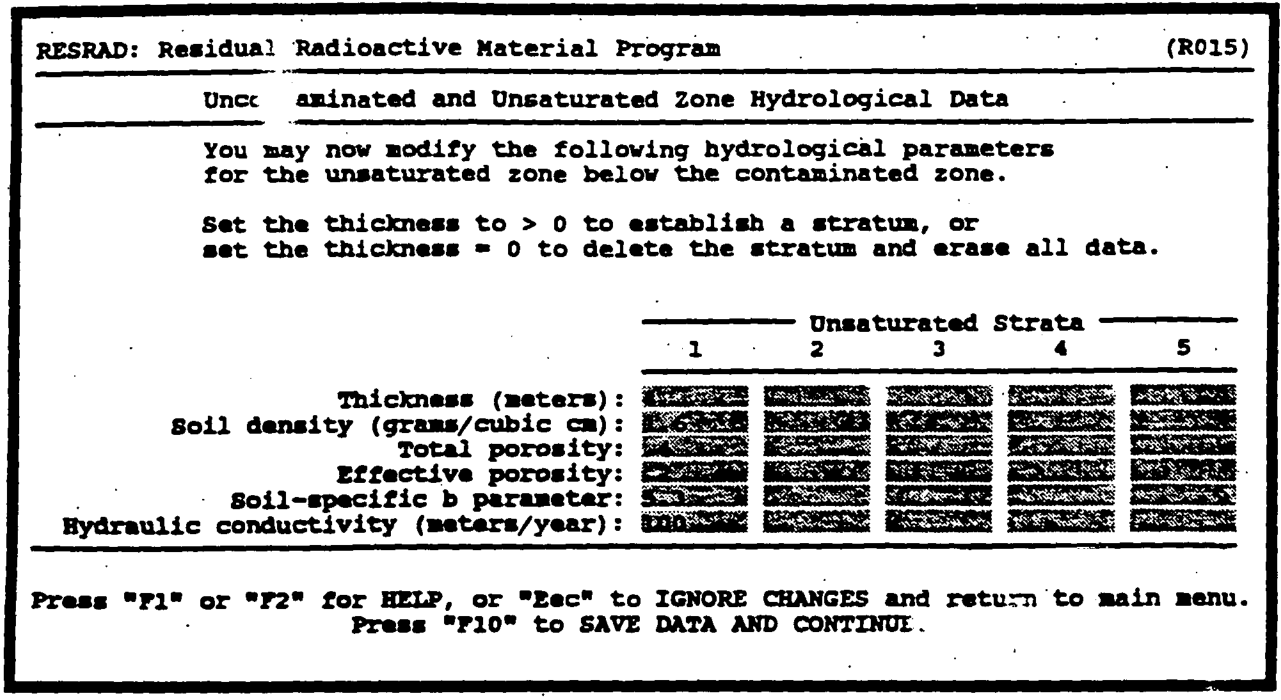
The WATER TABLE DROP RATE is the rate, in meters per year (m/yr), that the depth of the water table decreases. If the water table drop rate is not zero, the unsaturated zone thickness will be created or increased. The saturation of this newly created unsaturated zone is estimated by the hydrological parameters of the saturated zone. The SATURATED ZONE b PARAMETER appears on the form only if the water table drop rate is greater than zero.

The NONDISPERSION or MASS-BALANCE "parameter" selects which of two models will be used for water/soil concentration ratio calculations. A "0" (default) selects the ND model, and "1" selects the MB model. The MB model is not recommended for contaminated zones with an area greater than 1,000 m^2. The ND model can be used for an area of any size.

The INDIVIDUAL'S USE OF GROUNDWATER appears on the form only if the MB model is selected (see above). It is the total volume of well water withdrawn per year, in cubic meters per year (m^3/yr), for use by an individual. The default value is 150 m^3/yr.

4.5.5 Uncontaminated and Unsaturated Zone Hydrological Data

The uncontaminated and unsaturated zone is the portion of the uncontaminated zone that lies below the bottom of the contaminated zone and above the groundwater table. The code has provisions for up to five different horizontal strata within this zone. Each stratum is characterized by six radionuclide-independent parameters: thickness, soil density, total porosity, effective porosity, soil-specific b parameter, and hydraulic conductivity. Input form R015 for these data is shown in Figure 4.6.



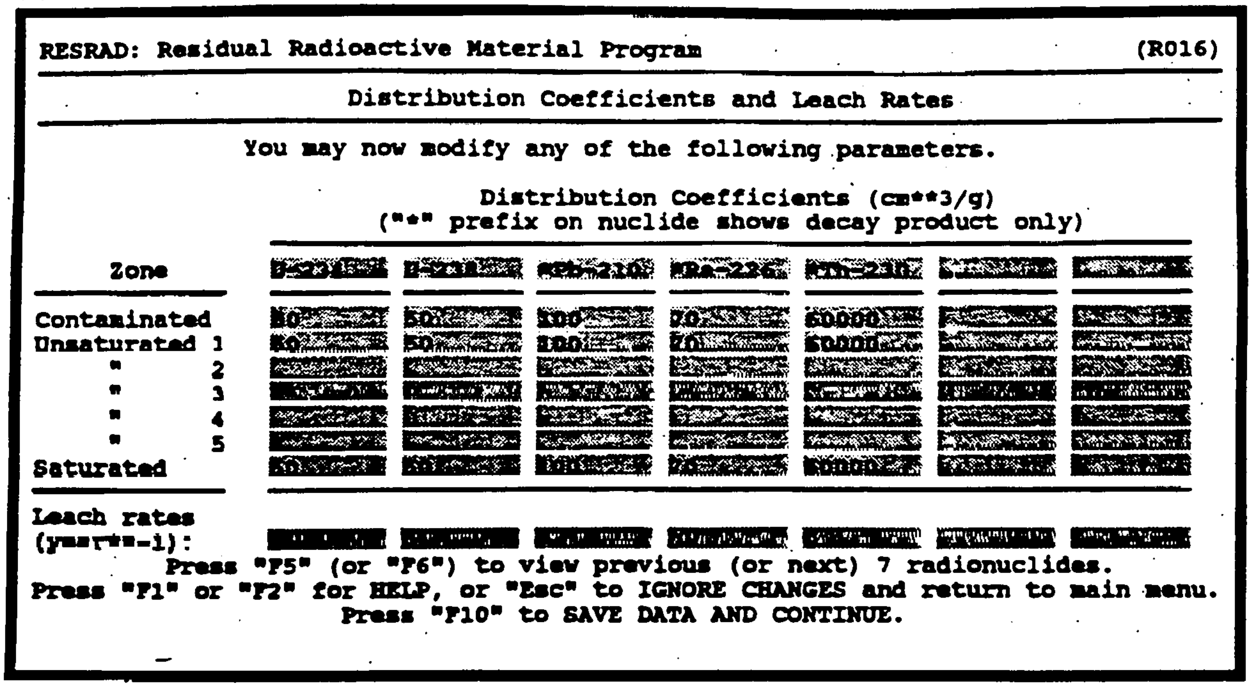
```
RESRAD: Residual Radioactive Material Program                        (R015)

            Uncc  aminated and Unsaturated Zone Hydrological Data

            You may now modify the following hydrological parameters
            for the unsaturated zone below the contaminated zone.

            Set the thickness to > 0 to establish a stratum, or
            set the thickness = 0 to delete the stratum and erase all data.

                                         ─────── Unsaturated Strata ──────
                                           1      2      3      4      5

                      Thickness (meters):
           Soil density (grams/cubic cm):
                         Total porosity:
                     Effective porosity:
                 Soil-specific b parameter:
     Hydraulic conductivity (meters/year):

Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                      Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.6 Input Form R015 for Uncontaminated and Unsaturated Zone Hydrological Data

Data must be entered for each stratum used in the calculation. Entering a nonzero thickness for a stratum activates that stratum and, similarly, changing the thickness to zero deletes the stratum. Default values will be supplied by the code for all parameters of an active stratum; however, the use of true site-specific data is strongly recommended. Data for estimating hydrological parameters for different soil types are given in Appendix E.

4.5.6 Distribution Coefficients and Leach Rates

Input form R016, shown in Figure 4.7, is used to access distribution coefficients and leach rates. If the radionuclide leach rates in the contaminated zone are known, these leach rates should be entered in the LEACH RATES fields. If a leach rate is entered (i.e., >0), it will be used to calculate the leaching of radionuclides from the contaminated zone; if a leach



```
RESRAD: Residual Radioactive Material Program                          (R016)
                Distribution Coefficients and Leach Rates
             You may now modify any of the following parameters.

                          Distribution Coefficients (cm**3/g)
                       ("*" prefix on nuclide shows decay product only)
     Zone          U-234   U-238   *Pb-210  *Ra-226  *Th-230
Contaminated       50      50      100      70       60000
Unsaturated 1      50      50      100      70       60000
     "      2
     "      3
     "      4
     "      5
Saturated          50      50      100      70       60000

Leach rates
(year**-1):
        Press "F5" (or "F6") to view previous (or next) 7 radionuclides.
Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                   Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.7 Input Form R016 for Distribution Coefficients and Leach Rates

rate is not entered (i.e., = 0), the program will calculate the leach rate using the distribution coefficient for the contaminated zone.

In addition to the on-site principal radionuclides, form R016 allows parameters to be entered for decay product principal radionuclides. A "*" prefix is added to the symbol of all principal radionuclides present as decay products only.

The input form displays lines for the contaminated and saturated zones and for each unsaturated stratum below the contaminated zone with a nonzero thickness. Default distribution coefficients are provided automatically by the code for most nuclides. However, these values should be used with care because site-specific distribution coefficients can vary over many orders of magnitude depending on soil type, pH, redox potential, and presence of other ions. Replacement of the default values with site-specific data is always recommended. Tables of generic distribution coefficients for various nuclides are given in Appendix E.

Seven radionuclides are displayed at once on the screen, but the user can page through all the radionuclides, seven at a time, by pressing the "F5" (previous set) or "F6" (next set) keys.

The on-site principal radionuclides can be changed by form R012, which is obtained by option 2 on the main menu. The decay product principal radionuclides are added to form R016 automatically. The number of unsaturated strata can be changed by using form R015, which is obtained by option 5 on the main menu.

The most critical groundwater transport data are the distribution coefficients. Site-specific data should be used whenever possible.

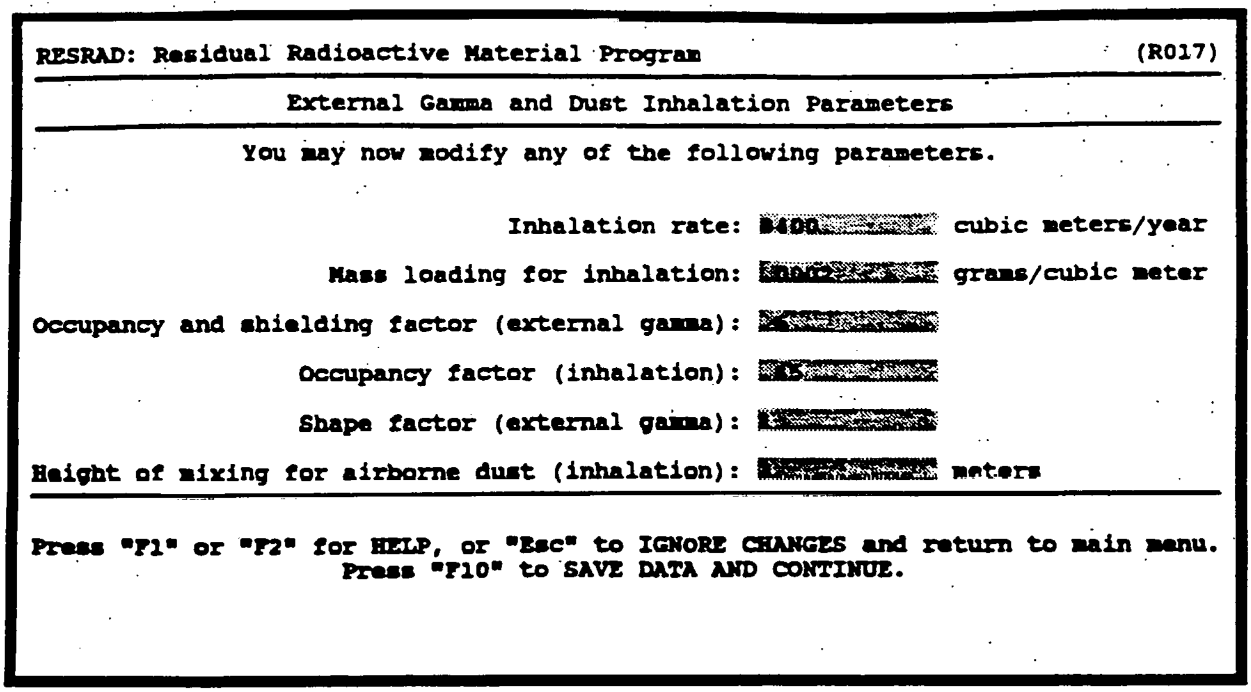
4.5.7 External Gamma and Dust Inhalation Parameters

Input form R017 for external gamma and dust inhalation parameters is shown in Figure 4.8. Values of these parameters applicable in the long term are not strongly site dependent; hence, in most circumstances the generic default values can be used.

4.5.8 Ingestion Pathway Data

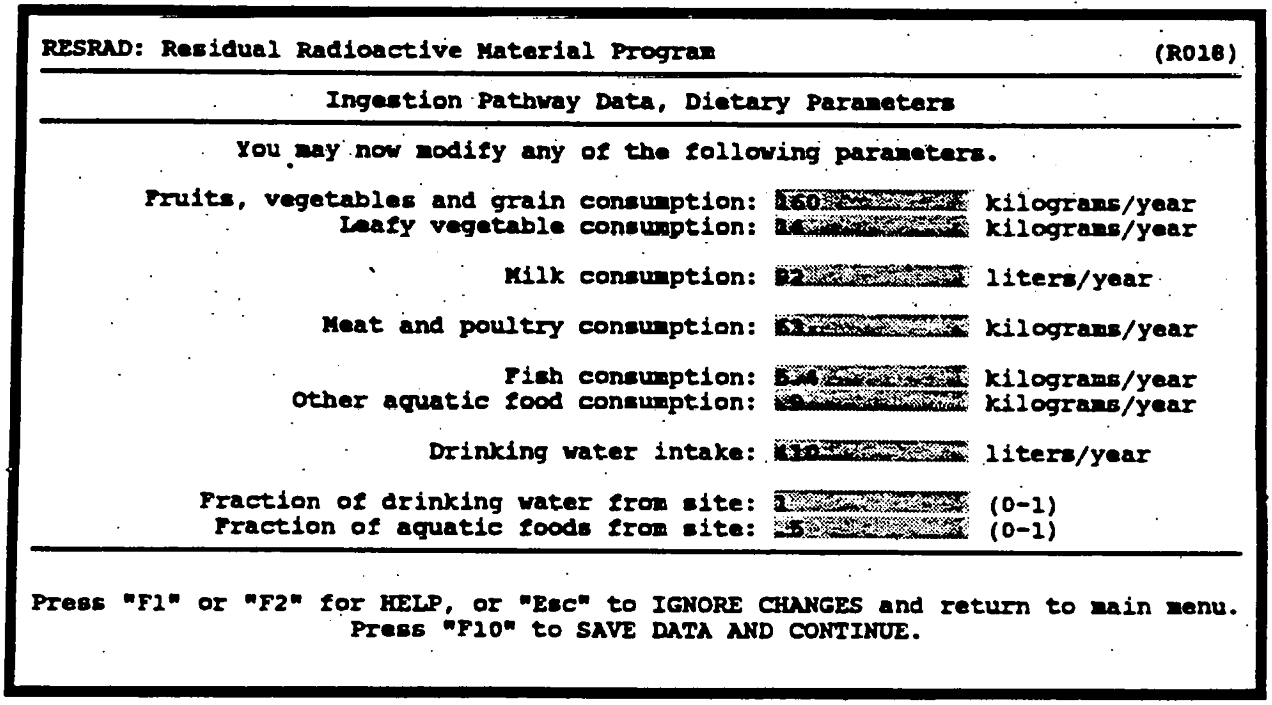
4.5.8.1 Dietary Parameters

Input form R018 for dietary parameters of the ingestion pathway is shown in Figure 4.9. The default parameter values have been chosen to correspond to national averages. The parameters, other than the final two, are not strongly site dependent; hence, use of the generic default values is



```
RESRAD: Residual Radioactive Material Program                      (R017)
                External Gamma and Dust Inhalation Parameters
             You may now modify any of the following parameters.

                                Inhalation rate: 8400      cubic meters/year
                       Mass loading for inhalation: .0002     grams/cubic meter
Occupancy and shielding factor (external gamma): [illegible]
                    Occupancy factor (inhalation): .45
                      Shape factor (external gamma): 1
Height of mixing for airborne dust (inhalation): [illegible] meters

Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                  Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.8 Input Form R017 for External Gamma and Dust Inhalation Parameters



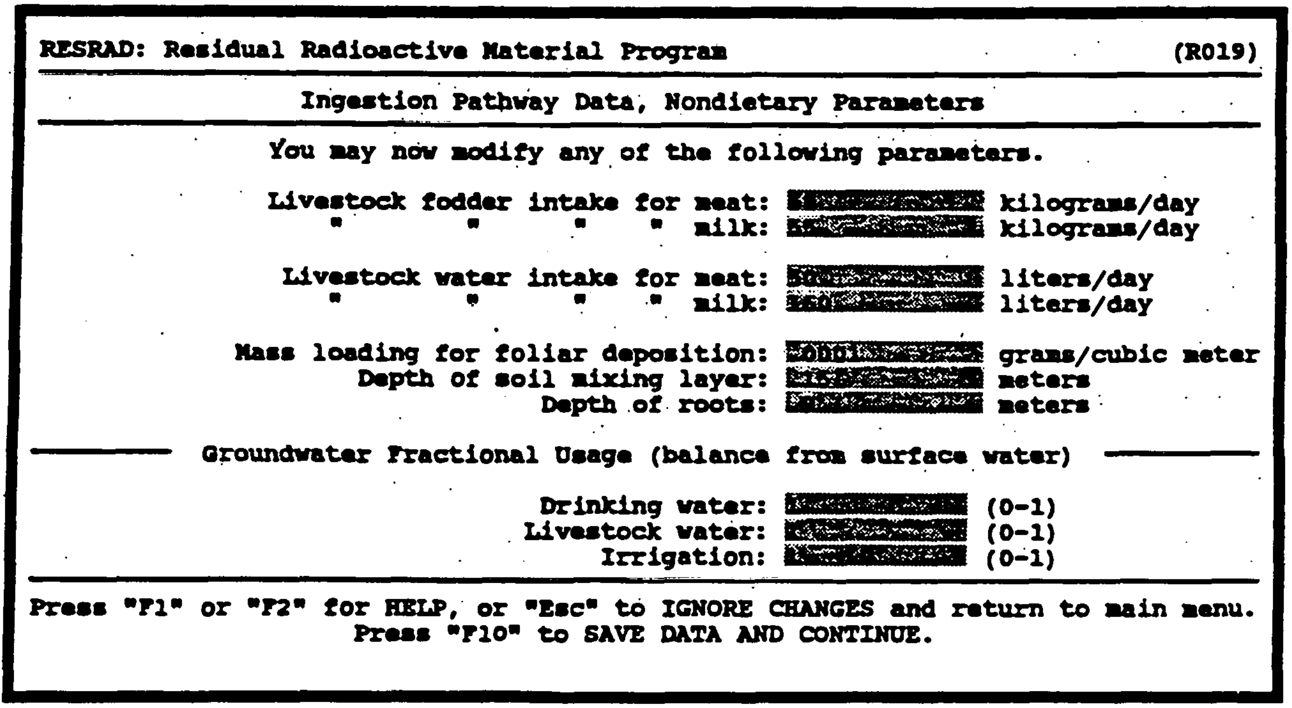
```
RESRAD: Residual Radioactive Material Program                        (R018)
                 Ingestion Pathway Data, Dietary Parameters
             You may now modify any of the following parameters.
       Fruits, vegetables and grain consumption: 160       kilograms/year
                      Leafy vegetable consumption: 14        kilograms/year

                                Milk consumption: 92        liters/year

                     Meat and poultry consumption: 63        kilograms/year

                                Fish consumption: 5.4       kilograms/year
                   Other aquatic food consumption: .9        kilograms/year

                           Drinking water intake: 510       liters/year

            Fraction of drinking water from site: 1         (0-1)
              Fraction of aquatic foods from site: .5        (0-1)

Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                  Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.9 Input Form R018 for Ingestion Pathway, Dietary Parameters usually acceptable. Adjustments for regional differences in diet may sometimes be appropriate.

The FRACTION OF DRINKING WATER FROM SITE and FRACTION OF AQUATIC FOOD FROM SITE parameters allow specification of the fraction of contaminated intake for these pathways. The latter value applies to both fish and other aquatic foods. Off-site material is assumed to be uncontaminated. Defaults for the two fractions are 1.0 and 0.5, respectively.

4.5.8.2 Nondietary Parameters

Input form R019 for nondietary parameters of the ingestion pathway is shown in Figure 4.10. The default parameter values have been chosen to correspond to the national averages. The parameters, other than the final three, are not strongly site dependent; hence, use of the generic



```
RESRAD: Residual Radioactive Material Program                      (R019)
               Ingestion Pathway Data, Nondietary Parameters
               You may now modify any of the following parameters.

             Livestock fodder intake for meat: [illegible]      kilograms/day
               "        "      "     "   milk: [illegible]      kilograms/day

              Livestock water intake for meat: 50               liters/day
               "        "      "     "   milk: 160              liters/day

          Mass loading for foliar deposition: 0.0001           grams/cubic meter
                 Depth of soil mixing layer: [illegible]      meters
                            Depth of roots: [illegible]      meters
-------  Groundwater Fractional Usage (balance from surface water)  -------

                            Drinking water: [illegible]      (0-1)
                           Livestock water: [illegible]      (0-1)
                               Irrigation: [illegible]      (0-1)

Press "F1" or "F2" for HELP, or "Esc" to IGNORE CHANGES and return to main menu.
                    Press "F10" to SAVE DATA AND CONTINUE.
```

FIGURE 4.10 Input Form R019 for Ingestion Pathway, Nondietary Parameters default values is usually acceptable. Adjustments for regional differences in farming practices may sometimes be appropriate.

The three groundwater fractional usage parameters -- drinking water, livestock water, and irrigation -- are included primarily to allow groundwater (well) and surface water (pond) scenarios. Hence, the fractions will usually be set at 1 or 0. A value of 1 selects 100% groundwater usage, and 0 selects 100% surface water. The default for all three parameters is 1. For livestock water and irrigation, all usage is assumed to be from the site. The fraction of contaminated drinking water, i.e., from the site, can be varied via a parameter in form R018.

4.6 CONTROL OF RESRAD PATHWAYS

RESRAD always computes the radiation dose resulting from seven potential pathways: (1) direct exposure to external radiation from the contaminated soil material, (2) internal radiation from inhalation, (3) internal radiation from ingestion of plant foods grown on-site and irrigated with water drawn from an on-site well or pond, (4) internal radiation from ingestion of meat from livestock fed with fodder grown on-site and water drawn from an on-site well or pond, (5) internal radiation from ingestion of milk from livestock fed with fodder grown on-site and water drawn from an on-site well or pond, (6) internal radiation from ingestion of aquatic foods from an on-site pond, and (7) internal radiation from drinking water from an on-site well or pond.

In many situations, certain of these pathways are not important or the user may deliberately wish to suppress selected pathways. Table 4.1 is a list of key parameters that, if set to zero, will eliminate the indicated pathway.

TABLE 4.1 Key Parameters to Eliminate Pathways

Pathway	Menu	Parameter[a]
1	R017	Shape factor (external gamma)
2	R017	Occupancy factor (inhalation)
3	R018	Fruits, vegetables, and grain consumption[b]
3	R018	Leafy vegetable consumption[b]
4	R018	Meat and poultry consumption
5	R018	Milk consumption
6	R018	Fraction of aquatic foods from site
7	R018	Fraction of drinking water from site

[a] Set indicated parameter(s) to zero to suppress pathway.

[b] Set both parameters to zero for pathway 3.

However, no savings in computation time will result by eliminating a pathway; only the dose contribution is set to zero.

4.7 RESRAD OUTPUT

When option 10 of the main menu is selected, RESRAD enters its RESMAIN computation and report generation phase. A number of brief progress reports will appear on the display monitor, ending with

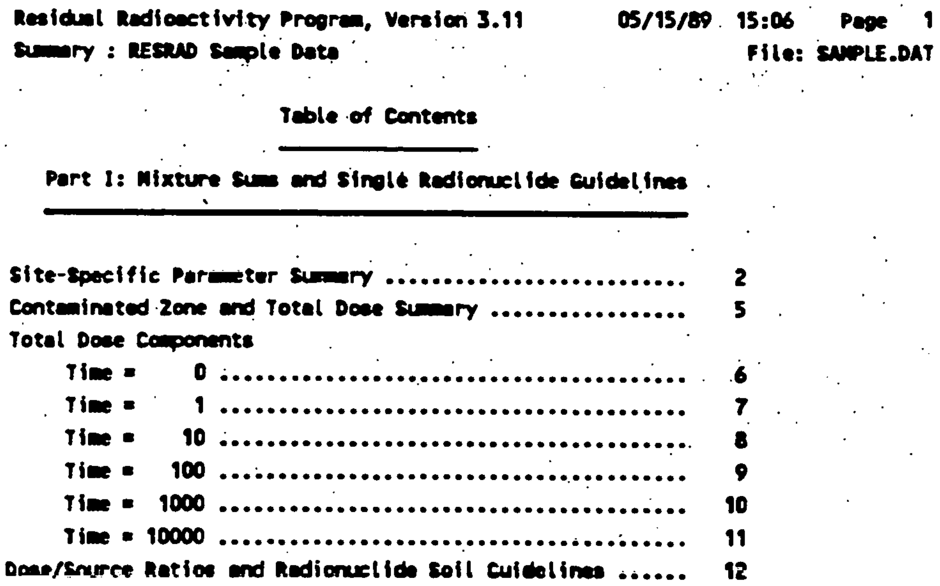
```
RESMAIN program complete, elapsed time =   xxxx.xxxx   seconds.
```

Shortly thereafter, the main menu, R010, will appear on the screen, and the user may continue with any of the options discussed in Section 4.5. However, the customary course will be to examine the output reports just generated via main menu option 11.

When option 11 is selected, the user is prompted to select for viewing the summary or detailed report from the last RESRAD run or any ASCII file. The last option is particularly useful for examining saved output from a prior RESRAD run. It should be noted that it is not necessary to execute RESRAD (or, more strictly, RESMAIN) before using option 11. In general, SUMMARY.REP and DETAILED.REP files from the last RESRAD run will be available and may be examined at any time. However, if appropriate, the user will be warned that the report file may not match the current site-specific data file.

4.7.1 Summary Report

When the summary report option is selected, the first page (table of contents) of a multiple page summary report is displayed on the screen. The table of contents is shown in Figure 4.11. Subsequent report pages can be displayed by entering the page number and pressing either the "Enter" or "F10" function key. Pressing "Esc" will return to the main menu, R010. On-line help, primarily key strokes to aid in the page display, is available via the "F1" function key. Function keys "F7" and "F8" can be used to produce hard copy output (see Section 4.7.3).



```
Residual Radioactivity Program, Version 3.11        05/15/89  15:06   Page   1
Summary : RESRAD Sample Data                                   File: SAMPLE.DAT

                              Table of Contents
                              _________________

     Part I: Mixture Sums and Single Radionuclide Guidelines
     _______________________________________________________

Site-Specific Parameter Summary ..........................     2
Contaminated Zone and Total Dose Summary .................     5
Total Dose Components
     Time =      0 ........................................    6
     Time =      1 ........................................    7
     Time =     10 ........................................    8
     Time =    100 ........................................    9
     Time =   1000 ........................................   10
     Time =  10000 ........................................   11
Dose/Source Ratios and Radionuclide Soil Guidelines ......   12
```

FIGURE 4.11 Typical Table of Contents for Summary Report

The summary report starts with a listing of all site-specific parameter values, both as used and default, for the current run. This is followed by a brief summary of the contaminated zone. Summaries of the estimated total dose, TDOSE(t), and the total mixture sum, M(t), for the user-selected time periods, t, are listed next [TDOSE(t) in mrem/yr and t in yr]. If a total dose maximum occurred within the time frame selected, this maximum and its time of occurrence is determined by a standard iterative procedure (Press et al. 1986, p. 283). Convergence is ensured but caution is advised because more than one local maximum may exist within the time frame specified. Repeat runs with different time periods is the only check on the solution. The total dose maximum and time of occurrence (within the listed plus or minus range) will not change if the maximum is unique.

The total mixture sum, M(t), is the estimated effective dose equivalent -- expressed as a multiple of the basic dose limit -- that a member of the critical population group might receive at time t following the radiological survey as a consequence of the residual radioactivity. A site may be certified in compliance with guidelines only if the mixture sum does not exceed the value of one at any time within the time horizon.

The summary report continues with the total dose components for all individual pathways at different times. The results are presented both as actual dose (mrem/yr) and as percent contribution to the total.

The final table series of the summary report presents:

- Total dose/source ratios, DSR(i,t) in (mrem/yr)/(pCi/g), summed over all pathways for radionuclide i and time t;
- Single-radionuclide soil guidelines, G(i,t) in pCi/g, for radionuclide i and time t; and

- Values of DSR(i,t) and G(i,t) for t equals the minimum G(i,t) for radionuclide i and for t equals the maximum total dose.

The single-radionuclide soil guidelines are the concentration guidelines that would apply if only one radionuclide were present. A single-radionuclide soil guideline, G(i,t), is the magnitude of the initial concentration of the i^{th} principal radionuclide that would result in a potential radiation dose equal to the basic radiation limit to a member of the critical population group at time t. Thus, if only the i^{th} radionuclide were present, the initial concentration S(i,0) would have to satisfy the inequality $S(i,0)/G(i,t) \leq 1$ for all times up to the time horizon. G(i,t) is readily obtained by dividing the basic radiation dose limit by the total dose/source ratio, DSR(i,t). The minimum G(i,t) values, or more correctly the corresponding maximum DSR(i,t), are obtained for each radionuclide i with the same iterative procedure used for the maximum total dose.

The total mixture sum enables the user to determine whether the concentrations of residual radioactive material are in compliance with guidelines. The total dose components and single-radionuclide soil guidelines enable the user to identify the critical pathways and radionuclides.

4.7.2 Detailed Report

When the detailed report option is selected, the first page (table of contents) of a multipage detailed report is displayed on the screen. A typical table of contents is shown in Figure 4.12. Individual report pages can be displayed and printed (or saved) in the same manner as for the summary output displays. The detailed displays present all of the intermediate results used to evaluate the formulas in Appendixes B through E for the

```
Residual Radioactivity Program, Version 3.11       05/15/89  15:06   Page  1
Detailed: RESRAD Sample Data                                  File: SAMPLE.DAT

                            Table of Contents
                            _________________

Part II: Source Terms, Factors, and Parameters for Individual Pathways
______________________________________________________________________


Iteration Logs
    Maximum U-234  Dose/Source Ratio .....................    2
    Maximum U-238  Dose/Source Ratio .....................    3
    Maximum Total Dose ...................................    4
Source Factors for Ingrowth and Decay ....................    5
Ground Pathway
    Source Term Parameters ...............................    6
    Time Dependence of Source Geometry ...................    7
    Occupancy, Area and Depth Factors ....................    8
    Dose Conversion and Environmental Transport Factors .    8
    Dose/Source Ratios ...................................    8
Inhalation Pathway
    Dose/Source Ratios ...................................    9
    Pathway Factors ......................................    9
Groundwater and Surface Water Pathway Segments
    Transport Time Parameters for Unsaturated Zone Strata   10
    Dilution Factor and Rise Time Parameters for
            Nondispersion (ND) Model .....................   11
    Primary Parameters Used to Calculate Ratios .........   11
    Water/Soil Concentration Ratios .....................   12
Food Pathways
    Area and Depth Factors ..............................   13
    Water Exposure Factors ..............................   15
    Environmental Transport and Dose Conversion Factors
        Plant ...........................................   18
        Meat ............................................   19
        Milk ............................................   20
        Fish ............................................   21
        Drinking Water ..................................   21
    Dose/Source Ratios
        Plant ...........................................   22
            Plant Total .................................   23
        Meat ............................................   24
            Meat Total ..................................   25
        Milk ............................................   26
            Milk Total ..................................   27
        Fish ............................................   28
        Drinking Water ..................................   28
    Concentration Ratios
        Plant/Air and Plant/Water .......................   29
        Plant/Soil ......................................   29
        Meat/Fodder, Fodder/Air, Fodder/Water ...........   31
        Fodder/Soil .....................................   32
        Meat/Soil .......................................   33
        Milk/Soil .......................................   34
```

FIGURE 4.12 Typical Table of Contents for Detailed Report different pathway contributions. Users will not normally make use of these tables. The detailed data are used for verifying the code and can be useful for gaining insight into the transport mechanisms by enabling a more detailed investigation of the effect of parameter changes on pathway factors, environmental transport factors, and dose/source ratios for the different pathways.

4.7.3 Hard Copy

At any time while viewing RESRAD output (or any other ASCII file) via main menu option 11, hard copy can be produced (if the printer is turned on) by pressing the "F7" or "F8" function keys. The "F8" key produces an immediate copy of the current screen page only.

The "F7" function is a more versatile but complicated option. The user is first informed of the current hard copy destination (printer or disk file name) and allowed to make a change, if desired. If an existing disk file is selected, the user can specify to overwrite or append. The following series of options can then be exercised repeatedly (until "Esc"):

- Enter the first and last page numbers, separated by a space or comma, to print (or save). Enter a single page number to print that page only.
- Press "Enter" key only to print current page (current page number will be displayed on screen).
- Press "D" or "d" to print entire document (file).
- Press "P" or "p" to pass ASCII control characters to the printer.
- Press "Esc" to resume normal RESRAD viewing of the file.

To preserve an option of an earlier RESRAD version, the "*" key is identical to the "F7" key.

4.8 TECHNICAL SUMMARY

This manual is not intended to be a programmer's guide, but a brief technical summary should be useful. For more information, the user is referred to RESRAD.DOC, which is included among the supplementary files in the distribution diskette. The following material is selected from that file.

RESRAD consists of two main programs: RESPC and RESMAIN. RESPC is a compiled Basic program (Microsoft QuickBASIC 4.5) that handles all user interaction with RESRAD, and RESMAIN is the Fortran code (Lahey F77L, Ver. 3.0) that does the actual computations and generates the output tables.

RESPC is loaded first (via RESRAD.BAT, NEXT.ORI, and NEXT.BAT). Within RESPC, the user may input selected site-specific data files, review and/or modify the parameters, save the data in a selected file, view and/or print selected results from the last RESMAIN computation, and run RESMAIN for a new computation. If RESMAIN is called, RESPC is reloaded automatically when RESMAIN completes, and the user may continue with any of the RESPC options.

Most of the tasks accomplished by RESPC are actually performed by FORM.BAS and PAGER.BAS, a library of subroutines comprising the FORMPACK package developed by the Energy and Environmental Systems Division of Argonne National Laboratory. FORMPACK provides a standard system for menus, data entry, and output report display for microcomputer application programs. RESPC is the driving program.

To conserve memory, RESMAIN is organized with an overlay structure. The code is highly optimal in that no overlay is called a second time. PLINK86plus, Ver. 2.24, Phoenix Technologies Ltd., is the overlay linkage editor used. Object modules are produced as usual via the Lahey Fortran compiler, and the file RESMAIN.LNK automates the link process by the command

```
PLINK86 @RESMAIN
```

5 APPLICATION OF THE AS LOW AS REASONABLY ACHIEVABLE PROCESS

Application of the as low as reasonably achievable (ALARA) process is required in DOE environmental protection and health and safety orders and in the DOE residual radioactive material guidelines. The objective of the ALARA process is the attainment of dose levels that are as far below applicable limits as is practicable and reasonably achievable, taking into account many factors — including technical, economic, safety, and social. Unlike the "below regulatory concern" or "de minimis" concepts, which can define a lower limit of cleanup that should be carried out before remedial action is taken, no set of defined dose levels exists for determining when ALARA is achieved. The ALARA process must be applied throughout the project, from planning to field work. Application of the ALARA process through planning activities occurs both in the development of guidelines and in the site-specific application of the guidelines. This does not necessarily require a significant level of additional effort.

Application of the ALARA process to any radiation protection project is necessary but somewhat subjective. The success of the process depends on good judgment of the health physicists and managers who are responsible for its application. Applying the ALARA process to remedial actions is, in many cases, more complex than applying it to operating situations. In operating facilities, one can identify the waste streams, pathways, potential exposed individuals, and likely time periods for the operation of the facility; also, the dose reductions from specific control actions having specific costs over the expected time period can be estimated and compared. In the case of contaminated facilities requiring remedial action, it is more difficult to define all of these elements, particularly if potential effects must be projected for many years into the future.

For sites to be certified in compliance with the guidelines, the intent is to reduce the concentrations of residual radioactive materials to as low as reasonably achievable considering technical, economic, and social factors. At sites where the residual materials are reduced to levels requiring that restrictions be placed on the use of the facility, the ALARA process includes the establishment of institutional controls to reduce exposure.

In remedial action or decontamination and decommissioning work under DOE jurisdiction, the concentration limit or guideline for a particular radionuclide in soil or on surfaces — as derived from the basic dose limit by pathway analysis — is generally assumed to be the upper limit of that nuclide for remediation at a site. Each remedial action will be different from another in some respect. In one situation, the contamination might be spread uniformly over the surface whereas, in others, hot spots of radioactive material might be buried at various depths below the surface. Also, a portion of the radioactive material might have been eroded to another area by wind or water or, in buildings, the contamination might be buried beneath paint or a covering wall. Nonuniform distribution of the contamination is usually the rule rather than the exception. In applying the ALARA process, the first task is to ensure that the area of concern is at or below the chosen limit and the second task is to determine that the remaining contamination is ALARA.

5.1 DETERMINING ALARA

Two approaches are used for determining ALARA: (1) qualitative, or judgmental, procedures and (2) quantitative, or optimization, procedures. Both procedures involve consideration of economic, social, and technical factors to ultimately determine a limit for the site under consideration. The level of detail and documentation involved in a qualitative comparison can vary significantly. The effort should be commensurate with risk and cost. If overall risks are very low and costs for further reduction in potential doses are very high, the decision is obvious and minimal analysis and documentation are required. However, as potential doses approach levels that are significant in comparison to the dose limit, more detail is warranted. The quantitative procedure involves a cost-benefit analysis, as proposed by the ICRP (1973, 1977, 1983), to quantify the costs of improved radiological protection and the costs associated with health detriment from radiation and from other factors not related to radiation.

5.1.1 Qualitative Analysis

The more traditional method for determining ALARA consists of making individual decisions for each situation encountered. A number of such decisions may be made for different parts of the overall area as work progresses; that is, it is not necessary to make ALARA decisions only after all of the work has been completed. Frequently, there are subareas with difficult problems and, before moving to the next subarea, the work can be concentrated on these difficult areas until the ALARA process is satisfied. The temptation to set an "ALARA limit" should be avoided because ALARA is a process rather than a limit and, furthermore, unexpected developments may result in greatly increased costs to meet the derived limit or in even the inability to meet the limit. If an area cannot be decontaminated to the derived concentration guideline, a decision must then be made as to whether the land can be used without radiologically based restrictions or whether controls will be required. A key factor in determining ALARA is the individual judgment of the responsible person, based on that person's understanding of the relevant economic, social, and technical factors.

Economic factors are primarily the concern of those responsible for the remedial action. The most important factor is, of course, the cost of doing the work. The cost of remedial action typically includes the initial planning cost as well as the cost of operation and associated labor, materials, equipment, energy, and services over the period of remediation. The initial planning cost includes all costs relative to planning and engineering tasks -- such as performance of radiation surveys; calculation of contamination inventories; performance of engineering studies; preparation of a remedial action plan, with descriptions and specifications; and preparation of detailed cleanup and removal procedures. The operating cost includes all costs associated with physical tasks such as removal of contamination; removal of equipment and structures; storage, packaging, shipping, and burial of wastes; and final site and facility restoration or preservation. In addition to the cost of doing the work, economic factors should also include the potential impact on current facilities, operations, and the environment. The impact and associated cost of either the action or no action may have a positive or negative effect on current operations or the environment, and such cost may be significant in comparison with the cost of performing the remedial action.

Social factors involve the risk to the people exposed, i.e., the risk to persons carrying out the operations, the risk to persons in the vicinity of the area during operations, and the perceived risk to those persons in the surrounding area after operations. It is easy to consider only the potential exposure scenarios and the risk to those persons who might occupy the area immediately following cleanup; however, many of the sites are contaminated with long-lived radioactive materials, and cleanup will affect the potential risks received over many years and perhaps over areas away from the site. Therefore, in evaluating risks, these temporal and spatial factors must be taken into account. In carrying out remedial actions on contaminated areas and vicinity properties, occupational exposures are likely to occur. Controlling the radiation exposure of the work force should be based on DOE guidance for occupational exposure, and all unnecessary doses should be eliminated. In applying the ALARA process, the reductions in risks to users of the site and to individuals in the area surrounding the disposal site following the remedial action should be compared with the incremental risks to individuals in the same area and to workers during remedial action activities. It is most desirable that such a comparison would result in a beneficial balance in overall risk.

Technical factors are primarily related to technological alternatives for treatment and disposal of the contaminated materials. Also, the technique used to determine the level and extent of contamination is important. The number of soil samples that can be taken might be limited, and the methods of measurement might have limitations with regard to sensitivity. Increasing the number of samples taken and using state-of-the-art instruments or methods for all samples in order to accurately determine the above-background distribution of all radionuclides would be prohibitively expensive. A decision must be made regarding the number and location of samples and the sensitivity of the instruments used for establishing compliance with concentration guidelines and for implementing the ALARA process.

5.1.2 Cost-Benefit Analysis

The ICRP has recommended the use of cost-benefit analysis to determine the levels of population exposure that are reasonably achievable. This quantitative technique compares the cost of health detriment and other factors relative to radiation with the cost of radiation dose reduction.

To apply cost-benefit analysis to radiation protection, a policy decision is often made that assumes the action is already justified (a net positive benefit); this decision may be based on a policy decision or on broad political, economic, or social considerations. Under this assumption, a simplified form of differential cost-benefit analysis can be used. The objective is to ensure that the resultant exposures are kept ALARA and below the dose limit. In this differential analysis, consideration shifts from the total benefit to the change in net benefit that might result from requiring the action to be performed at one level of exposure rather than another. The benefit of a reduction in exposure is compared with the cost of achieving the reduction. The theoretical optimum is achieved when a marginal increase in the cost of protection is exactly balanced by a marginal decrease in the cost of detriment, i.e.,

$$\left.\frac{dX}{dS}\right|_{S^*} = -\left.\frac{dY}{dS}\right|_{S^*} \qquad (5.1)$$

where

X = cost of achieving a given level of protection and

Y = cost of the radiological detriment at the level of protection.

S* is the optimum level of collective dose equivalent, where the reduction of detriment per unit dose equivalent balances the increase in cost of protection per unit dose equivalent.

Such a cost-benefit analysis is a helpful procedure for quantifying both the costs of improved protection and the costs associated with radiological detriment from alternative courses of action. This technique is particularly useful when extensive remedial action activities for radiological protection are being considered, when individuals are being exposed to levels close to the dose limit, or when potential collective doses are large. For small-scale problems, simplified analyses are sufficient. However, because a cost-benefit analysis does not encompass all judgments that might be involved in a specific decision concerning radiological protection, such an analysis is only one procedure for quantitatively determining some of the input to that decision.

In order to implement the ALARA process using cost-benefit analysis, the costs of radiological protection and radiological detriment must be assessed. The degree of sophistication and effort given to a cost-benefit assessment should reflect the scale of the radiological problem being considered. For a large-scale problem, the assessment might be an evaluation combining engineering designs, control technologies, accounting techniques, and the use of mathematical models for pathway analyses and dose calculations. For a small-scale problem, the assessment might require only simple, intuitive estimates. The estimation of protection costs is generally straightforward.

The assignment of costs to health detriment requires both scientific and other judgment. For example, the NRC established ALARA guidelines for radionuclide emissions from light-water reactors after several years of intensive study of the emissions and the proper levels to which exposures should be limited based on ALARA (NRC 1974). This was possible because the emissions from most light-water reactors are similar and the costs to control the emissions are similar. The cost/risk ratio for deriving ALARA guidelines was assumed to be $1,000 per person-rem. This same cost/risk ratio was also used by NRC in its proposed safety goals for nuclear power plants (NRC 1982); these safety goals were intended to set design objectives for the reliability of safety systems. The EPA has also used an ALARA approach when deriving many of its regulations. Table 5.1 summarizes some examples of expenditures to limit cancer risk, derived from criteria and guidelines issued by the EPA and NRC. The costs range from \$1 to \$200 million per cancer averted. Using a lifetime cancer fatality risk estimate of 1.25×10^{-4} per person-rem, the cost/risk ratios in Table 5.1 are equivalent to economic values of about \$125 to \$25,000 per person-rem of radiological detriment. However, the applicability of these cost/risk ratios to remedial actions is limited at best. Long time periods and uncertainties in property use make them very difficult to use.

TABLE 5.1 Examples of Expenditures to Limit Cancer Risk

Example	Agency	Cost per Cancer Averted[a] (10^6 \$)	Lifetime Risk Level	Comments
Routine radioactive releases from light-water reactor effluents	NRC	10	6×10^{-5}	1975 dollars; no correction for inflation; only fatal cancers
Dioxin contamination at Times Beach, Missouri	EPA	2-160	1×10^{-2}	Includes nonfatal cancers
Drinking water criteria	EPA	20-200	1×10^{-5}	Includes nonfatal cancers
Safety goals for nuclear power plants	NRC	10	1×10^{-4}	1982 dollars; inflation correction included; only fatal cancers
Proposed asbestos ban	EPA	1	4×10^{-5}	Fatal cancers

[a] Estimated cancers, not actual incidence.

Source: McKone (1986).

5.1.3 Summary

In applying the ALARA process, based on either qualitative or quantitative analysis, the social risks incurred from implementing remedial action must first be justified by the reduction in risk that will result. The ALARA process requires that the responsible persons use judgment with respect to what is "reasonably achievable." The economic, social, and technical factors influencing this judgment are highly variable and site specific. At a minimum, the following parameters should be evaluated to the extent practicable before making a decision:

- Maximum dose to an individual member of the public;
- Collective dose to the population;
- Time and duration of exposure of the affected individual or population;
- Likelihood of a specific use scenario occurring at a site;
- Technological alternatives, such as alternative methods for treatment and disposal of the contaminated materials, alternative operating methods for conducting cleanup activities, or alternative institutional controls for reducing exposure;
- Demographic data;
- Occupational doses associated with each technological alternative;
- Costs of each technological alternative; and
- Variations in social impact associated with the various alternatives.

In allocating resources relative to ALARA considerations, the total value should depend on the size of the affected population, the level of risk already experienced from background radiation sources, and the magnitude of risk for each individual. More cost should be assigned to collective doses made up of individual doses approaching the dose limit than to doses made up of individual doses far below the limit.

Because most radionuclides considered for FUSRAP/SFMP sites have long half-lives, the potential radiation risk must be calculated for populations that might exist in hundreds or thousands of years. This calculation becomes increasingly uncertain as the time period of radiological dose assessment increases. To realistically account for these uncertainties, it is reasonable to attach more weight to doses predicted for the present or the near future than to doses predicted for the far future, especially when use scenarios appear unlikely.

5.2 DOCUMENTING ALARA

The documentation of any decision regarding the ALARA process in remedial action provides an invaluable record to show that the process was applied. Because DOE is responsible for the work being done, any judgment regarding the ALARA process is ultimately the responsibility of DOE, and DOE personnel should be involved in the decisions or alternatively in plans that document how the process will be applied for the project and in the field. It is, therefore, essential that ALARA decisions be well documented, including those made in the field and those made prior to or following field work.

6 REFERENCES

Ahrends, S.W., 1987, *Revised Post Remedial Action Survey Protocol,* Attachment to letter from S.W. Ahrends (U.S. Department of Energy, Oak Ridge Operations, Oak Ridge, Tenn.) to G. Hovey (Bechtel National, Inc., Oak Ridge, Tenn.), dated May 26, 1987.

Eckerman, K.F., A.B. Wolbarst, and A.C.B. Richardson, 1988, *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion,* EPA-520/1-88-020, Federal Guidance Report No. 11, prepared by Oak Ridge National Laboratory, Oak Ridge, Tenn., for U.S. Environmental Protection Agency, Office of Radiation Programs, Washington, D.C.

Eve, I.S., 1966, *A Review of the Physiology of the Gastrointestinal Tract in Relation to Radiation Doses from Radioactive Material,* Health Physics, *12*:131.

Gilbert, T.L., et al., 1983, *Pathways Analysis and Radiation Dose Estimates for Radioactive Residues at Formerly Utilized MED/AEC Sites,* ORO-832 (Rev.), prepared by Division of Environmental Impact Studies, Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations, Oak Ridge, Tenn. (March 1983 [reprinted with corrections: January 1984]).

Healy, J.W., 1977, *An Examination of the Pathways from Soil to Man for Plutonium,* LA-6741-MS, Los Alamos Scientific Laboratory, Los Alamos, N.M.

International Commission on Radiological Protection, 1966, *Deposition and Retention Models for Internal Dosimetry of the Human Respiratory Tract,* Health Physics, *12*:173.

International Commission on Radiological Protection, 1968, *The Metabolism of Compounds of Plutonium and Other Activities,* ICRP Publication 19, Pergamon Press, Oxford.

International Commission on Radiological Protection, 1972, *Alkaline Earth Metabolism in Adult Man,* ICRP Publication 20, Pergamon Press, Oxford.

International Commission on Radiological Protection, 1973, *Implications of Commission Recommendations that Doses be Kept As Low As Readily Achievable,* ICRP Publication 22, Pergamon Press, New York.

International Commission on Radiological Protection, 1975, *Report of the Task Group on Reference Man,* prepared by a Task Group of Committee 2, adopted by the Commission in October 1974, ICRP Publication 23, Pergamon Press, New York.

International Commission on Radiological Protection, 1977, *Recommendations of the International Commission on Radiological Protection,* ICRP Publication 26, Annals of the ICRP, *1*(2).

International Commission on Radiological Protection, 1978, *Radionuclide Release into the Environment: Assessment of Doses to Man,* a report of Committee 4 of the International Commission on Radiological Protection, adopted by the Commission in October 1978, ICRP Publication 29, Annals of the ICRP, *2*(2).

International Commission on Radiological Protection, 1979-1982, *Limits for Intakes of Radionuclides by Workers,* a report of Committee 2 of the International Commission on Radiological Protection, adopted by the Commission in July 1978, ICRP Publication 30, Part 1 (and Supplement), Part 2 (and Supplement), Part 3 (and Supplements A and B), and Index, Annals of the ICRP.

International Commission on Radiological Protection, 1983, *Cost-Benefit Analysis on the Optimization of Radiation Protection,* ICRP Publication 37, Annals of the ICRP, *10*(2/3).

International Commission on Radiological Protection, 1984, *A Compilation of the Major Concepts and Quantities in Use by ICRP,* ICRP Publication 42, Annals of the ICRP, *14*(4).

Kocher, D.C., 1983, *Dose-Rate Conversion Factors for External Exposure to Photons and Electrons,* Health Physics, *45*(3):665.

Kocher, D.C., and A.L. Sjoreen, 1985, *Dose-Rate Conversion Factors for External Exposure to Photon Emitters in Soil,* Health Physics, *28*:193-205.

McKone, T.E., 1986, *The Implicit Valuation of Environmental Cancer by U.S. Regulatory Agencies,* UCRL-93355, Lawrence Livermore National Laboratory, Livermore, Calif. (May).

Momeni, M.H., Y. Yuan, and A.J. Zielen, 1979, *The Uranium Dispersion and Dosimetry (UDAD) Code, Version IX,* NUREG/CR-0553; ANL/ES-72, prepared by Argonne National Laboratory, Argonne, Ill., for U.S. Nuclear Regulatory Commission (May).

National Council on Radiation Protection and Measurements, 1984, *Radiological Assessment: Predicting the Transport, Bioaccumulation, and Uptake by Man of Radionuclides Released to the Environment,* NCRP Report No. 76, Bethesda, Md. (March 15).

Neuder, S., 1986, personal communication from S. Neuder (U.S. Nuclear Regulatory Commission) to T.L. Gilbert (Argonne National Laboratory), Interagency Meeting on Source-to-Dose Assessment Methods for Residual Radioactivity, U.S. Environmental Protection Agency, Washington, D.C., November 5-6.

Oak Ridge National Laboratory, 1982, *Procedures Manual; Remedial Action Survey and Certification Activities,* ORNL-6000, Oak Ridge National Laboratory, Oak Ridge, Tenn.

Press, W.H., et al., 1986, *Numerical Recipes, The Art of Scientific Computing,* Cambridge University Press, New York.

Till, J.E., and H.R. Meyer (eds.), 1983, *Radiological Assessment; A Textbook on Environmental Dose Analysis,* NUREG/CR-3332; ORNL-5968, prepared by Oak Ridge National Laboratory, Oak Ridge, Tenn., for Division of Systems Integration, U.S. Nuclear Regulatory Commission, Washington, D.C. (Sept.).

U.S. Department of Energy, 1984, *Formerly Utilized Sites Remedial Action Program, Summary Protocol: Identification - Characterization - Designation - Remedial Action - Certification,* Division of Remedial Action Projects (April).

U.S. Department of Energy, 1986a, *Formerly Utilized Sites Remedial Action Program, Summary Protocol: Identification - Characterization - Designation - Remedial Action - Certification,* Office of Nuclear Energy, Division of Facility and Site Decommissioning Projects (Jan.).

U.S. Department of Energy, 1986b, *Formerly Utilized Sites Remedial Action Program, Designation/Elimination Protocol -- Supplement No. 1 to the FUSRAP Summary Protocol,* Office of Nuclear Energy, Division of Facility and Site Decommissioning Projects (Jan.).

U.S. Department of Energy, 1987, *U.S. Department of Energy Guidelines for Residual Radioactive Material at Formerly Utilized Sites Remedial Action Program and Remote Surplus Facilities Management Program Sites (Revision 2, March 1987)* [reproduced as Appendix A of this document].

U.S. Department of Energy, 1988a, *Internal Dose Conversion Factors for Calculation of Dose to the Public,* DOE/EH-0071, Assistant Secretary for Environment, Safety and Health (July).

U.S. Department of Energy, 1988b, *External Dose-Rate Conversion Factors for Calculation of Dose to the Public,* DOE/EH-0070, Assistant Secretary for Environment, Safety and Health (July).

U.S. Department of Energy, 1989, *Radiation Protection of the Public and the Environment,* Draft DOE Order 5400.xx.

U.S. Nuclear Regulatory Commission, 1974, *Numerical Guides for Design Objectives and Limiting Conditions for Operation to Meet the Criterion "As Low As Is Reasonably Achievable" for Radioactive Material in Light Water Cooled Nuclear Power Reactors Effluents,* Code of Federal Regulations 10 Part 50, Appendix I.

U.S. Nuclear Regulatory Commission, 1977, *Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50. Appendix I,* Regulatory Guide 1.109, Rev. 1, Washington, D.C.

U.S. Nuclear Regulatory Commission, 1981, *Draft Environmental Impact Statement on 10 CFR Part 61, "Licensing Requirements for Land Disposal of Radioactive Waste,"* NUREG-0782, Office of Nuclear Material Safety and Safeguards, Washington, D.C., 4 Vol. (Sept.).

U.S. Nuclear Regulatory Commission, 1982, *Safety Goals for Nuclear Power Plants: A Discussion Paper,* NUREG-0880, Office of Policy Evaluation, Washington, D.C.

Yu, C., W.A. Jester, and A.R. Jarrett, 1986, *Hydrogeologic Parameter Identification from Nuclear Tracer Breakthrough Curve Data for Waste Disposal in Porous Media,* Radioactive Waste Management and the Nuclear Fuel Cycle, 7(1):63-82.

APPENDIX A

DOE GUIDELINES FOR RESIDUAL RADIOACTIVE MATERIAL

APPENDIX A: DOE GUIDELINES FOR RESIDUAL RADIOACTIVE MATERIAL

[reproduced from U.S. Department of Energy, 1987, *U.S. Department of Energy Guidelines for Residual Radioactive Material at Formerly Utilized Sites Remedial Action Program and Remote Surplus Facilities Management Program Sites (Revision 2, March 1987)*]

A. INTRODUCTION

This document presents U.S. Department of Energy (DOE) radiological protection guidelines for cleanup of residual radioactive material and management of the resulting wastes and residues. It is applicable to sites identified by the Formerly Utilized Sites Remedial Action Program (FUSRAP) and remote sites identified by the Surplus Facilities Management Program (SFMP).* The topics covered are basic dose limits, guidelines and authorized limits for allowable levels of residual radioactive material, and requirements for control of the radioactive wastes and residues.

Protocols for identification, characterization, and designation of FUSRAP sites for remedial action; for implementation of the remedial action; and for certification of a FUSRAP site for release for unrestricted use are given in a separate document (U.S. Department of Energy 1986) and subsequent guidance. More detailed information on applications of the guidelines presented herein, including procedures for deriving site-specific guidelines for allowable levels of residual radioactive material from basic dose limits, is contained in "A Manual for Implementing Residual Radioactive Material Guidelines" (U.S. Department of Energy 1987), referred to herein as the "supplement".

"Residual radioactive material" is used in these guidelines to describe radioactive material derived from operations or sites over which DOE has authority. Guidelines or guidance to limit the levels of radioactive material and to protect the public and the environment are provided for (1) residual concentrations of radionuclides in soil,** (2) concentrations of airborne

---

*A remote SFMP site is one that is excess to DOE programmatic needs and is located outside a major operating DOE research and development or production area.

**"Soil" is defined herein as unconsolidated earth material, including rubble and debris that may be present in earth material.

radon decay products, (3) external gamma radiation levels, (4) surface contamination levels, and (5) radionuclide concentrations in air or water resulting from or associated with any of the above.

A "basic dose limit" is a prescribed standard from which limits for quantities that can be monitored and controlled are derived; it is specified in terms of the effective dose equivalent as defined by the International Commission on Radiological Protection (ICRP 1977, 1978). The basic dose limits are used for deriving guidelines for residual concentrations of radionuclides in soil. Guidelines for residual concentrations of thorium and radium in soil, concentrations of airborne radon decay products, allowable indoor external gamma radiation levels, and residual surface contamination concentrations are based on existing radiological protection standards (U.S. Environmental Protection Agency 1983; U.S. Nuclear Regulatory Commission 1982; and DOE Departmental Orders). Derived guidelines or limits based on the basic dose limits for those quantities are used only when the guidelines provided in the existing standards cited above are shown to be inappropriate.

A "guideline" for residual radioactive material is a level of radioactivity or radioactive material that is acceptable if use of the site is to be unrestricted. Guidelines for residual radioactive material presented herein are of two kinds: (1) generic, site-independent guidelines taken from existing radiation protection standards and (2) site-specific guidelines derived from basic dose limits using site-specific models and data. Generic guideline values are presented in this document. Procedures and data for deriving site-specific guideline values are given in the supplement. The basis for the guidelines is generally a presumed worst-case plausible-use scenario for the site.

An "authorized limit" is a level of residual radioactive material or radioactivity that must not be exceeded if the remedial action is to be considered completed and the site is to be released for unrestricted use. The authorized limits for a site will include (1) limits for each radionuclide or group of radionuclides, as appropriate, associated with residual radioactive material in soil or in surface contamination of structures and equipment, (2) limits for each radionuclide or group of radionuclides, as appropriate, in air or water, and, (3) where appropriate, a limit on external gamma radiation resulting from the residual material. Under normal circumstances, expected to occur at most sites, authorized limits for residual radioactive material or radioactivity are set equal to guideline values. Exceptional conditions for which authorized limits might differ from guideline values are specified in Sections D and F of this document. A site may be released for unrestricted use only if site conditions do not exceed the authorized limits or approved supplemental limits, as defined in Section F.1, at the time remedial action is completed. Restrictions and controls on use of the site must be established and enforced if site conditions exceed the approved limits, or if there is potential to exceed the basic dose limit if use of the site is not restricted (Section F.2). The applicable controls and restrictions are specified in Section E.

DOE policy requires that all exposures to radiation be limited to levels that are as low as reasonably achievable (ALARA). For sites to be released for unrestricted use, the intent is to reduce residual radioactive material to levels that are as far below authorized limits as reasonable considering technical, economic, and social factors. At sites where the residual material is not reduced to levels that permit release for unrestricted use, ALARA policy is implemented by establishing controls to reduce exposure to levels that are as low as reasonably achievable. Procedures for implementing ALARA policy are discussed in the supplement. ALARA policies, procedures, and actions shall be documented and filed as a permanent record upon completion of remedial action at a site.

B. BASIC DOSE LIMITS

The basic limit for the annual radiation dose received by an individual member of the general public is 100 mrem/yr. The internal committed effective dose equivalent, as defined in ICRP Publication 26 (ICRP 1977) and calculated by dosimetry models described in ICRP Publication 30 (ICRP 1978), plus the dose from penetrating radiation sources external to the body, shall be used for determining the dose. This dose shall be described as the "effective dose equivalent". Every effort shall be made to ensure that actual doses to the public are as far below the basic dose limit as is reasonably achievable.

Under unusual circumstances, it will be permissible to allow potential doses to exceed 100 mrem/yr where such exposures are based upon scenarios that do not persist for long periods and where the annual lifetime exposure to an individual from the subject residual radioactive material would be expected to be less than 100 mrem/yr. Examples of such situations include conditions that might exist at a site scheduled for remediation in the near future or a possible, but improbable, one-time scenario that might occur following remedial action. These levels should represent doses that are as low as reasonably achievable for the site. Further, no annual exposure should exceed 500 mrem.

C. GUIDELINES FOR RESIDUAL RADIOACTIVE MATERIAL

C.1 Residual Radionuclides in Soil

Residual concentrations of radionuclides in soil shall be specified as above-background concentrations averaged over an area of 100 m^2. Generic guidelines for thorium and radium are specified below. Guidelines for residual concentrations of other radionuclides shall be derived from the basic dose limits by means of an environmental pathway analysis using site-specific data where available. Procedures for these derivations are given in the supplement.

If the average concentration in any surface or below-surface area less than or equal to 25 m^2 exceeds the authorized limit or guideline by a factor of $(100/A)^{1/2}$, where A is the area of the elevated region in square meters, limits for "hot spots" shall also be applicable. Procedures for calculating these hot spot limits, which depend on the extent of the elevated local concentrations, are given in the supplement. In addition, every reasonable effort shall be made to remove any source of radionuclide that exceeds 30 times the appropriate limit for soil, irrespective of the average concentration in the soil.

Two types of guidelines are provided, generic and derived. The generic guidelines for residual concentrations of Ra-226, Ra-228, Th-230, and Th-232 are:

- 5 pCi/g, averaged over the first 15 cm of soil below the surface
- 15 pCi/g, averaged over 15-cm-thick layers of soil more than 15 cm below the surface

These guidelines take into account ingrowth of Ra-226 from Th-230 and of Ra-228 from Th-232, and assume secular equilibrium. If either Th-230 and Ra-226 or Th-232 and Ra-228 are both present, not in secular equilibrium, the appropriate guideline is applied as a limit to the radionuclide with the higher concentration. If other mixtures of radionuclides occur, the concentrations of individual radionuclides shall be reduced so that (1) the dose for the mixtures will not exceed the basic dose limit or (2) the sum of the ratios of the soil concentration of each radionuclide to the allowable limit for that radionuclide will not exceed 1 ("unity"). Explicit formulas for calculating residual concentration guidelines for mixtures are given in the supplement.

C.2 Airborne Radon Decay Products

Generic guidelines for concentrations of airborne radon decay products shall apply to existing occupied or habitable structures on private property that are intended for unrestricted use; structures that will be demolished or buried are excluded. The applicable generic guideline (40 CFR Part 192) is: In any occupied or habitable building, the objective of remedial action shall be, and a reasonable effort shall be made to achieve, an annual average (or equivalent) radon decay product concentration (including background) not to exceed 0.02 WL.* In any case, the radon decay product concentration (including background) shall not exceed 0.03 WL. Remedial actions by DOE are not required in order to comply with this guideline when there is reasonable assurance that residual radioactive material is not the cause.

---

*A working level (WL) is any combination of short-lived radon decay products in one liter of air that will result in the ultimate emission of 1.3×10^5 MeV of potential alpha energy.

C.3 External Gamma Radiation

The average level of gamma radiation inside a building or habitable structure on a site to be released for unrestricted use shall not exceed the background level by more than 20 μR/h and shall comply with the basic dose limit when an appropriate-use scenario is considered. This requirement shall not necessarily apply to structures scheduled for demolition or to buried foundations. External gamma radiation levels on open lands shall also comply with the basic dose limit, considering an appropriate-use scenario for the area.

C.4 Surface Contamination

The generic surface contamination guidelines provided in Table 1 are applicable to existing structures and equipment. These guidelines are adapted from standards of the U.S. Nuclear Regulatory Commission (NRC 1982)* and will be applied in a manner that provides a level of protection consistent with the Commission's guidance. These limits apply to both interior and exterior surfaces. They are not directly intended for use on structures to be demolished or buried, but should be applied to equipment or building components that are potentially salvageable or recoverable scrap. If a building is demolished, the guidelines in Section C.1 are applicable to the resulting contamination in the ground.

C.5 Residual Radionuclides in Air and Water

Residual concentrations of radionuclides in air and water shall be controlled to levels required by DOE Environmental Protection Guidance and Orders, specifically DOE Order 5480.1A and subsequent guidance. Other Federal and/or state standards shall apply when they are determined to be appropriate.

D. AUTHORIZED LIMITS FOR RESIDUAL RADIOACTIVE MATERIAL

Authorized limits shall be established to (1) ensure that, as a minimum, the basic dose limits specified in Section B will not be exceeded under the worst-case plausible-use scenario consistent with the procedures and guidance provided or (2) be consistent with applicable generic guidelines, where such guidelines are provided. The authorized limits for each site and its vicinity properties shall be set equal to the generic or derived guidelines except where it can be clearly established on the basis of site-specific data -- including health, safety, and socioeconomic considerations -- that the guidelines are not appropriate for use at the specific site. Consideration should also be given to ensure that the limits comply with or provide a level of protection equivalent to other appropriate limits and guidelines (i.e., state or

---

*These guidelines are functionally equivalent to Section 4 -- Decontamination for Release for Unrestricted Use -- of NRC Regulatory Guide 1.86 (U.S. Atomic Energy Commission 1974), but they are applicable to non-reactor facilities.

TABLE 1 SURFACE CONTAMINATION GUIDELINES

	Allowable Total Residual Surface Contamination (dpm/100 cm^2)[a]		
Radionuclides[b]	Average[c,d]	Maximum[d,e]	Removable[d,f]
Transuranics, Ra-226, Ra-228, Th-230, Th-228, Pa-231, Ac-227, I-125, I-129	100	300	20
Th-Natural, Th-232, Sr-90, Ra-223, Ra-224, U-232, I-126, I-131, I-133	1,000	3,000	200
U-Natural, U-235, U-238, and associated decay products	5,000 α	15,000 α	1,000 α
Beta-gamma emitters (radionuclides with decay modes other than alpha emission or spontaneous fission) except Sr-90 and others noted above	5,000 β-γ	15,000 β-γ	1,000 β-γ

[a] As used in this table, dpm (disintegrations per minute) means the rate of emission by radioactive material as determined by correcting the counts per minute measured by an appropriate detector for background, efficiency, and geometric factors associated with the instrumentation.

[b] Where surface contamination by both alpha- and beta-gamma-emitting radionuclides exists, the limits established for alpha- and beta-gamma-emitting radionuclides should apply independently.

[c] Measurements of average contamination should not be averaged over an area of more than 1 m^2. For objects of less surface area, the average should be derived for each such object.

[d] The average and maximum dose rates associated with surface contamination resulting from beta-gamma emitters should not exceed 0.2 mrad/h and 1.0 mrad/h, respectively, at 1 cm.

[e] The maximum contamination level applies to an area of not more than 100 cm^2.

[f] The amount of removable radioactive material per 100 cm^2 of surface area should be determined by wiping that area with dry filter or soft absorbent paper, applying moderate pressure, and measuring the amount of radioactive material on the wipe with an appropriate instrument of known efficiency. When removable contamination on objects of surface area less than 100 cm^2 is determined, the activity per unit area should be based on the actual area and the entire surface should be wiped. The numbers in this column are maximum amounts.

other Federal). Documentation supporting such a decision should be similar to that required for supplemental limits and exceptions (Section F), but should be generally more detailed because the documentation covers the entire site.

Remedial action shall not be considered complete unless the residual radioactive material levels comply with the authorized limits. The only exception to this requirement will be for those special situations where the supplemental limits or exceptions are applicable and approved as specified in Section F. However, the use of supplemental limits and exceptions should be considered only if it is clearly demonstrated that it is not reasonable to decontaminate the area to the authorized limit or guideline value. The authorized limits are developed through the project offices in the field and are approved by the headquarters program office.

E. CONTROL OF RESIDUAL RADIOACTIVE MATERIAL AT FUSRAP AND REMOTE SFMP SITES

Residual radioactive material above the guidelines at FUSRAP and remote SFMP sites must be managed in accordance with applicable DOE Orders. The DOE Order 5480.1A and subsequent guidance or superceding Orders require compliance with applicable Federal and state environmental protection standards.

The operational and control requirements specified in the following DOE Orders shall apply to interim storage, interim management, and long-term management.

a. 5000.3, Unusual Occurrence Reporting System

b. 5440.1C, Implementation of the National Environmental Policy Act

c. 5480.1A, Environmental Protection, Safety, and Health Protection Program for DOE Operations, as revised by DOE 5480.1 change orders and the 5 August 1985 memorandum from Vaughan to Distribution

d. 5480.2, Hazardous and Radioactive Mixed Waste Management

e. 5480.4, Environmental Protection, Safety, and Health Protection Standards

f. 5482.1A, Environmental, Safety, and Health Appraisal Program

g. 5483.1A, Occupational Safety and Health Program for Government-Owned Contractor-Operated Facilities

h. 5484.1, Environmental Protection, Safety, and Health Protection Information Reporting Requirements

i. 5820.2, Radioactive Waste Management

E.1 Interim Storage

a. Control and stabilization features shall be designed to ensure, to the extent reasonably achievable, an effective life of 50 years and, in any case, at least 25 years.

b. Above-background Rn-222 concentrations in the atmosphere above facility surfaces or openings shall not exceed (1) 100 pCi/L at any given point, (2) an annual average concentration of 30 pCi/L over the facility site, and (3) an annual average concentration of 3 pCi/L at or above any location outside the facility site (DOE Order 5480.1A, Attachment XI-1).

c. Concentrations of radionuclides in the groundwater or quantities of residual radioactive material shall not exceed existing Federal or state standards.

d. Access to a site shall be controlled and misuse of on-site material contaminated by residual radioactive material shall be prevented through appropriate administrative controls and physical barriers -- active and passive controls as described by the U.S. Environmental Protection Agency (1983--p. 595). These control features should be designed to ensure, to the extent reasonable, an effective life of at least 25 years. The Federal government shall have title to the property or shall have a long-term lease for exclusive use.

E.2 Interim Management

a. A site may be released under interim management when the residual radioactive material exceeds guideline values if the residual radioactive material is in inaccessible locations and would be unreasonably costly to remove, provided that administrative controls are established to ensure that no member of the public shall receive a radiation dose exceeding the basic dose limit.

b. The administrative controls, as approved by DOE, shall include but not be limited to periodic monitoring as appropriate, appropriate shielding, physical barriers to prevent access, and appropriate radiological safety measures during maintenance, renovation, demolition, or other activities that might disturb the residual radioactive material or cause it to migrate.

c. The owner of the site or appropriate Federal, state, or local authorities shall be responsible for enforcing the administrative controls.

E.3 Long-Term Management

Uranium, Thorium, and Their Decay Products

a. Control and stabilization features shall be designed to ensure, to the extent reasonably achievable, an effective life of 1,000 years and, in any case, at least 200 years.

b. Control and stabilization features shall be designed to ensure that Rn-222 emanation to the atmosphere from the wastes shall not (1) exceed an annual average release rate of 20 $pCi/m^2/s$ and (2) increase the annual average Rn-222 concentration at or above any location outside the boundary of the contaminated area by more than 0.5 pCi/L. Field verification of emanation rates is not required.

c. Prior to placement of any potentially biodegradable contaminated wastes in a long-term management facility, such wastes shall be properly conditioned to ensure that (1) the generation and escape of biogenic gases will not cause the requirement in paragraph b. of this section (E.3) to be exceeded and (2) biodegradation within the facility will not result in premature structural failure in violation of the requirements in paragraph a. of this section (E.3).

d. Groundwater shall be protected in accordance with appropriate Departmental Orders and Federal and state standards, as applicable to FUSRAP and remote SFMP sites.

e. Access to a site should be controlled and misuse of on-site material contaminated by residual radioactivity should be prevented through appropriate administrative controls and physical barriers -- active and passive controls as described by the U.S. Environmental Protection Agency (1983--p. 595). These controls should be designed to be effective to the extent reasonable for at least 200 years. The Federal government shall have title to the property.

Other Radionuclides

f. Long-term management of other radionuclides shall be in accordance with Chapters 2, 3, and 5 of DOE Order 5820.2, as applicable.

F. SUPPLEMENTAL LIMITS AND EXCEPTIONS

If special site-specific circumstances indicate that the guidelines or authorized limits established for a given site are not appropriate for a portion of that site or for a vicinity property, then the field office may request that supplemental limits or an exception be applied. In either case, the field office must justify that the subject guidelines or authorized limits are not appropriate and that the alternative action will provide adequate protection, giving due consideration to health and safety, the environment, and costs. The field office shall obtain approval for specific supplemental limits or exceptions from headquarters as specified in Section D of these guidelines and shall provide to headquarters those materials required for the justification as specified in this section (F) and in the FUSRAP and SFMP protocols and subsequent guidance documents. The field office shall also be responsible for coordination with the state or local government of the limits or exceptions and associated restrictions as appropriate. In the case of exceptions, the field office shall also work with the state and/or local governments to ensure that restrictions or conditions of release are adequate and mechanisms are in place for their enforcement.

F.1 Supplemental Limits

The supplemental limits must achieve the basic dose limits set forth in this guideline document for both current and potential unrestricted uses of a site and/or vicinity property. Supplemental limits may be applied to a vicinity property or a portion of a site if, on the basis of a site-specific analysis, it is determined that (1) certain aspects of the vicinity property or portion of the site were not considered in the development of the established authorized limits and associated guidelines for that vicinity property or site and, (2) as a result of these unique characteristics, the established limits or guidelines either do not provide adequate protection or are unnecessarily restrictive and costly.

F.2 Exceptions

Exceptions to the authorized limits defined for unrestricted use of a site or vicinity property may be applied to a vicinity property or a portion of a site when it is established that the authorized limits cannot be achieved and restrictions on use of the vicinity property or portion of the site are necessary to provide adequate protection of the public and the environment. The field office must clearly demonstrate that the exception is necessary and that the restrictions will provide the necessary degree of protection and will comply with the requirements for control of residual radioactive material as set forth in Section E of these guidelines.

F.3 Justification for Supplemental Limits and Exceptions

Supplemental limits and exceptions must be justified by the field office on a case-by-case basis using site-specific data. Every effort should be made to minimize use of the supplemental limits and exceptions. Examples of specific situations that warrant use of the supplemental standards and exceptions are:

a. Where remedial action would pose a clear and present risk of injury to workers or members of the general public, notwithstanding reasonable measures to avoid or reduce risk.

b. Where remedial action -- even after all reasonable mitigative measures have been taken -- would produce environmental harm that is clearly excessive compared to the health benefits to persons living on or near affected sites, now or in the future. A clear excess of environmental harm is harm that is long-term, manifest, and grossly disproportionate to health benefits that may reasonably be anticipated.

c. Where it is clear that the scenarios or assumptions used to establish the authorized limits do not, under plausible current or future conditions, apply to the property or portion of the site identified and where more appropriate scenarios or assumptions indicate that other limits are applicable or necessary for protection of the public and the environment.

d. Where the cost of remedial action for contaminated soil is unreasonably high relative to long-term benefits and where the residual radioactive material does not pose a clear present or future risk after taking necessary control measures. The likelihood that buildings will be erected or that people will spend long periods of time at such a site should be considered in evaluating this risk. Remedial action will generally not be necessary where only minor quantities of residual radioactive material are involved or where residual radioactive material occurs in an inaccessible location at which site-specific factors limit their hazard and from which they are costly or difficult to remove. Examples include residual radioactive material under hard-surface public roads and sidewalks, around public sewer lines, or in fence-post foundations. A site-specific analysis must be provided to establish that it would not cause an individual to receive a radiation dose in excess of the basic dose limits stated in Section B, and a statement specifying the level of residual radioactive material must be included in the appropriate state and local records.

e. Where there is no feasible remedial action.

G. SOURCES

Limit or Guideline	Source
Basic Dose Limits	
Dosimetry model and dose limits	International Commission on Radiological Protection (1977, 1978)
Generic Guidelines for Residual Radioactivity	
Residual concentrations of radium and thorium in soil	40 CFR Part 192
Airborne radon decay products	40 CFR Part 192
External gamma radiation	40 CFR Part 192
Surface contamination	Adapted from U.S. Nuclear Regulatory Commission (1982)
Control of Radioactive Wastes and Residues	
Interim storage	DOE Order 5480.1A and subsequent guidance
Long-term management	DOE Order 5480.1A and subsequent guidance; 40 CFR Part 192; DOE Order 5820.2

H. REFERENCES

International Commission on Radiological Protection, 1977. Recommendations of the International Commission on Radiological Protection (Adopted January 17, 1977). ICRP Publication 26. Pergamon Press, Oxford. [As modified by "Statement from the 1978 Stockholm Meeting of the ICRP." Annals of the ICRP, Vol. 2, No. 1, 1978.]

International Commission on Radiological Protection, 1978. Limits for Intakes of Radionuclides by Workers. A Report of Committee 2 of the International Commission on Radiological Protection. Adopted by the Commission in July 1978. ICRP Publication 30. Part 1 (and Supplement), Part 2 (and Supplement), Part 3 (and Supplements A and B), and Index. Pergamon Press, Oxford.

U.S. Atomic Energy Commission, 1974. Regulatory Guide 1.86, Termination of Operating Licenses for Nuclear Reactors. June 1974.

U.S. Department of Energy, 1986. Formerly Utilized Sites Remedial Action Program. Summary Protocol: Identification - Characterization - Designation - Remedial Action - Certification. Office of Nuclear Energy, Office of Terminal Waste Disposal and Remedial Action, Division of Remedial Action Projects. January 1986.

U.S. Department of Energy, 1987. Supplement to U.S. Department of Energy Guidelines for Residual Radioactive Material at Formerly Utilized Sites Remedial Action Program and Remote Surplus Facilities Management Program Sites. A Manual for Implementing Residual Radioactive Material Guidelines. Prepared by Argonne National Laboratory, Los Alamos National Laboratory, Oak Ridge National Laboratory, and Pacific Northwest Laboratory for the U.S. Department of Energy. [In press.]

U.S. Environmental Protection Agency, 1983. Standards for Remedial Actions at Inactive Uranium Processing Sites; Final Rule (40 CFR Part 192). Federal Register 48(3):590-604 (January 5, 1983).

U.S. Nuclear Regulatory Commission, 1982. Guidelines for Decontamination of Facilities and Equipment Prior to Release for Unrestricted Use or Termination of Licenses for Byproduct, Source, or Special Nuclear Material. Division of Fuel Cycle and Material Safety, Washington, D.C. July 1982.

APPENDIX B

EXTERNAL GROUND RADIATION PATHWAY FACTORS

APPENDIX B: EXTERNAL GROUND RADIATION PATHWAY FACTORS

Models, formulas, and data for calculating pathway factors for the external ground radiation pathway are presented in this appendix. Exposure to external radiation occurs primarily as a result of radiation emanating from radionuclides in a contaminated zone. Exposure can also occur from radiation emanating from radionuclides in the air, water, or ground surface that have been transported from their original location in the contaminated zone. In general, the radiation dose resulting from these secondary sources is negligibly small compared with the dose resulting from direct exposure to the primary source. Therefore, these secondary sources are not taken into account in deriving soil concentration guidelines.

The contribution to the effective dose equivalent from the external ground radiation pathway for the i^{th} principal radionuclide at time t following the radiological survey is given by the dose/source ratio $DSR_{i1}(t)$. This ratio may be expressed as a sum of products of dose conversion factors, environmental transport factors, and source factors (see Equation 3.8). Tables of dose conversion factors for the external ground radiation pathway are given in Section B.1. Models and formulas for calculating the environmental transport factors are given in Section B.2. Formulas for calculating the source factors are given in Section 3.2.3 and Appendix F.

B.1 DOSE CONVERSION FACTORS

The dose conversion factor DCF_{i1} for the external ground radiation pathway is the annual effective dose equivalent received from exposure to radiation from the i^{th} principal radionuclide present at unit concentration in a uniformly contaminated zone of infinite depth and lateral extent. The radiation field is assumed to be equal to the radiation level at a distance of 1 m above the ground surface. The DCFs for surface contamination (infinite thinness and lateral extent) were taken from a DOE report (DOE 1988). The DCFs for volume contamination (infinite depth and lateral extent) were calculated using the methods of Kocher and Sjoreen (1985). The results are given in Table B.1. In the RESRAD code, the volume contamination dose conversion factors are used. Values for densities other than 1.0 and 1.8 g/cm^3 are obtained by linear interpolation or extrapolation of log(DCF).

B.2 ENVIRONMENTAL TRANSPORT FACTORS

The environmental transport factor ETF_{i1} for the external ground radiation pathway is the ratio of the effective dose equivalent for the actual source to the effective dose equivalent for the standard source, multiplied by an occupancy and shielding factor. The standard source is a contaminated zone of infinite depth and lateral extent with no cover. The actual source is approximated by a cylindrical contaminated zone of radius R and depth T located at distance C_d below the ground surface (see Figure 2.1).

Calculations by Napier et al. (1984) have indicated that the dependence on source size can be approximated reasonably well by an area factor. The ETF for the external ground radiation pathway is expressed as the product

$$ETF_{i1}(t) = \rho_b^{(cz)} \times FO_1 \times FS_1 \times FA_1 \times FD_{i1}(t) \times FC_{i1}(t) \qquad (B.1)$$

TABLE B.1 Effective Dose Equivalent Conversion Factors (DCF_{i1}) for External Gamma Radiation from Contaminated Ground

Radionuclide i	Surface Factors[a] (mrem/yr)/(pCi/cm^2)	Volume Factors[a] (mrem/yr)/(pCi/cm^3)	
		ρ_b = 1.0 g/cm^3	ρ_b = 1.8 g/cm^3
H-3	0	0	0
C-14	0	0	0
Fe-55	2.21×10^{-4}	0	0
Co-60	2.27	2.27×10^{1}	1.25×10^{1}
Ni-59	4.16×10^{-4}	0	0
Ni-63	0	0	0
Sr-90+D	0	0	0
Nb-94	1.59	1.42×10^{1}	7.83
Tc-99	6.26×10^{-7}	1.68×10^{-6}	9.32×10^{-7}
I-129	2.20×10^{-2}	3.24×10^{-2}	1.75×10^{-2}
Cs-135	0	0	0
Cs-137+D	6.11×10^{-1}	5.03	2.77
Eu-152	1.11	9.91	5.47
Eu-154	1.21	1.10×10^{1}	6.06
Pb-210+D	3.00×10^{-3}	4.87×10^{-3}	2.31×10^{-3}
Ra-226+D	1.69	1.55×10^{1}	8.56
Ra-228+D	9.10×10^{-1}	8.18	4.51
Ac-227+D	4.70×10^{-1}	2.76	1.52
Th-228+D	1.45	1.33×10^{1}	7.36
Th-229+D	3.44×10^{-1}	2.20	1.21
Th-230	9.07×10^{-4}	2.11×10^{-3}	1.03×10^{-3}
Th-232	6.66×10^{-4}	1.35×10^{-3}	6.04×10^{-4}
Pa-231	3.58×10^{-2}	2.21×10^{-1}	1.21×10^{-1}
U-232	1.03×10^{-3}	2.19×10^{-3}	1.01×10^{-3}
U-233	5.00×10^{-4}	1.40×10^{-3}	7.12×10^{-4}
U-234	8.07×10^{-4}	1.58×10^{-3}	6.97×10^{-4}
U-235+D	1.90×10^{-1}	8.94×10^{-1}	4.90×10^{-1}
U-236	7.33×10^{-4}	1.35×10^{-3}	5.80×10^{-4}
U-238+D	2.46×10^{-2}	1.27×10^{-1}	6.97×10^{-2}

TABLE B.1 (Cont'd)

Radionuclide i	Surface Factors[a] (mrem/yr)/(pCi/cm^2)	Volume Factors[a] (mrem/yr)/(pCi/cm^3) ρ_b = 1.0 g/cm^3	Volume Factors[a] (mrem/yr)/(pCi/cm^3) ρ_b = 1.8 g/cm^3
Np-237+D	2.68×10^{-1}	1.61	8.90×10^{-1}
Pu-238	8.58×10^{-4}	1.56×10^{-3}	6.65×10^{-4}
Pu-239	3.78×10^{-4}	8.14×10^{-4}	3.76×10^{-4}
Pu-240	8.20×10^{-4}	1.48×10^{-3}	6.35×10^{-4}
Pu-241	0	0	0
Pu-242	6.82×10^{-4}	1.24×10^{-3}	5.29×10^{-4}
Am-241	2.99×10^{-2}	4.79×10^{-2}	2.58×10^{-2}
Am-243+D	2.58×10^{-1}	1.08	5.95×10^{-1}
Cm-243	1.46×10^{-1}	7.26×10^{-1}	4.01×10^{-1}
Cm-244	8.29×10^{-4}	1.51×10^{-3}	6.67×10^{-4}

[a] Surface factors represent infinite thinness; volume factors represent infinite depth.

where*

$\rho_b^{(cz)}$ = bulk density of soil material in the contaminated zone (1.6 g/cm^3),

FO_1 = occupancy and shielding factor (0.6, dimensionless),

FS_1 = shape factor (1.0, dimensionless),

FA_1 = area factor (dimensionless),

$FD_{i1}[\rho_b^{(cz)}, T(t)]$ = depth factor for radionuclide i (dimensionless),

$FC_{i1}[\rho_b^{(cv)}, C_d(t)]$ = cover factor for radionuclide i at time t (dimensionless),

---

*Numbers preceding the units in parentheses are default values, given only for parameters for which site-specific input values can be provided by the user. If the units are not preceded by a number, the parameter is a derived quantity.

$T(t)$ = thickness of contaminated zone at time t (m),

$\rho_b^{(cv)}$ = bulk density of cover material (1.6 g/cm^3), and

$C_d(t)$ = thickness of cover at time t (m).

The area factor is calculated in the RESRAD code by linear interpolation using the contaminated zone area (as input parameter) and the values listed in Table B.2. The area factor calculated is for a circular-area-equivalent contaminated zone. A shape factor (input parameter) is used to correct for the noncircular-shape area factor. The shape factor for a circular contaminated area is 1.0. For an irregularly shaped contaminated area, the shape factor may be obtained by enclosing the irregularly shaped contaminated area in a circle, multiplying the area factor of each annulus by the fraction of the annulus area that is contaminated, summing the products, and dividing by the area factor of a circular-area-equivalent contaminated zone. The area

TABLE B.2 Area Factors for External Gamma Radiation from Contaminated Ground[a]

Contaminated Area (m^2)	Radius[b] (m)	Area Factor,[c] FA_1
1	0.56	0.016
25	2.8	0.4
100	5.6	0.55
500	13	0.8
1,200	20	1.0

[a] Napier et al. (1984).

[b] Radius for a circular contaminated area.

[c] Intermediate values may be obtained by linear interpolation.

factor of an annulus is the area factor from an annular zone bounded by the radii tabulated in Table B.2. For example, the area factor is 0.2 for the annulus with inner and outer radii of 13 m and 20 m, respectively; 0.25 for the annulus with radii of 5.6 and 13 m; and so forth.

B.2.1. Depth Factor

The depth factor FD_{il} for a specified contaminated zone thickness is obtained by interpolation or extrapolation from Table B.3. It is assumed that the depth factor can be approximated by the equation

$$FD_{il}[\rho_b^{(cz)}, T(t)] = 1 - \exp[-k_i \rho_b^{(cz)} T(t)] \tag{B.2}$$

where

k_i = an empirical constant (m^2/kg), and

$\rho_b^{(cz)}$ = bulk density of soil material in the contaminated zone (1,600 kg/m^3),

and the other parameters are as defined for Equation B.1. Note that the units for $\rho_b^{(cz)}$ in Equations B.1 and B.2 are different. The empirical constant k_i for radionuclide i is a function of soil material density; it is determined from the tabulated values of the quantity $FD_{il}[\rho_b^{(cz)}, T(t)]$ in Table B.3. The equation used to calculate k_i for $\rho_b^{(cz)}$ equals 1,000 and 1,800 kg/m^3 is

$$k_i(\rho_b^{(cz)}) = -\log_e[1 - FD_{il}(\rho_b^{(cz)}, 0.15)]/(0.15\rho_b^{(cz)}) \tag{B.3}$$

TABLE B.3 Depth Factors (FD_{i1}) for External Gamma Radiation from Contaminated Ground as a Function of Thickness (T) of a Contaminated Layer Exposed at the Ground Surface and Soil Density (ρ_b) for Principal and Associated Radionuclides

Radionuclide i^a	$\rho_b = 1.0\ g/cm^3$			$\rho_b = 1.8\ g/cm^3$		
	T = 0.15 m	T = 0.5 m	T = 1.0 m	T = 0.15 m	T = 0.5 m	T = 1.0 m
H-3	1.0	1.0	1.0	1.0	1.0	1.0
C-14	9.6×10^{-1}	1.0	1.0	1.0	1.0	1.0
Fe-55	1.0	1.0	1.0	1.0	1.0	1.0
Co-60	6.8×10^{-1}	1.0	1.0	8.6×10^{-1}	1.0	1.0
Ni-59	1.0	1.0	1.0	1.0	1.0	1.0
Ni-63	1.0	1.0	1.0	1.0	1.0	1.0
Sr-90+D	8.1×10^{-1}	9.8×10^{-1}	1.0	9.0×10^{-1}	1.0	1.0
Nb-94	7.2×10^{-1}	1.0	1.0	8.9×10^{-1}	1.0	1.0
Tc-99	9.6×10^{-1}	1.0	1.0	1.0	1.0	1.0
I-129	1.0	1.0	1.0	1.0	1.0	1.0
Cs-135	9.6×10^{-1}	1.0	1.0	1.0	1.0	1.0
Cs-137+D	7.2×10^{-1}	9.8×10^{-1}	1.0	9.1×10^{-1}	1.0	1.0
Eu-152	6.8×10^{-1}	9.6×10^{-1}	1.0	8.5×10^{-1}	1.0	1.0
Eu-154	6.9×10^{-1}	9.8×10^{-1}	1.0	8.6×10^{-1}	1.0	1.0
Pb-210+D	8.8×10^{-1}	1.0	1.0	9.7×10^{-1}	1.0	1.0
Ra-226+D	6.3×10^{-1}	9.2×10^{-1}	1.0	8.5×10^{-1}	1.0	1.0
Ra-228+D	6.8×10^{-1}	9.7×10^{-1}	1.0	8.5×10^{-1}	1.0	1.0
Ac-227+D	7.9×10^{-1}	9.7×10^{-1}	1.0	9.1×10^{-1}	1.0	1.0
Th-228+D	6.1×10^{-1}	9.4×10^{-1}	1.0	7.5×10^{-1}	1.0	1.0
Th-229+D	6.5×10^{-1}	9.5×10^{-1}	1.0	8.5×10^{-1}	9.9×10^{-1}	1.0

TABLE B.3 (Cont'd)

Radionuclide i^a	$\rho_b = 1.0$ g/cm^3			$\rho_b = 1.8$ g/cm^3		
	T = 0.15 m	T = 0.5 m	T = 1.0 m	T = 0.15 m	T = 0.5 m	T = 1.0 m
Th-230+D	9.3×10^{-1}	1.0	1.0	1.0	1.0	1.0
Th-232+D	9.5×10^{-1}	1.0	1.0	1.0	1.0	1.0
Pa-231	7.9×10^{-1}	1.0	1.0	9.2×10^{-1}	1.0	1.0
U-232	8.8×10^{-1}	1.0	1.0	1.0	1.0	1.0
U-233	9.6×10^{-1}	1.0	1.0	1.0	1.0	1.0
U-234	9.0×10^{-1}	1.0	1.0	1.0	1.0	1.0
U-235+D	8.7×10^{-1}	1.0	1.0	1.0	1.0	1.0
U-236	9.4×10^{-1}	1.0	1.0	1.0	1.0	1.0
U-238+D	7.8×10^{-1}	1.0	1.0	8.8×10^{-1}	1.0	1.0
Np-237+D	8.2×10^{-1}	1.0	1.0	9.3×10^{-1}	1.0	1.0
Pu-238	9.3×10^{-1}	1.0	1.0	1.0	1.0	1.0
Pu-239	9.2×10^{-1}	1.0	1.0	1.0	1.0	1.0
Pu-240	9.2×10^{-1}	1.0	1.0	1.0	1.0	1.0
Pu-241+D	9.4×10^{-1}	1.0	1.0	1.0	1.0	1.0
Pu-242	9.6×10^{-1}	1.0	1.0	1.0	1.0	1.0
Am-241	9.4×10^{-1}	1.0	1.0	1.0	1.0	1.0
Am-243+D	9.0×10^{-1}	1.0	1.0	9.7×10^{-1}	1.0	1.0
Cm-243	8.7×10^{-1}	1.0	1.0	9.6×10^{-1}	1.0	1.0
Cm-244	9.6×10^{-1}	1.0	1.0	1.0	1.0	1.0

[a] A "+D" signifies that the contributions from the associated decay chain are included; radionuclides in the associated decay chain are identified in Table 3.1.

Linear interpolation is then used to obtain $k_i(\rho_b^{(cz)})$ for the actual soil density. If $FD_{i1}(\rho_b^{(cz)}, 0.15) = 1.0$, then the tabulated value is replaced by $FD_{i1}(\rho_b^{(cz)}, 0.15) = 0.995$.

The time dependence of the contaminated zone thickness is given by

$$\begin{aligned} T(t) &= T(0), & 0 \le t \le t_c \\ &= T(0) - v^{(cz)}(t - t_c), & t_c < t \end{aligned} \tag{B.4}$$

where

$T(t)$ = thickness of the contaminated zone at time t (m),

$T(0)$ = initial thickness of the contaminated zone (1 m),

$t_c = C_d(0)/v^{(cv)}$ = time for the cover to be removed by erosion (yr),

$v^{(cv)}$ = erosion rate of cover material (0.001 m/yr), and

$v^{(cz)}$ = erosion rate of contaminated zone (0 m/yr).

Erosion rates for both the cover and the contaminated zone may be estimated by means of the Universal Soil Loss Equation (USLE), an empirical model that has been developed for predicting the rate of soil loss by sheet and rill erosion. If sufficient site-specific data are available, a site-specific erosion rate can be calculated. Details are discussed in Wischmeier and Smith (1978) and Foster (1979). Estimates based on the range of erosion rates for typical sites in humid areas east of the Mississippi River (based on model site calculations for locations in New York, New Jersey, Ohio, and Missouri) may also be used (Knight 1983). For a site with a 2% slope, these model calculations predict a range of 8×10^{-7} to 3×10^{-6} m/yr for natural succession vegetation, 1×10^{-5} to 6×10^{-5} m/yr for permanent pasture, and 9×10^{-5} to 6×10^{-4} m/yr for row-crop agriculture. The rate increases by a factor of about 3 for a 5% slope, 7 for a 10% slope, and 15 for a 15% slope. If these generic values are used for a farm/garden scenario, in which the dose contribution from food ingestion pathways is expected to be significant, an erosion rate of 0.06 cm/yr should be assumed for a site with a 2% slope. This would lead to erosion of 0.6 m of soil in 1,000 years. A proportionately higher erosion rate must be used if the slope exceeds 2%. An erosion rate of 6×10^{-5} m/yr, leading to erosion of 0.006 m of soil in 1,000 years, may be used for a site with a 2% slope if it can be reasonably shown that the farm/garden scenario is unreasonable, e.g., because the site is, and will likely continue to be, unsuitable for agricultural use.

The erosion rates are more difficult to estimate for arid sites in the West than for humid sites in the East. Although water erosion is generally more important than wind erosion, the latter can also be significant. Water erosion in the West is more difficult to estimate because it is likely to be due to infrequent heavy rainfalls for which the empirical constants used in the USLE may not be applicable. Long-term erosion rates are generally less for sites in arid locations than for sites in humid locations; hence, values estimated in the manner described above for humid sites can be used because they can be expected to provide conservative values. The cover factors and the formula for time dependence of the cover depth are described in Section B.2.2.

B.2.2 Cover Factor

The cover factor for radionuclide i in the contaminated zone with a cover thickness of $C_d(t)$ and a cover density of $\rho_b^{(cv)}$ is approximated by the formula

$$FC_{i1}[\rho_b^{(cv)}, C_d(t)] = \exp[-k_i \rho_b^{(cv)} C_d(t)] \tag{B.5}$$

where

k_i = empirical constant (m^2/kg),
$\rho_b^{(cv)}$ = bulk density of the cover material (1,600 kg/m^3), and
$C_d(t)$ = cover depth at time t (m).

The empirical constant k_i for radionuclide i is a function of soil material density. Linear interpolation is used to obtain $k_i(\rho_b^{(cv)})$ for the actual cover density (see Section B.2.1).

The time dependence of the cover depth is given by

$$\begin{aligned} C_d(t) &= C_d(0) - v^{(cv)}t, & 0 \le t < t_c \\ &= 0, & t_c \le t \end{aligned} \tag{B.6}$$

where

$C_d(t)$ = cover depth at time t (m),
$C_d(0)$ = initial cover depth (0 m),
$v^{(cv)}$ = erosion rate of the cover material (0.001 m/yr), and
t_c = time for the cover to be removed by erosion (yr).

B.3 REFERENCES

Foster, G.R., 1979, *Sediment Yield from Farm Fields: The Universal Soil Loss Equation and Onfarm 208 Plan Implementation,* in Universal Soil Loss Equation: Past, Present, and Future, Soil Science Society of America Special Publication No. 8, pp. 17-24.

Knight, M.J., 1983, *The Effect of Soil Erosion on the Long-Term Stability of FUSRAP Near-Surface Waste-Burial Sites,* ANL/EIS-18, Revised, prepared by Division of Environmental Impact Studies, Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations Office (April).

Kocher, D.C., and A.L. Sjoreen, 1985, *Dose-Rate Conversion Factors for External Exposure to Photon Emitters in Soil,* Health Physics, *28*:193-205.

Napier, B.A., et al., 1984, *Intruder Dose Pathway Analysis for the Onsite Disposal of Radioactive Wastes: The ONSITE/MAXI1 Computer Program,* NUREG/CR-3620; PNL-4054, prepared by Pacific Northwest Laboratory, Richland, Wash., for Division of Waste Management, U.S. Nuclear Regulatory Commission, Washington, D.C.

U.S. Department of Energy, 1988, *External Dose-Rate Conversion Factors for Calculation of Dose to the Public,* DOE/EH-0070, Assistant Secretary for Environment, Safety and Health (July).

Wischmeier, W.H., and D.D. Smith, 1978, *Predicting Rainfall Erosion Losses - A Guide to Conservation Planning,* Agricultural Handbook No. 537, prepared by Science and Education Administration, Washington, D.C., for U.S. Department of Agriculture.

APPENDIX C

INHALATION PATHWAY FACTORS

APPENDIX C: INHALATION PATHWAY FACTORS

Models, formulas, and data for calculating pathway factors for the inhalation pathway are presented in this appendix. Inhalation exposure can occur from inhalation of dust, radon and radon decay products, and other gaseous airborne radionuclides. The radon pathway is not included in the current version of the RESRAD code because generic soil guidelines have been established for radium (Ra-226 and Ra-228), which is the source of all radon (see Appendix A). The other gaseous airborne decay products are primarily tritium (in tritiated water vapor) and C-14 (in CO_2); these are not included in the current version because they constitute only a small, usually negligible, fraction of the residual radioactivity at FUSRAP and SFMP sites and are unlikely to contribute significantly to the dose received by any individual.

The contribution to the committed effective dose equivalent from the dust inhalation pathway for the i^{th} principal radionuclide at time t following the radiological survey is given by the dose/source ratio $DSR_{i2}(t)$. This ratio may be expressed as a sum of products of dose conversion factors, environmental transport factors, and source factors (see Equation 3.8). A tabulation of dose conversion factors for dust inhalation is presented in Section C.1. Models and formulas for calculating the environmental transport factors are given in Section C.2. Formulas for calculating the source factors are given in Section 3.2.3 and Appendix F.

C.1 DOSE CONVERSION FACTORS

A dose conversion factor for inhalation is the dose/exposure ratio $DCF_{i2} = H_{E,i2}/E_{i2}$ for the committed effective dose equivalent $H_{E,i2}$ that is incurred by an individual from exposure by inhalation of a quantity E_{i2} of the i^{th} principal radionuclide in contaminated dust. Values of dose conversion factors for inhalation were taken from a DOE report (DOE 1988) and are listed in Table C.1. Similar values are given in an EPA report (Eckerman et al. 1988). The values listed in Table C.1 are for dust particles with an activity median aerodynamic diameter (AMAD) of 1 micron. Values for different inhalation classes are also listed in Table C.1. The inhalation class for inhaled radioactive material is defined according to its rate of clearance from the lung. The three inhalation classes D, W, and Y correspond to retention half-times of less than 10 days, 10 to 100 days, and greater than 100 days, respectively. If the inhalation class for a radionuclide is not known, the largest dose conversion factor for that radionuclide should be used. The default value for a radionuclide used in the RESRAD code is the largest dose conversion factor for that radionuclide.

C.2 ENVIRONMENTAL TRANSPORT FACTORS

An environmental transport factor for dust inhalation is the ratio $ETF_{i2}(t) = E_{i2}(t)/S_i(t)$ of the annual intake $E_{i2}(t)$ of the i^{th} principal radionuclide by dust inhalation to the concentration $S_i(t)$ of that radionuclide in the soil for the t^{th} year following the radiological survey. It can be expressed as the product

$$ETF_{i2}(t) = ASR_2 \times FA_2 \times FCD_2(t) \times FO_2 \times FI_2 \tag{C.1}$$

where

$ETF_{i2}(t)$ = environmental transport factor at time t for dust inhalation for the i^{th} principal radionuclide (g/yr),

TABLE C.1 Committed Effective Dose Equivalent Conversion Factors (DCF_{i2}) for Inhalation[a]

Radionuclide i^b	Inhalation Class[c]	DCF_{i2} (mrem/pCi)
H-3	* (H_2O)	6.3×10^{-8}
C-14	* (organic)	2.1×10^{-6}
	* (CO)	2.9×10^{-9}
	* (CO_2)	2.4×10^{-8}
Fe-55	D	2.6×10^{-6}
	W	1.2×10^{-6}
Co-60	W	3.0×10^{-5}
	Y	1.5×10^{-4}
Ni-59	D	1.3×10^{-6}
	W	7.0×10^{-7}
	* (vapor)	2.7×10^{-6}
Ni-63	D	3.0×10^{-6}
	W	1.9×10^{-6}
	* (vapor)	6.3×10^{-6}
Sr-90+D	D	2.3×10^{-4}
	Y	1.3×10^{-3}
Nb-94	W	2.6×10^{-5}
	Y	3.3×10^{-4}
Tc-99	D	8.4×10^{-7}
	W	7.5×10^{-6}
I-129	D	1.8×10^{-4}
Cs-135	D	4.5×10^{-6}
Cs-137+D	D	3.2×10^{-5}
Eu-152	W	2.2×10^{-4}
Eu-154	W	2.6×10^{-4}
Pb-210+D	D	2.1×10^{-2}
Ra-226+D	W	7.9×10^{-3}
Ra-228+D	W	4.5×10^{-3}
Ac-227+D	D	6.7
	W	1.7
	Y	1.2
Th-228+D	W	2.5×10^{-1}
	Y	3.1×10^{-1}
Th-229+D	W	2.0
	Y	1.7
Th-230	W	3.2×10^{-1}
	Y	2.6×10^{-1}
Th-232	W	1.6
	Y	1.1
Pa-231	W	1.3
	Y	8.6×10^{-1}
U-232	D	1.2×10^{-2}
	W	1.3×10^{-2}
	Y	6.7×10^{-1}
U-233	D	2.7×10^{-3}
	W	7.1×10^{-3}
	Y	1.3×10^{-1}

TABLE C.1 (Cont'd)

Radio-nuclide i^b	Inhalation Class^c	DCF_{i2} (mrem/pCi)	Radio-nuclide i^b	Inhalation Class^c	DCF_{i2} (mrem/pCi)
U-234	D	2.7×10^{-3}	Pu-239	W	5.1×10^{-1}
	W	7.1×10^{-3}		Y	3.3×10^{-1}
	Y	1.3×10^{-1}			
			Pu-240	W	5.1×10^{-1}
U-235+D	D	2.5×10^{-3}		Y	3.3×10^{-1}
	W	6.7×10^{-3}			
	Y	1.2×10^{-1}	Pu-241	W	1.0×10^{-2}
				Y	5.7×10^{-3}
U-236	D	2.5×10^{-3}			
	W	6.7×10^{-3}	Pu-242	W	4.8×10^{-1}
	Y	1.2×10^{-1}		Y	3.1×10^{-1}
U-238+D	D	2.4×10^{-3}	Am-241	W	5.2×10^{-1}
	W	6.2×10^{-3}			
	Y	1.2×10^{-1}	Am-243+D	W	5.2×10^{-1}
Np-237+D	W	4.9×10^{-1}	Cm-243	W	3.5×10^{-1}
Pu-238	W	4.6×10^{-1}	Cm-244	W	2.7×10^{-1}
	Y	3.0×10^{-1}			

---

[a] Inhalation factors are for an AMAD of 1 micron.

[b] Dose conversion factors for entries labeled by "+D" are aggregated dose conversion factors for intake of a principal radionuclide together with radionuclides of the associated decay chain in secular equilibrium (see Section 3.1 and Table 3.1).

[c] An asterisk (*) indicates a gaseous material. The three inhalation classes D, W, and Y correspond to retention half-times of less than 10 days, 10 to 100 days, and greater than 100 days, respectively.

ASR_2 = air/soil concentration ratio = average mass loading of airborne contaminated soil particles (2×10^{-4} g/m^3),

FA_2 = area factor (dimensionless),

$FCD_2(t)$ = cover and depth factor (dimensionless),

FO_2 = occupancy factor (0.45, dimensionless), and

FI_2 = annual intake of air (8,400 m^3/yr).

The mass loading is a conservative estimate that takes into account short periods of high mass loading and sustained periods of normal activity on a typical farm (Gilbert et al. 1983, Appendix A).

Three models are commonly used for the process by which dust becomes airborne (Healy and Rodgers 1979, Appendix E; Oztunali et al. 1981, Appendix A; Gilbert et al. 1983, Appendix A). One is a resuspension factor model in which the airborne dust concentration (C_{dust}) is given as a function of an empirically determined resuspension factor (R_f), the effective depth of the layer of dust from which resuspension occurs (d_r), and the bulk soil density (ρ_b). The formula relating these variables is

$$C_{dust} = R_f \, d_r \, \rho_b \tag{C.2}$$

The second is a resuspension rate model in which the airborne dust concentration is given as a function of an empirically determined resuspension rate (R_r), surface dust concentration ($\sigma_s = \rho_b \, d_r$), and average deposition velocity (v_d). The formula is

$$C_{dust} = R_r \, \sigma_s / v_d \tag{C.3}$$

The third is a mass loading model in which an average value of the airborne dust concentration is specified on the basis of empirical data.

The resuspension factor and resuspension rate are related by the equality $R_r = v_d R_f$; hence, the resuspension factor and resuspension rate models are not independent. Both resuspension models require two parameters that must be determined empirically and can vary over a wide range: (1) a resuspension factor or rate and (2) the thickness of the resuspendable layer. (The deposition velocity can also vary because of dependence on particle size, but the uncertainty in assigning a value is somewhat smaller.) The mass loading model uses a single parameter that is more directly measurable and for which empirical data are more readily available. The mass loading model has, therefore, been used in RESRAD for estimating the airborne dust concentration near the source.

For on-site exposure, the transport process may be regarded as a dilution process in which the resuspended contaminated dust is mixed with uncontaminated dust blown in from off-site. This dilution can be modeled by a Gaussian plume model (using an area distribution of point sources with zero release height) or by a simple mixing model that assumes perfect mixing of resuspended on-site contaminated dust with off-site uncontaminated dust within a volume defined by a mixing height and the area of the exposed contaminated zone. The Gaussian plume model, as applied to annual average meteorological data, using sector averages, is generally regarded as applicable over distances from one hundred meters to several thousand meters; it has not been validated for on-site exposure in which the exposure occurs immediately above the source. There are uncertainties regarding exposure in the immediate area, primarily because of possible air turbulence or downdrafts created by buildings and structures. Considering these uncertainties regarding the

Gaussian plume model for on-site exposure and the large increase in the size of the code that incorporation of an area-source Gaussian plume model would entail, a simple mixing model has been used in the current version of RESRAD.

The area factor represents the fraction of airborne dust that is contaminated. It is calculated using a mixing model for estimating the dilution of contaminated dust that is resuspended on-site by uncontaminated dust blown in from off-site and is given by the formula

$$FA_2 = A^{1/2}/[A^{1/2} + DL] \tag{C.4}$$

where

A = area of contaminated zone (10,000 m^2) and

DL = dilution length (3 m).

The dilution length depends on the wind speed, mixing height, resuspension rate, and thickness of the resuspendable dust layer (Gilbert et al. 1983, Appendix A). Estimates of lower and upper bounds of DL for bounding values of the independent variables are 0.03 m and 250 m, respectively. The geometric mean of the bounds, DL = 3 m, is used as the default value. The model has not been tested experimentally; hence, the accuracy and range of values of A and DL for which the model is applicable are not known.

The cover and depth factor is the fraction of resuspendable soil particles at the ground surface that are contaminated. It is calculated by assuming that mixing of the soil will occur within a layer of thickness d_m at the surface. The cover and depth factor $FCD_{i2}(t)$ is calculated by the formula

$$FCD_{i2}(t) = \begin{cases} 1, & C_d(t) = 0,\ T(t) \geq d_m \\ T(t)/d_m, & C_d(t) + T(t) < d_m \\ 1 - C_d(t)/d_m, & C_d(t) < d_m,\ C_d(t) + T(t) \geq d_m \\ 0, & C_d(t) \geq d_m \end{cases} \qquad (C.5)$$

where

d_m = depth of soil mixing layer (0.15 m),
$C_d(t)$ = cover depth at time t (m), and
$T(t)$ = thickness of contaminated zone at time t (m).

The default occupancy factor of $FO_2 = 0.45$ used in the RESRAD code is based on the assumption that 50% of a person's time is spent indoors [where the dust level is 40% of the outdoor level (Alzona et al. 1979)], 25% is spent outdoors in the contaminated area, and 25% is spent in uncontaminated areas.

The annual air intake of 8,400 m^3/yr used in the RESRAD code is the value recommended by the International Commission on Radiological Protection (1975).

C.3 REFERENCES

Alzona, J., et al., 1979, *Indoor-Outdoor Relationships for Airborne Particulate Matter of Outdoor Origin,* Atmospheric Environment, *13*:55-60.

Eckerman, K.F., A.B. Wolbarst, and A.C.B. Richardson, 1988, *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion,* EPA-520/1-88-020, Federal Guidance Report No. 11, prepared by Oak Ridge National Laboratory, Oak Ridge, Tenn., for U.S. Environmental Protection Agency, Office of Radiation Programs, Washington, D.C.

Gilbert, T.L., et al., 1983, *Pathways Analysis and Radiation Dose Estimates for Radioactive Residues at Formerly Utilized MED/AEC Sites,* ORO-832 (Rev.), prepared by Division of Environmental Impact Studies, Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations, Oak Ridge, Tenn. (March 1983 [reprinted with corrections: January 1984]).

Healy, J.W., and J.C. Rodgers, 1979, *Limits for the Burial of the Department of Energy Transuranic Wastes,* LA-UR-79-100, Los Alamos Scientific Laboratory, Los Alamos, N.M.

International Commission on Radiological Protection, 1975, *Report of the Task Group on Reference Man,* prepared by a Task Group of Committee 2, adopted by the Commission in October 1974, ICRP Publication 23, Pergamon Press, New York.

Oztunali, O.I., et al., 1981, *Data Base for Radioactive Waste Management, Impact Analyses Methodology Report,* NUREG/CR-1759, Vol. 3, prepared by Dames & Moore, White Plains, N.Y., for Office of Nuclear Material Safety and Safeguards, U.S. Nuclear Regulatory Commission, Washington, D.C.

U.S. Department of Energy, 1988, *Internal Dose Conversion Factors for Calculation of Dose to the Public,* DOE/EH-0071, Assistant Secretary for Environment, Safety and Health (July).

APPENDIX D

INGESTION PATHWAY FACTORS

APPENDIX D: INGESTION PATHWAY FACTORS

Ingestion pathways consist of five environmental pathways and the common exposure pathway to which they contribute (see Table 2.1). The doses from these pathways are specified by the dose/source ratios described in Chapter 3. Dose/source ratios may be analyzed into sums of products of dose conversion factors (which characterize the exposure pathways), environmental transport factors (which characterize the environmental pathways), and source factors (which characterize ingrowth and decay and leaching of the radionuclides) (see Equation 3.8). Dose conversion factors for ingestion are described in Section D.1. Environmental transport factors and the models and formulas used to calculate them are presented in Sections D.2-D.4. Formulas for calculating the source factors are given in Section 3.2.3 and Appendix F.

D.1 DOSE CONVERSION FACTORS

A dose conversion factor for ingestion is the dose/exposure ratio $DCF_{i3} = H_{E,i3}/E_i$, where $H_{E,i3}$ is the committed effective dose equivalent incurred by an individual from intake by ingestion of a quantity E_i of the i^{th} principal radionuclide. Values of dose conversion factors for ingestion were taken from a DOE report (DOE 1988) and are tabulated in Table D.1. Similar values are given in an EPA report (Eckerman et al. 1988). Dose conversion factors depend on the chemical form, which determines the fraction f_1 of a radionuclide entering the gastrointestinal (GI) tract that reaches body fluids. Data on the appropriate fractions for different chemical forms are given in Publication No. 30 of the International Commission on Radiological Protection (ICRP 1979-1982). The dose conversion factors used in RESRAD are the values corresponding to the largest values of f_1 in Table D.1. If the chemical form of a radionuclide is known and a different value of DCF_{i3} is

TABLE D.1 Committed Effective Dose Equivalent Conversion Factors (DCF_{i3}) for Internal Radiation from Ingestion

Radionuclide i^a	$f_1{}^b$	DCF_{i3} (mrem/pCi)	Radionuclide i^a	$f_1{}^b$	DCF_{i3} (mrem/pCi)
H-3	1.0	6.3×10^{-8}	U-232	5×10^{-2}	1.3×10^{-3}
C-14	1.0	2.1×10^{-6}		2×10^{-3}	6.8×10^{-5}
Fe-55	1×10^{-1}	5.8×10^{-7}	U-233	5×10^{-2}	2.7×10^{-4}
Co-60	3×10^{-1}	2.6×10^{-5}		2×10^{-3}	2.5×10^{-5}
	5×10^{-2}	1.0×10^{-5}	U-234	5×10^{-2}	2.6×10^{-4}
Ni-59	5×10^{-2}	2.0×10^{-7}		2×10^{-3}	2.5×10^{-5}
Ni-63	5×10^{-2}	5.4×10^{-7}	U-235+D	5×10^{-2}	2.5×10^{-4}
Sr-90+D	3×10^{-1}	1.4×10^{-4}		2×10^{-3}	2.6×10^{-5}
	1×10^{-2}	1.2×10^{-5}	U-236	5×10^{-2}	2.5×10^{-4}
Nb-94	1×10^{-2}	5.1×10^{-6}		2×10^{-3}	2.4×10^{-5}
Tc-99	8×10^{-1}	1.3×10^{-6}	U-238+D	5×10^{-2}	2.5×10^{-4}
I-129	1.0	2.8×10^{-4}		2×10^{-3}	3.8×10^{-5}
Cs-135	1.0	7.1×10^{-6}	Np-237+D	1×10^{-3}	3.9×10^{-3}
Cs-137+D	1.0	5.0×10^{-5}	Pu-238	1×10^{-3}	3.8×10^{-3}
Eu-152	1×10^{-3}	6.0×10^{-6}		1×10^{-5}	5.4×10^{-5}
Eu-154	1×10^{-3}	9.1×10^{-6}	Pu-239	1×10^{-3}	4.3×10^{-3}
Pb-210+D	2×10^{-1}	6.7×10^{-3}		1×10^{-5}	5.8×10^{-5}
Ra-226+D	2×10^{-1}	1.1×10^{-3}	Pu-240	1×10^{-3}	4.3×10^{-3}
Ra-228+D	2×10^{-1}	1.2×10^{-3}		1×10^{-5}	5.8×10^{-5}
Ac-227+D	1×10^{-3}	1.5×10^{-2}	Pu-241	1×10^{-3}	8.6×10^{-5}
Th-228+D	2×10^{-4}	7.5×10^{-4}		1×10^{-5}	9.2×10^{-7}
Th-229+D	2×10^{-4}	4.3×10^{-3}	Pu-242	1×10^{-3}	4.1×10^{-3}
Th-230	2×10^{-4}	5.3×10^{-4}		1×10^{-5}	5.6×10^{-5}
Th-232	2×10^{-4}	2.8×10^{-3}	Am-241	1×10^{-3}	4.5×10^{-3}
Pa-231	1×10^{-3}	1.1×10^{-2}	Am-243+D	1×10^{-3}	4.5×10^{-3}
			Cm-243	1×10^{-3}	2.9×10^{-3}
			Cm-244	1×10^{-3}	2.3×10^{-3}

[a] Dose conversion factors for entries labeled by "+D" are aggregated dose conversion factors for intake of a principal radionuclide together with radionuclides of the associated decay chain in secular equilibrium (see Section 3.1).

[b] Fraction of a stable element entering the GI tract that reaches body fluids.

needed, the value can be entered by modifying the DCF_{i3} data file contained in the RESRAD package.

D.2 ENVIRONMENTAL TRANSPORT FACTORS

D.2.1 Plant, Meat, and Milk Pathways

D.2.1.1 Environmental Transport Factor Components

Environmental transport factors for the plant, meat, and milk pathways can be factored into the product

$$ETF_{ipq}(t) = FA_p \times FCD_{ipq}(t) \times \sum_k DF_{pk} \times FSR_{ipqk}(t) \qquad (D.1)$$

where

$ETF_{ipq}(t)$ = environmental transport factor for the i^{th} principal radionuclide and pq^{th} environmental pathway at time t (g/yr),

p = primary pathway index for the plant (p = 3), meat (p = 4), and milk (p = 5) pathways,

q = secondary index for root uptake (q = 1), foliar deposition (q = 2), ditch irrigation (q = 3), overhead irrigation (q = 4), and livestock water (q = 5),

FA_p = area factor for p^{th} primary pathway (dimensionless),

$FCD_{ipq}(t)$ = cover and depth factor for the i^{th} principal radionuclide and pq^{th} ingestion pathway at time t (dimensionless),

DF_{pk} = dietary factor = annual consumption of the k^{th} food class for the p^{th} food pathway (g/yr),

k = food class index, and

$FSR_{ipqk}(t)$ = food/soil concentration ratio for the i^{th} principal radionuclide, pq^{th} ingestion pathway, and k^{th} food class at time t (dimensionless).

The sum is over the applicable food classes. The plant pathway consists of two food classes: (1) fruit, nonleafy vegetables, and grain (k = 1) and (2) leafy vegetables (k = 2). For the meat and milk pathways, only a single food class is used in the current version of RESRAD. Dietary factors are commonly given in kg/yr; they must be converted to g/yr before substitution into Equation D.1 in order for FSR = C/S to be a dimensionless quantity when the soil concentration S is specified in pCi/g.

For water-dependent pathways (pathways with q = 3, 4, or 5 — which include a groundwater or surface water pathway segment), the food/soil concentration ratios can be factored into the products

$$FSR_{ipqk}(t) = FWR_{ipqk} \times WSR_{ir}(t) \tag{D.2}$$

where

FWR_{ipqk} = food/water concentration ratio for the i^{th} principal radionuclide, pq^{th} water-dependent pathway, and k^{th} food class (L/g),

$WSR_{ir}(t)$ = water/soil concentration ratio for the r^{th} water pathway segment (g/L), and

r = water pathway segment index for groundwater (r = 1) or surface water (r = 2).

Substituting Equation D.2 into Equation D.1, one obtains

$$ETF_{ipqr}(t) = WEF_{ipq}(t) \times WSR_{ir}(t) \tag{D.3}$$

where

$$WEF_{ipq}(t) = FA_p \times FCD_{ipq}(t) \times \sum_k DF_{pk} \times FWR_{ipqk} \tag{D.4}$$

is the water exposure factor for the pq^th pathway in units of L/yr, and the quantities on the right are as defined for Equations D.1 and D.2. A water exposure factor is equal to the ratio of the annual intake of the i^th principal radionuclide in food contaminated through the pq^th water-dependent pathway to the concentration of the radionuclide in the contaminated water. Water/soil concentration ratios are defined and discussed in Appendix E.

D.2.1.2 Area Factors

The area factor for the plant (FA_3), meat (FA_4), and milk (FA_5) pathways is given by

$$\begin{aligned} FA_3 &= A/2{,}000, & 0 \le A \le 1{,}000\ m^2 \\ &= 0.5, & A > 1{,}000\ m^2 \\ FA_4 = FA_5 &= A/20{,}000, & 0 \le A \le 20{,}000\ m^2 \\ &= 1, & A > 20{,}000\ m^2 \end{aligned} \tag{D.5}$$

where A = area of contaminated zone (10,000 m^2).* An area of 10,000 m^2 = 1 hectare (ha).

---

*If a number is included with the unit identification, the variable is an input variable and the number is the default value.

D.2.1.3 Cover and Depth Factors

The cover and depth factors FCD_{ipq} for $q = 1$ through $q = 4$ are applicable to the plant ($p = 3$), meat ($p = 4$), and milk ($p = 5$) pathways. The cover and depth factor FCD_{ip5} for the livestock water pathway is applicable only to the meat and milk pathways.

Root Uptake. The cover and depth factor for root uptake ($q = 1$) is

$$
\begin{aligned}
FCD_{ip1}(t) &= 1, && C_d(t) = 0,\ T(t) \ge d_r \\
&= T(t)/d_r, && C_d(t) + T(t) < d_r \\
&= 1 - C_d(t)/d_r, && C_d(t) < d_r,\ C_d(t) + T(t) \ge d_r \\
&= 0, && C_d(t) \ge d_r
\end{aligned}
\qquad (D.6)
$$

where

$C_d(t)$ = cover depth at time t (m),

$T(t)$ = thickness of contaminated zone at time t (m), and

d_r = maximum root depth (0.9 m).

The cover depth is given by Equation B.6. Equation D.6 is based on the assumption of a sharp boundary between the bottom of any uncontaminated cover and the top of the contaminated zone. The effect of mixing uncontaminated and contaminated soil in a surface layer by plowing or other disturbance of the soil close to the ground surface is not taken into account.

Foliar Deposition. The cover and depth factor $FCD_{ip2}(t)$ for foliar deposition for the ingestion pathways is the same as the cover and depth factor $FCD_{i2}(t)$ for the inhalation pathway (see Equation C.5).

Ditch Irrigation, Overhead Irrigation, and Livestock Water. The contributions from ditch irrigation ($q = 3$), overhead irrigation ($q = 4$), and livestock water ($q = 5$) are independent of the depth of the contaminated zone in the models used because the infiltrating water will carry the contamination to the aquifer and contaminate the water regardless of the depth, and the subsequent contamination mechanisms do not depend on depth. Hence, the cover and depth factors for these subpathways are

$$FCD_{ip3}(t) = FCD_{ip4}(t) = FCD_{ip5}(t) = 1 \tag{D.7}$$

D.2.1.4 Dietary Factors

The dietary factor for human food consumption is a tabulated quantity. The values used are given in Table D.2.

TABLE D.2 Dietary Factors (DF_{pk}) for Human Food Consumption

Primary Pathway Index, p	Ingestion Pathway	Food Class Index, k	Class Description	Dietary Factor, DF_{pk}
3	Plant foods	1	Fruits, vegetables, and grain	160 kg/yr
3	Plant foods	2	Leafy vegetables	14 kg/yr
4	Meat	1	Meat and poultry	63 kg/yr
5	Milk	1	Milk	92 L/yr
6	Aquatic foods	1	Fish	5.4 kg/yr
6	Aquatic foods	2	Crustacea and mollusks	0.9 kg/yr
7	Drinking water	1	Drinking water	410 L/yr

D.2.1.5 Food/Soil Concentration Ratios for Plant Foods

Root Uptake. The plant-food/soil concentration ratios for root uptake are given by

$$FSR_{i31k} = B_{iv} \qquad (D.8)$$

where B_{iv} are the vegetable/soil transfer factors listed in Table D.3. It is assumed that the same root uptake transfer factors can be used for nonleafy vegetables (k = 1) and leafy vegetables (k = 2).

Foliar Deposition. The plant-food/soil concentration ratio for uptake from airborne contaminants that deposit on foliage is given by

$$FSR_{i32k} = FA_2 \times FAR_{i32k} \times ASR_3 \qquad (D.9)$$

where

FSR_{i32k} = plant-food/soil concentration ratio for foliar deposition for the i^{th} principal radionuclide and k^{th} food class (dimensionless),

FA_2 = area factor for dilution of resuspended contaminated dust (dimensionless),

FAR_{i32k} = plant-food/air concentration ratio for radionuclide transfer by airborne foliar deposition for the i^{th} principal radionuclide and k^{th} food class (m^3/g), and

TABLE D.3 Vegetable/Soil Transfer Factors (B_{iv}) for Root Uptake

Element	B_{iv}	Element	B_{iv}	Element	B_{iv}
H	0	Sr	2.0×10^{-1}	Pm	2.5×10^{-3}
Be	4.7×10^{-4}	Y	2.5×10^{-3}	Sm	2.5×10^{-3}
C	0	Zr	1.7×10^{-4}	Eu	2.5×10^{-3}
N	7.5	Nb	9.4×10^{-3}	Tb	2.6×10^{-3}
F	2.0×10^{-2}	Mo	1.3×10^{-1}	Ho	2.6×10^{-3}
Na	5.0×10^{-2}	Tc	2.5×10^{-1}	W	1.8×10^{-2}
P	5.0×10^{-1}	Ru	1.0×10^{-2}	Ir	9.9×10^{-4}
Ar	0	Rh	1.3×10^{-1}	Hg	3.8×10^{-1}
Ca	4.0×10^{-2}	Pd	5.0	Pb	6.8×10^{-2}
Sc	1.1×10^{-3}	Ag	1.5×10^{-1}	Bi	1.5×10^{-1}
Cr	2.5×10^{-4}	Cd	3.0×10^{-1}	Po	9.0×10^{-3}
Mn	3.0×10^{-2}	Sn	2.5×10^{-3}	Rn	0
Fe	4.0×10^{-4}	Sb	1.1×10^{-2}	Ra	1.4×10^{-3}
Co	9.4×10^{-3}	Te	1.3	Ac	2.5×10^{-3}
Ni	1.9×10^{-2}	I	2.0×10^{-2}	Th	4.2×10^{-3}
Cu	1.3×10^{-1}	Xe	0	Pa	2.5×10^{-3}
Zn	4.0×10^{-1}	Cs	2.0×10^{-3}	U	2.5×10^{-3}
As	1.0×10^{-2}	Ba	5.0×10^{-3}	Np	2.5×10^{-3}
Se	1.3	La	2.5×10^{-3}	Pu	2.5×10^{-4}
Br	7.6×10^{-1}	Ce	5.0×10^{-4}	Am	2.5×10^{-4}
Kr	0	Pr	2.5×10^{-3}	Cm	2.5×10^{-3}
Rb	1.3×10^{-1}	Nd	2.4×10^{-3}	Cf	2.5×10^{-3}

ASR_3 = air/soil concentration ratio, specified as the average mass loading of airborne contaminated soil particles in a garden during the growing season (1×10^{-4} g/m^3).

The area factor for dilution of resuspended contaminated dust by mixing with uncontaminated dust blown in from off-site is given by Equation C.2. The formula for the plant-food/air concentration ratio is

$$FAR_{i32k} = 3.16 \times 10^4 \frac{[V_{di} \times f_r \times T_{ivk}][1 - \exp(-\lambda_w t_{ek})]}{Y_{vk} \times \lambda_w} \qquad (D.10)$$

where

V_{di} = deposition velocity of contaminated dust (0 m/s for H, C, Ar, Kr, and Xe; 1×10^{-2} m/s for F, Br, I, and Cl; and 1×10^{-3} m/s for all remaining elements listed in Table D.3),

f_r = fraction of deposited radionuclides retained on the vegetation (0.25, dimensionless),

T_{ivk} = foliage-to-food radionuclide transfer coefficient for the ith principal radionuclide and kth food class ($T_{iv1} = 0.1$, $T_{iv2} = 1.0$, and $T_{iv3} = 1.0$, dimensionless),

Y_{vk} = wet-weight crop yield for the kth food class (Y_{v1} = 0.7 kg/m^2, Y_{v2} = 1.5 kg/m^2, and Y_{v3} = 1.1 kg/m^2),

λ_w = weathering removal constant for vegetation (20 yr^{-1}), and

t_{ek} = time of exposure of the kth food class to contamination during the growing season (t_{e1} = 0.17 yr, t_{e2} = 0.25 yr, and t_{e3} = 0.08 yr).

The numerical factor 3.16 × 10^4 (kg/g)(s/yr) is used to convert FAR_{i32k} to units of m^3/g. The foliage-to-food transfer coefficient T_{ivk} is assumed to be radionuclide independent in the current version of the code. The weathering removal constant corresponds to a contaminant removal half-time of two weeks. The values for k = 3 are for fodder; they are not included in the sum over k in Equation D.1.

Ditch Irrigation. The plant-food/soil concentration ratio for ditch irrigation is given by

$$FSR_{i33k}(t) = FWR_{i33k} \times \left[WSR_{i1}(t) \times FI1 + WSR_{i2}(t) \times (1 - FI1)\right] \quad (D.11)$$

where

$FSR_{i33k}(t)$ = plant-food/soil concentration ratio for ditch irrigation for the i^{th} principal radionuclide and k^{th} food class at time t (dimensionless),

FWR_{i33k} = plant-food/water concentration ratio for ditch irrigation for the i^{th} principal radionuclide and the k^{th} food class (L/g),

$WSR_{i1}(t)$ = well-water/soil concentration ratio for the i^{th} principal radionuclide (g/L),

FI1 = fraction of well water used for irrigation (balanced from surface water; 1.0, dimensionless), and

$WSR_{i2}(t)$ = surface-water/soil concentration ratio for the i^{th} principal radionuclide (g/L).

Plant-food/soil concentration ratios for root uptake are assumed to be the same for all food classes. The formula for the plant-food/water concentration ratio, FWR_{i33k}, for ditch irrigation is

$$FWR_{i33k} = \frac{I_{rr} \; B_{iv} \left[1 - \exp(- L_i t_{ek})\right]}{Y_{vk} \times L_i} \qquad \text{(D.12)}$$

where

I_{rr} = irrigation rate (0 m/yr), and

L_i = leach rate constant for radionuclide i (yr^{-1}).

FWR_{i33k} is in dimensions of L/g (m^3/kg), B_{iv} is as defined for Equation D.8, and t_{ek} and Y_{vk} are as defined for Equation D.10. The leach rate constant L_i is discussed in Appendix E. The default value used in the RESRAD code for fraction of well water used for irrigation is 1.0, i.e., 100% of irrigation water is well water. A value of 0.5 means 50% of irrigation water is well water and 50% is surface (pond) water. The water/soil concentration ratios are discussed in Appendix E.

Overhead Irrigation. The factoring of the plant-food/soil concentration ratio for overhead irrigation, FSR_{i34k}, is the same as the factoring for ditch irrigation in Equation D.11. That is,

$$FSR_{i34k}(t) = FWR_{i34k} \times \left[WSR_{i1}(t) \times FI1 + WSR_{i2}(t) \times (1 - FI1)\right] \qquad \text{(D.13)}$$

The only difference is that, for overhead irrigation, the plant-food/water concentration ratio is given by

$$FWR_{i34k} = \frac{[I_{rr} \times f_r \times T_{ivk}][1 - \exp(-\lambda_w t_{ek})]}{Y_{vk} \times \lambda_w} + FWR_{i33k} \qquad (D.14)$$

where FWR_{i34k} is in units of L/g (m^3/kg), I_{rr} is as defined for Equation D.12, and the remaining quantities are as defined for Equation D.10.

D.2.1.6 Food/Soil Concentration Ratios for Meat and Milk

The food/soil concentration ratios $FSR_{ipq}(t)$ for meat and milk can be factored into the product

$$FSR_{ipq}(t) = FQR_{ip} \times FI_{pq} \times QSR_{ipq}(t) \qquad (D.15)$$

where

FQR_{ip} = radionuclide transfer factor for meat ($p = 4$) or milk ($p = 5$) = ratio of the concentration of the i^{th} principal radionuclide in meat or milk in pCi/kg to the rate of intake in fodder or water by livestock of the i^{th} principal radionuclide in pCi/d (d/kg),

FI_{pq} = daily intake of fodder ($q = 1$, 2, 3, and 4) or water ($q = 5$) by livestock (kg/d -- see below for default values), and

$QSR_{ipq}(t)$ = fodder/soil or livestock-water/soil concentration ratio for meat ($p = 3$) or milk ($p = 5$) for the i^{th} principal radionuclide and q^{th} subpathway (dimensionless).

The radionuclide transfer factors FQR_{ip} are listed in Table D.4. The livestock fodder intake rates are $FI_{4q} = 68$ kg/d and $FI_{5q} = 55$ kg/d for q = 1, 2, 3, and 4 for meat and milk, respectively, and the livestock water intake rates are $FI_{45} = 50$ L/d and $FI_{55} = 160$ L/d for q = 5. (Note that 1 L of water weighs 1 kg; hence, units of kg/d may be used for all intakes.)

The expressions for the fodder/soil concentration ratios are the same as the expressions for the food/soil concentration ratios given earlier; the only difference is in a few parameter values. QSR_{ipq} is time independent for the water-independent pathways (q = 1, 2) and depends on time only through the water/soil concentration ratio $WSR_{i1}(t)$ for the water-dependent pathways (q = 3, 4 and 5). The formulas for the fodder/soil concentration ratios are as follows.

$$QSR_{i41} = QSR_{i51} = B_{iv} \qquad \text{(D.16)}$$

is the ratio for root uptake by fodder, where B_{iv} is the vegetable/soil transfer factor for root uptake, tabulated in Table D.3.

$$QSR_{i42} = QSR_{i52} = FA_2 \times FAR_{i323} \times ASR_3 \qquad \text{(D.17)}$$

is the ratio for foliar deposition on fodder, where FA_2 is an area factor and ASR_3 is the air/soil concentration ratio, both defined for Equation D.9, and FAR_{i323} is the plant-food/air concentration ratio defined by Equation D.10.

$$\begin{aligned} QSR_{i43}(t) &= QSR_{i53}(t) \\ &= FWR_{i333} \times \left[WSR_{i1}(t) \times FI1 + WSR_{i2} \times (1 - FI1)\right] \end{aligned} \qquad \text{(D.18)}$$

TABLE D.4 Meat (FQR_{i4}) and Milk (FQR_{i5}) Transfer Factors for Root Uptake

Element	Beef (FQR_{i4}) (d/kg)	Milk (FQR_{i5}) (d/L)	Element	Beef (FQR_{i4}) (d/kg)	Milk (FQR_{i5}) (d/L)
H	0	0	Sn	9.9×10^{-4}	1.3×10^{-3}
Be	8.0×10^{-4}	2.0×10^{-6}	Sb	3.0×10^{-3}	7.5×10^{-4}
C	0	0	Te	5.0×10^{-2}	5.0×10^{-4}
N	9.9×10^{-4}	1.0×10^{-2}	I	2.0×10^{-2}	1.0×10^{-2}
F	2.0×10^{-2}	7.0×10^{-3}	Xe	0	0
Na	5.0×10^{-2}	4.0×10^{-2}	Cs	3.0×10^{-2}	5.0×10^{-3}
P	5.0×10^{-2}	1.2×10^{-2}	Ba	5.0×10^{-4}	4.0×10^{-4}
Ar	0	0	La	5.0×10^{-3}	2.5×10^{-6}
Ca	3.3×10^{-3}	8.0×10^{-3}	Ce	1.0×10^{-3}	1.0×10^{-5}
Sc	6.0×10^{-3}	2.5×10^{-6}	Pr	5.0×10^{-3}	2.5×10^{-6}
Cr	9.9×10^{-4}	1.1×10^{-3}	Nd	5.0×10^{-3}	2.5×10^{-6}
Mn	5.0×10^{-3}	1.0×10^{-4}	Pm	5.0×10^{-3}	2.5×10^{-6}
Fe	2.0×10^{-2}	6.0×10^{-4}	Sm	5.0×10^{-3}	2.5×10^{-6}
Co	1.0×10^{-3}	5.0×10^{-4}	Eu	5.0×10^{-3}	2.5×10^{-6}
Ni	1.0×10^{-3}	3.4×10^{-3}	Tb	5.0×10^{-3}	2.5×10^{-6}
Cu	1.0×10^{-2}	7.0×10^{-3}	Ho	5.0×10^{-3}	2.5×10^{-6}
Zn	5.0×10^{-2}	6.0×10^{-3}	W	9.9×10^{-4}	2.5×10^{-4}
As	1.5×10^{-3}	3.0×10^{-3}	Ir	9.9×10^{-4}	9.9×10^{-4}
Se	1.0	2.3×10^{-2}	Hg	1.0×10^{-1}	1.9×10^{-2}
Br	2.0×10^{-2}	2.5×10^{-2}	Pb	9.9×10^{-4}	1.0×10^{-5}
Kr	0	0	Bi	9.9×10^{-4}	2.5×10^{-4}
Rb	1.5×10^{-1}	1.0×10^{-2}	Po	9.9×10^{-4}	1.2×10^{-4}
Sr	3.0×10^{-4}	1.5×10^{-3}	Rn	0	0
Y	5.0×10^{-3}	5.0×10^{-6}	Ra	9.9×10^{-4}	2.0×10^{-4}
Zr	5.0×10^{-4}	2.5×10^{-6}	Ac	5.0×10^{-3}	2.5×10^{-6}
Nb	5.0×10^{-4}	1.2×10^{-3}	Th	5.0×10^{-3}	2.5×10^{-6}
Mo	1.0×10^{-2}	4.0×10^{-3}	Pa	5.0×10^{-3}	2.5×10^{-6}
Tc	9.9×10^{-4}	1.2×10^{-2}	U	5.0×10^{-3}	6.0×10^{-4}
Ru	1.0×10^{-3}	5.0×10^{-7}	Np	5.0×10^{-3}	2.5×10^{-6}
Rh	1.0×10^{-3}	5.0×10^{-3}	Pu	5.0×10^{-3}	2.5×10^{-8}
Pd	1.0×10^{-3}	5.0×10^{-3}	Am	5.0×10^{-3}	2.5×10^{-6}
Ag	9.9×10^{-4}	2.5×10^{-2}	Cm	5.0×10^{-3}	2.5×10^{-6}
Cd	1.6×10^{-2}	6.2×10^{-5}	Cf	5.0×10^{-3}	7.5×10^{-7}

is the ratio for ditch irrigation of fodder, where FWR_{i333} is the plant-food/water concentration ratio given by Equation D.12; $WSR_{i1}(t)$ and WSR_{i2} are the water/soil concentration ratios for well water and surface water, respectively, as derived in Appendix E; and FI1 is as defined for Equation D.11.

$$QSR_{i44}(t) = QSR_{i54}(t)$$
$$= FWR_{i343} \times \left[WSR_{i1}(t) \times FI1 + WSR_{i2}\ (1 - FI1)\right] \quad (D.19)$$

is the ratio for overhead irrigation of fodder, where FWR_{i343} is the fodder/water concentration ratio for overhead irrigation given by Equation D.14.

$$QSR_{i45}(t) = QSR_{i55}(t) = WSR_{i1}(t) \times FL1 + WSR_{i2}\ (1 - FL1) \quad (D.20)$$

is the ratio for intake of contaminated livestock water, where FL1 is the fraction of well water used for feeding livestock (balanced from surface water; 1.0, dimensionless).

D.2.2 Aquatic Food Pathway

The environmental transport factor for the aquatic food pathway (p = 6) can be factored into the product

$$ETF_{i6}(t) = FR_6 \times \left[\sum_k DF_{6k} \times FWR_{i6k}\right] \times WSR_{i2}(t) \quad (D.21)$$

where the summation is over the aquatic food classes (see Table D.2) and

$ETF_{i6}(t)$ = environmental transport factor for the aquatic food pathway (fish, crustacea, and mollusks) (g/yr),

DF_{6k} = dietary factors for annual consumption of fish (k = 1) and crustacea and mollusks (k = 2) (kg/yr),

FR_6 = fraction of aquatic food consumed that is contaminated (0.5, dimensionless),

FWR_{i6k} = fish/water (k = 1) and crustacea-mollusk/water (k = 2) concentration ratios (bioaccumulation factors) (L/kg), and

$WSR_{i2}(t)$ = water/soil concentration ratio for surface water (g/L).

Dietary factors are given in Table D.2. Bioaccumulation factors are given in Table D.5. The water/soil concentration ratio for surface water is calculated using the surface-water model described in Appendix E.

The environmental transport factors may also be written as products of a water exposure factor and a water/soil concentration ratio

$$ETF_{i6}(t) = WEF_{i6} \times WSR_{i2}(t) \qquad (D.22)$$

where

$$WEF_{i6}(t) = FR_6 \times \left[\sum_k DF_{6k} \times FWR_{i6k}\right] \qquad (D.23)$$

D.2.3 Drinking Water Pathway

The formula for the environmental transport factors for the drinking water pathway is

$$ETF_{i7}(t) = DF_7 \times FDW \times \left[WSR_{i1}(t) \times FD1 + WSR_{i2}(t) \times (1 - FD1)\right] \qquad (D.24)$$

TABLE D.5 Aquatic Bioaccumulation Factors (FWR_{i6k}) for Fresh Water

Element	Fish (k=1) (L/kg)	Crustacea and Mollusks (k=2) (L/kg)	Element	Fish (k=1) (L/kg)	Crustacea and Mollusks (k=2) (L/kg)
H	9.0×10^{-1}	9.0×10^{-1}	Sn	3.0×10^{3}	1.0×10^{3}
Be	2.0	1.0×10^{1}	Sb	1.0	1.0×10^{1}
C	4.6×10^{3}	9.1×10^{3}	Te	4.0×10^{2}	7.5×10^{1}
N	0	0	I	1.5×10^{1}	5.0
F	1.0×10^{1}	1.0×10^{2}	Xe	1.0	1.0
Na	1.0×10^{2}	2.0×10^{2}	Cs	2.0×10^{3}	1.0×10^{2}
P	1.0×10^{5}	2.0×10^{4}	Ba	4.0	2.0×10^{2}
Ar	1.0	1.0	La	2.5×10^{1}	1.0×10^{3}
Ca	4.0×10^{1}	3.3×10^{2}	Ce	1.0	1.0×10^{3}
Sc	2.0	1.0×10^{3}	Pr	2.5×10^{1}	1.0×10^{3}
Cr	2.0×10^{1}	2.0×10^{3}	Nd	2.5×10^{1}	1.0×10^{3}
Mn	4.0×10^{2}	9.0×10^{4}	Pm	2.5×10^{1}	1.0×10^{3}
Fe	1.0×10^{2}	3.2×10^{3}	Sm	2.5×10^{1}	1.0×10^{3}
Co	5.0×10^{1}	2.0×10^{2}	Eu	2.5×10^{1}	1.0×10^{3}
Ni	1.0×10^{2}	1.0×10^{2}	Tb	2.5×10^{1}	1.0×10^{3}
Cu	5.0×10^{1}	4.0×10^{2}	Ho	2.5×10^{1}	1.0×10^{3}
Zn	2.0×10^{3}	1.0×10^{4}	W	1.2×10^{3}	1.0×10^{1}
As	3.0×10^{2}	3.0×10^{2}	Ir	5.0×10^{1}	2.0×10^{2}
Se	1.7×10^{2}	1.7×10^{2}	Hg	2.0×10^{4}	2.0×10^{4}
Br	4.2×10^{2}	3.3×10^{2}	Pb	1.0×10^{2}	1.0×10^{2}
Kr	1.0	1.0	Bi	1.5×10^{1}	1.0×10^{1}
Rb	2.0×10^{3}	1.0×10^{3}	Po	5.0×10^{2}	2.0×10^{4}
Sr	3.0×10^{1}	1.0×10^{2}	Rn	5.7×10^{1}	1.0
Y	2.5×10^{1}	1.0×10^{3}	Ra	5.0×10^{1}	2.5×10^{2}
Zr	3.3	6.7	Ac	2.5×10^{1}	1.0×10^{3}
Nb	3.0×10^{4}	1.0×10^{2}	Th	3.0×10^{1}	5.0×10^{2}
Mo	1.0×10^{1}	1.0×10^{1}	Pa	1.1×10^{1}	1.1×10^{2}
Tc	1.5×10^{1}	5.0	U	2.0	6.0×10^{1}
Ru	1.0×10^{1}	3.0×10^{2}	Np	1.0×10^{1}	4.0×10^{2}
Rh	1.0×10^{1}	3.0×10^{2}	Pu	3.5	1.0×10^{2}
Pd	1.0×10^{1}	3.0×10^{2}	Am	2.5×10^{1}	1.0×10^{3}
Ag	2.3	7.7×10^{2}	Cm	2.5×10^{1}	1.0×10^{3}
Cd	2.0×10^{2}	2.0×10^{3}	Cf	2.5×10^{1}	1.0×10^{3}

where

$ETF_{i7}(t)$ = environmental transport factors for the drinking water pathway (g/yr),

DF_7 = annual intake of drinking water (410 L/yr),

FDW = fraction of drinking water from site (dimensionless),

$WSR_{i1}(t)$ = water/soil concentration ratios for well water (g/L),

$WSR_{i2}(t)$ = water/soil concentration ratios for surface water (g/L), and

FD1 = fraction of well water used for drinking (balanced from surface water; 1.0, dimensionless).

The annual intake of drinking water is a dietary factor from Table D.2. The water/soil concentration ratios are calculated using the models described in Appendix E.

D.3 REFERENCES

Eckerman, K.F., A.B. Wolbarst, and A.C.B. Richardson, 1988, *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion,* EPA-520/1-88-020, Federal Guidance Report No. 11, prepared by Oak Ridge National Laboratory, Oak Ridge, Tenn., for U.S. Environmental Protection Agency, Office of Radiation Programs, Washington, D.C.

International Commission on Radiological Protection, 1979-1982, *Limits for Intakes of Radionuclides by Workers,* a report of Committee 2 of the International Commission on Radiological Protection, adopted by the Commission in July 1978, ICRP Publication 30, Part 1 (and Supplement), Part 2 (and Supplement), Part 3 (and Supplements A and B), and Index.

U.S. Department of Energy, 1988, *Internal Dose Conversion Factors for Calculation of Dose to the Public,* DOE/EH-0071, Assistant Secretary for Environment, Safety and Health (July).

APPENDIX E

WATER PATHWAY FACTORS

APPENDIX E: WATER PATHWAY FACTORS

Water pathway factors are components of the environmental transport factors for water-dependent ingestion pathways. A water-dependent ingestion pathway can be divided into two segments: (1) a water pathway segment, extending from the contaminated zone to a point where transport through the food chain begins (a well or surface water body), and (2) a food chain pathway segment, extending from the point of entry of a radionuclide from water into the food chain to a point of human exposure. Transport through the water pathway segment is characterized by a water/soil concentration ratio, defined as the ratio of the concentration of the radionuclide in water used for drinking, irrigation, or livestock water to the concentration in the contaminated zone. Transport through the food chain pathway segment is characterized by a water exposure factor, defined as the ratio of the quantity of a radionuclide ingested annually to the concentration in water used for drinking, irrigation, or livestock water. The environmental transport factor for the water-dependent ingestion pathway can be expressed as a product of a water exposure factor and a water/soil concentration ratio:

$$ETF_{ipqr}(t) = WEF_{ipqr}(t) \times WSR_{ir}(t) \qquad (E.1)$$

where

$ETF_{ipqr}(t) = E_{ipqr}(t)/S_i(t)$ = environmental transport factor at time t for the i^{th} principal radionuclide transported through the pqr^{th} ingestion pathway (g/yr) -- $E_{ipqr}(t)$ is the rate of ingestion (pCi/yr) at time t of the i^{th} principal radionuclide transported through the pqr^{th} pathway from the contaminated zone to a point of human exposure and $S_i(t)$ is the average concentration (pCi/g) at time t of the i^{th} principal radionuclide in the contaminated zone,

p, q, r = pathway indices -- the indices p and q identify the food chain segment of the pathway (see Table 2.1 and Equation D.1) and the index r identifies the water pathway segment, i.e., the segment from the contaminated zone to well water (r = 1) or surface water (r = 2),

$WEF_{ipqr}(t) = E_{ipqr}(t)/W_{ir}(t)$ = water exposure factor at time t for the i^{th} principal radionuclide transported through the pqr^{th} pathway from the point of water use to the point of exposure (L/yr) -- $E_{ipqr}(t)$ is the rate of ingestion (pCi/yr) at time t of the i^{th} principal radionuclide transported through the pqr^{th} pathway, and $W_{ir}(t)$ is the concentration in water (pCi/L) at time t of the i^{th} principal radionuclide transported through the r^{th} water pathway segment at the point of entry into the pq^{th} food chain, and

$WSR_{ir}(t) = W_{ir}(t)/S_i(t)$ = water/soil concentration ratio at time t for the r^{th} water pathway segment (g/L) -- $W_{ir}(t)$ is the concentration in water (pCi/L) at time t of the i^{th} principal radionuclide transported through the r^{th} water pathway segment at the point of entry into a food chain pathway segment, and $S_i(t)$ is the average concentration (pCi/g) at time t of the i^{th} principal radionuclide in the contaminated zone.

The water exposure factor $WEF_{ipqr}(t)$ is discussed in Appendix D. The water/soil concentration ratio $WSR_{ir}(t)$ is discussed as follows.

A WSR is determined by the rate at which a radionuclide is leached from the contaminated zone, the time for this radionuclide to be transported along the water pathway from the boundary of the contaminated zone to the point of water use, and the dilution that occurs along this pathway. The model for estimating radionuclide leaching and formulas for calculating the leach rate are given in Section E.1. Formulas that relate the radionuclide concentration in water at the point of use to parameters that characterize the leaching and transport processes are derived in Section E.2.

E.1 RADIONUCLIDE LEACHING FROM THE CONTAMINATED ZONE

Radionuclides adsorbed in soil are subject to leaching by infiltrating water. Radionuclide leaching from the contaminated zone is the source for groundwater contamination. Therefore, the first step in calculating radionuclide concentrations in groundwater is to estimate the leaching of radionuclides from the contaminated zone.

A sorption-desorption, ion-exchange leaching model is used in the RESRAD code. This model is characterized by a nuclide-dependent, first-order leach rate constant, L_i, which is defined as the fraction of available radionuclide i leached out per unit time. The radionuclide release rate (source strength, in Ci/yr), $\dot{R}_i(t)$, can be written as (Yu 1987):

$$\dot{R}_i(t) = L_i \, \rho_b^{(cz)} \, A \, T(t) \, S_i(t) \tag{E.2}$$

where

L_i = leach rate for radionuclide i (yr^{-1}),
$\rho_b^{(cz)}$ = bulk density of the contaminated zone (kg/m^3),
A = area of the contaminated zone (m^2),
$T(t)$ = thickness of the contaminated zone at time t (m), and
$S_i(t)$ = average concentration of the i^{th} principal radionuclide in the contaminated zone available for leaching at time t (pCi/kg).

The first-order leach rate constant used in the current version of RESRAD is a time-independent radionuclide leach rate constant that is estimated based on the soil residence time for the initial thickness of the contaminated zone. A time-dependent radionuclide leach rate constant for radionuclide i, $L_i(t)$, may be written as

$$L_i(t) = \frac{I}{\theta^{(cz)} \; T(t) \; R_{d_i}^{(cz)}} \tag{E.3}$$

where

I = infiltration rate (m/yr),
$\theta^{(cz)}$ = volumetric water content of the contaminated zone, and
$R_{d_i}^{(cz)}$ = retardation factor in the contaminated zone for radionuclide i (dimensionless).

The infiltration rate is given by

$$I = (1 - C_e)\left[(1 - C_r)P_r + I_{rr}\right] \tag{E.4}$$

where

C_e = evapotranspiration coefficient (0.7, dimensionless),

C_r = runoff coefficient (0.6, dimensionless),

P_r = precipitation rate (annual rainfall, 1.0 m/yr), and

I_{rr} = irrigation rate (0 m/yr).

To calculate the infiltration rate, the average evapotranspiration coefficient is used rather than the average evapotranspiration rate (see Geraghty et al. [1973] for U.S. average). Using the average evapotranspiration rate does not take into account the correlation between precipitation and evapotranspiration and, for arid regions, can give a spurious negative infiltration rate. The evapotranspiration coefficient is related to the evapotranspiration rate by the formula $C_e = E_t/[(1 - C_r)P_r + I_{rr}]$. Runoff coefficients for a specific site may be obtained from Table E.1. It is assumed that irrigation water is controlled by ditching or by the duration of application so that none will be lost by runoff. The default irrigation rate is for humid regions where irrigation is unnecessary; an appropriate generic value for arid regions would be $I_{rr} = 1$ m/yr. Site-specific values for the precipitation and irrigation rates should be used whenever possible.

The volumetric water content of the contaminated zone, $\theta^{(cz)}$, is the product of the saturated water content of the contaminated zone, $\theta_{sat}^{(cz)}$, and the saturation ratio of the contaminated zone, $R_{sat}^{(cz)}$. The saturated water content is the water content when the soil material is saturated. Hence, θ_{sat}

TABLE E.1 Runoff Coefficient Values

Type of Area	Coefficient	Value
Agricultural Environment[a]		
Flat land, with average slopes of 0.3 to 0.9 m per mile	c_1	0.3
Rolling land, with average slopes of 4.6 to 6.1 m per mile	c_1	0.2
Hilly land, with average slopes of 46 to 76 m per mile	c_1	0.1
Open sandy loam	c_2	0.4
Intermediate combinations of clay and loam	c_2	0.2
Tight, impervious clay	c_2	0.1
Woodlands	c_3	0.2
Cultivated lands	c_3	0.1
Urban Environment		
Flat, residential area — about 30% imprevious	C_r	0.4
Moderately steep, residential area — about 50% impervious	C_r	0.65
Moderately steep, built-up area — about 70% impervious	C_r	0.8

[a] The runoff coefficient for an agricultural environment is given by $C_r = 1 - c_1 - c_2 - c_3$.

Source: Data from Gray (1970).

equals p_t, where p_t is the total porosity of the soil material, that is*

$$\theta_{sat} = p_t \tag{E.5}$$

The saturation ratio, R_s, is defined as the ratio of θ over θ_{sat}, that is

$$R_s = \theta/\theta_{sat} = \theta/p_t \tag{E.6}$$

When the medium is saturated, R_s equals unity. Under unsaturated infiltration conditions, the saturation ratio is a function of the infiltration rate, the saturated hydraulic conductivity, and the texture of the soil. The saturation ratio can be estimated using the following equation (Clapp and Hornberger 1978):

$$R_s = \left(\frac{I}{K_{sat}}\right)^{\frac{1}{2b + 3}} \tag{E.7}$$

where

I = infiltration rate (m/yr),

K_{sat} = saturated hydraulic conductivity (m/yr), and

b = soil-specific exponential parameter (dimensionless).

Representative values of K_{sat}, θ_{sat}, and b for various soil textures are listed in Table E.2.

---

*The superscript is omitted for a general definition. The definition applies for all zones.

TABLE E.2 Representative Values of Saturated Hydraulic Conductivity, Saturated Water Content, and the Soil-Specific Exponential Parameter

Texture	Hydraulic Conductivity, K_{sat} (m/yr)	Saturated Water Content, θ_{sat}	Soil-Specific Exponential Parameter, b
Sand	5.55×10^3	0.395	4.05
Loamy sand	4.93×10^3	0.410	4.38
Sandy loam	1.09×10^3	0.435	4.90
Silty loam	2.27×10^2	0.485	5.30
Loam	2.19×10^2	0.451	5.39
Sandy clay loam	1.99×10^2	0.420	7.12
Silty clay loam	5.36×10^1	0.477	7.75
Clay loam	7.73×10^1	0.476	8.52
Sandy clay	6.84×10^1	0.426	10.40
Silty clay	3.26×10^1	0.492	10.40
Clay	4.05×10^1	0.482	11.40

Source: Data from Clapp and Hornberger (1978).

The retardation factor for radionuclide i, R_{d_i}, is the ratio of average pore water velocity to radionuclide transport velocity. Assuming that the adsorption-desorption process can be represented with a linear Freundlich isotherm, the retardation factor can be calculated with the formula (Yu 1987)

$$R_{d_i} = 1 + \frac{\rho_b K_{d_i}}{\theta} = 1 + \frac{\rho_b K_{d_i}}{p_t R_s} \qquad \text{(E.8)}$$

where

ρ_b = bulk soil density (g/cm^3),

K_{d_i} = distribution coefficient for the ith principal radionuclide (cm^3/g), and

θ = volumetric water content (dimensionless).

The distribution coefficient is the radionuclide equilibrium concentration ratio of the adsorbed radionuclide (in soil) to the desorbed radionuclide (in water). Representative distribution coefficients are given in Tables E.3 through E.6.

E.2 WATER/SOIL CONCENTRATION RATIOS IN TERMS OF NUCLIDE WATER-TRANSPORT PARAMETERS

A water/soil concentration ratio can be expressed in terms of functions that characterize the source terms and transport processes and are applicable for both simple and complex hydrogeological strata. By introducing simplifying approximations for the functional form of the breakthrough curve* that are generally applicable even for complex hydrogeological structures, the transport and source functions can be specified by a small number of nuclide water-transport parameters, and various models can be used to derive relations between these parameters and measurable quantities. The analysis is applicable to either the groundwater or surface water pathway; hence, in the following derivation, the subscript r used to identify different water pathways has been omitted in order to simplify the expressions.

Let $W_i^s(t')$ be the average concentration at time t' of the i^{th} radionuclide in water that has percolated through the contaminated zone, measured at the lower boundary of the contaminated zone. This source of groundwater contamination will result in a concentration $W_i(t)$ at time t ($t \geq t'$) of the i^{th} radionuclide in water at the point of use (e.g., a well or surface water body). One may express the relation between the source concentration $W_i^s(t')$

---

*The breakthrough curve is the concentration of a radionuclide in water at the point of use as a function of time.

water/soil concentration ratio for the i^{th} principal radionuclide in the contaminated effluent water from the contaminated zone:

$$Q_i(t') = \theta^{(cz)} W_i^s(t')/[\rho_b^{(cz)} S_i(t')] \tag{E.13}$$

where

$W_i^s(t')$ = average volume concentration of the i^{th} principal radionuclide in effluent contaminated water at the beginning of the water pathway as it crosses the lower boundary of the contaminated zone at time t' (Ci/m^3), and

$S_i(t')$ = average concentration of the i^{th} principal radionuclide in the soil in the contaminated zone at time t', including contributions from ingrowth from other principal radionuclides and the decrease due to removal by radioactive decay and leaching (Ci/kg).

$W_i^s(t')$ can be estimated as the ratio of the source release rate $\dot{R}_i(t')$ and the water flow rate IA. That is,

$$W_i^s(t') = \dot{R}_i(t')/IA. \tag{E.14}$$

Substituting Equations E.2 and E.14 into Equation E.13 yields

$$Q_i(t') = \frac{1}{R_{d_i}^{(cz)}} . \tag{E.15}$$

Thus, the leaching ratio for a sorption-desorption, ion-exchange leaching model is a constant.

If the change in the hydrogeological environment within the time horizon is small, the nuclide water-transport function will depend only on the time difference t - t', and the nuclide water-transport function will have the functional form $G_i(t,t') = G_i(t - t')$. If it is assumed that all decay products migrate at the same rate as the parent radionuclide, the nuclide water-transport function can be written as

$$G_i(t,t') = ID_i(t,t')\, g_i(t - t') \tag{E.16}$$

where

$ID_i(t,t')$ = ingrowth and decay correction factor for radionuclide i from time t' to time t (dimensionless)

$= [\sum_j S_j(0)\, ID_{ji}(t)]/[\sum_j S_j(0)\, ID_{ji}(t')]$, and

$g_i(t - t')$ = ingrowth- and decay-independent nuclide water-transport function for the i^{th} principal radionuclide (1/yr).

The term $ID_{ji}(t)$ is defined in Appendix F.

By using the source factor $SF_{ji}(t)$ (defined in Appendix F), the average concentration of the i^{th} principal radionuclide in the contaminated zone at time t can be written as

$$S_i(t) = \sum_j S_j(0)\, SF_{ji}(t) \tag{E.17}$$

In order to simplify Equation E.12, it is further assumed for the groundwater pathway that the source factor $SF_{ji}(t)$ can be replaced by an ingrowth and decay factor $ID_{ji}(t)$ and a leaching factor $L_{f_i}(t)$, that is

$$SF_{ji}(t) = ID_{ji}(t)\, L_{f_i}(t) \tag{E.18}$$

The leaching factor may be written as

$$L_{f_i}(t) = \exp(-\, L_i t) \tag{E.19}$$

where L_i is the leach rate for a first-order, ion-exchange leaching mechanism as defined in Section E.1.

The expression for the water/soil concentration ratios (Equation E.12) then reduces to

$$WSR_i(t) = \frac{\rho_b^{(cz)}}{\theta^{(cz)}\, L_{f_i}(t)} \int_0^t g_i(t - t')\, Q_i(t')\, L_{f_i}(t')\, dt' \tag{E.20}$$

Equation E.20 is used in this manual for deriving formulas for estimating the concentrations of radionuclides in water used for drinking or irrigation.

The general form of the nuclide water-transport function is such that, if a short pulse of contaminated water is released from the contaminated zone during the time period $0 \le t' \le \delta t'$, there will be a time interval Δt_i (the breakthrough time) before the contamination reaches the point of use. Following breakthrough, the concentration of the i^{th} radionuclide at the point of use will increase to a maximum and then decrease back to a near-zero value in a time interval δt_i. This time is referred to as the rise time because it is also the time for the contamination at the point of use to increase from zero to a maximum value when the pulse release is replaced by a steady release starting at the same time.* The pulse of contamination at the point of use may be idealized as a rectangular pulse of duration δt_i. Thus, the nuclide water-transport function may be approximated by a function of the form

$$\begin{aligned} g_i(t) &= 0, && t \le \Delta t_i \\ &= g_i, && \Delta t_i < t < \Delta t_i + \delta t_i \\ &= 0, && \Delta t_i + \delta t_i \le t \end{aligned} \tag{E.21}$$

where

Δt_i = breakthrough time for i^{th} principal radionuclide (yr),

$g_i = f_i/\delta t_i$ = rate of change of the concentration of i^{th} principal radionuclide at the point of water use for a unit input pulse (1/yr),

δt_i = rise time for i^{th} principal radionuclide (yr), and

f_i = steady-state dilution factor = ratio of average concentration of i^{th} principal radionuclide in water at the point of use to the average concentration in effluent water at the contaminated zone boundary for a steady-state release when there is no ingrowth or decay (dimensionless).

Using Equations E.15, E.20, and E.21, the time dependence of the water/soil concentration ratio for the groundwater pathway, after being

---

*The rise time δt_i and the release pulse duration $\delta t'$ are independent quantities when the pulse duration is small; δt_i is determined by the transport properties of the groundwater pathway and will remain finite as $\delta t' \to 0$.

corrected for radionuclide leaching, is

$$
\begin{aligned}
WSR_i(t) &= 0, && t \le \Delta t_i \\
&= \frac{\rho_b f_i}{\theta^{(cz)} R_{d_i}^{(cz)} \delta t_i L_i} \left[1 - \exp\left[-\left(L_i(t - \Delta t_i)\right)\right]\right], && \Delta t_i < t \le \Delta t_i + \delta t_i \\
&= \frac{\rho_b f_i}{\theta^{(cz)} R_{d_i}^{(cz)} \delta t_i L_i} \left[\exp[-L_i(t - \Delta t_i - \delta t_i)] - \exp\left[-\left(L_i(t - \Delta t_i)\right)\right]\right], && \Delta t_i + \delta t_i < t
\end{aligned}
\qquad (E.22)
$$

where the various parameters are as defined before.

If the rise time is negligibly small ($\delta t_i = 0$), Equation E.22 reduces to

$$
\begin{aligned}
WSR_i(t) &= 0, && t \le \Delta t_i \\
&= \frac{\rho_b^{(cz)} f_i}{\theta^{(cz)} R_{d_i}^{(cz)}} \exp\left[-\left(L_i(t - \Delta t_i)\right)\right], && \Delta t_i < t
\end{aligned}
\qquad (E.23)
$$

Formulas for calculating the transport parameters (dilution factors, f_i; breakthrough times, Δt_i; and rise times, δt_i) for simple hydrogeological systems are derived in Section E.3.

E.3 WATER TRANSPORT PARAMETERS

E.3.1 On-site Groundwater Pathway

The water transport parameters for radionuclide i are the breakthrough time Δt_i (the time following the radiological survey at which radionuclides first appear in the water at the point of use), the rise time δt_i (the time following the breakthrough time for the radionuclide concentration in the water to attain a maximum value), and the dilution factor f_i (the ratio between the concentration in the water at the point of use to the concentration in the infiltrating water as it leaves the contaminated zone). Two models are used for calculating these parameters: a mass-balance (MB) model and a nondispersion (ND) model. In the MB model, it is assumed that all of the radionuclides released from the contaminated zone are withdrawn through the well. In the ND model, it is assumed that the dispersivity is nil, the unsaturated zone consists of one or more horizontal homogeneous strata, the saturated zone is single homogeneous stratum, and water withdrawal introduces only a minor perturbation in the water flow. These assumptions lead to a pattern of flow lines from which the dilution factor can be estimated by geometric considerations.

The user has the option to select which model to use. Usually, the MB model is used for smaller contaminated areas (e.g., 1,000 m^2 or less), and the ND model is used for larger areas. The breakthrough times are the same for both models; the rise times and dilution factors are different.

E.3.1.1 Breakthrough Time

The well from which water is withdrawn for domestic use or irrigation is conservatively assumed to be located either in the center of the contaminated zone (in the MB model) or at the downgradient edge of the contaminated zone (in the ND model). For either location, radionuclides are assumed to enter the well as soon as they reach the water table; hence, the transport time through the aquifer is nil and the breakthrough time is equal to the transport time through the unsaturated zone, which is the sum of the times for a radionuclide to be transported through those strata of the zone that lie below the contaminated zone.

$$\Delta t_i = \sum_{m=1}^{n+1} \Delta t_{im}^{(uz)}$$

where

n = number of distinct horizontal strata in the unsaturated below the contaminated zone at time 0, and

$\Delta t_{im}^{(uz)}$ = transport time for the i^{th} principal radionuclide through t stratum (yr).

The upper bound, n+1, of the summation in Equation E.24 is for $(n+1)^{th}$ unsaturated stratum created by the decrease in the water table. T thickness of this $(n+1)^{th}$ stratum, Δz_{n+1}, is equal to the product of th table drop rate, v_{wt}, and time. That is,

$$\Delta z_{n+1} = v_{wt} \times t \qquad \text{(E.2}$$

where v_{wt} = water table drop rate (0 m/yr).

The hydrogeological and hydrogeochemical properties of the $(n+1)^{th}$ unsaturated stratum are assumed to be the same as those of the saturate stratum. The RESRAD code allows up to five horizontal strata below th contaminated zone, i.e., $n \leq 5$. If $n = 0$, the contaminated zone extends dow to the aquifer.

The formula for the transport time is:

$$\Delta t_{im}^{(uz)} = \Delta z_m \, R_{d_{im}}^{(uz)} \, p_{e_m}^{(uz)} \, R_{s_m}^{(uz)}/I \qquad (E.26)$$

where

Δz_m = thickness of the m^{th} stratum (Δz_1 = 4 m, Δz_2, ... , Δz_5 = 0)

$R_{d_{im}}^{(uz)}$ = retardation factor of the i^{th} principal radionuclide in the m^{th} stratum of the unsaturated zone,

$p_{e_m}^{(uz)}$ = effective porosity of the m^{th} stratum of the unsaturated zone (0.2, dimensionless), and

$R_{s_m}^{(uz)}$ = saturation ratio of the m^{th} stratum (0.5, dimensionless).

The unsaturated zone retardation factors, $R_{d_{im}}^{(uz)}$, are calculated by the formula

$$R_{d_{im}}^{(uz)} = 1 + \rho_{b_m}^{(uz)} \, K_{d_{im}}^{(uz)}/\left(p_{t_m}^{(uz)} \, R_{s_m}^{(uz)}\right) \qquad (E.27)$$

where

$\rho_{b_m}^{(uz)}$ = bulk soil density in the m^{th} stratum (1.6 g/cm^3),

$K_{d_{im}}^{(uz)}$ = distribution coefficient for the i^{th} principal radionuclide in the m^{th} stratum (cm^3/g), and

$p_{t_m}^{(uz)}$ = total porosity of the m^{th} stratum (0.4, dimensionless).

The saturation ratio, $R_{s_m}^{(uz)}$, can be determined using Equation E.7.

Data that may be used to estimate breakthrough times when site-specifi measurements are not available are given in Tables E.2 through E.7, as follows: saturated water contents (porosities), soil-specific exponential parameters, and saturated hydraulic conductivities in Table E.2; distribution coefficients in Tables E.3 through E.6; and total porosities and effective porosities in Table E.7.

Hydraulic conductivity is a critical hydrological parameter that can differ by several orders of magnitude from one site to another. Site-specific values should be used for deriving soil guidelines. Default values of the distribution-coefficient values used in the RESRAD code are the values for soils and clays from Table E.3. The default distribution coefficient for technetium is zero. The default distribution coefficient for plutonium in Table E.3 is used for all transuranic elements. For other elements not listed in Table E.3, the default value is the value of the nearest listed element in the same column of the periodic table. If there are no other elements in the same column, a zero value is assumed.

Distribution coefficients depend strongly on soil type, the pH and Eh of the soil, and the presence of other ions (see Tables E.4 through E.6). Thus, the uncertainty introduced by using default values for the distribution coefficients can be very large. This is a critical matter, especially in those cases where the water-dependent pathways are the dominant contributors to the total dose/source concentration ratios; the single-radionuclide soil guidelines will be directly proportional to the distribution coefficients for the contaminated zone. Default values for the distribution coefficients are provided only for the purpose of obtaining preliminary estimates; site-specific values should be used for deriving soil guidelines whenever possible.

TABLE E.7 Representative Porosity Values

Material	Total Porosity, p_t Range	Total Porosity, p_t Arithmetic Mean	Effective Porosity, p_e Range	Effective Porosity, p_e Arithmetic Mean
Sedimentary Material				
Sandstone (fine)			0.02 - 0.40	0.21
Sandstone (medium)	0.14 - 0.49	0.34	0.12 - 0.41	0.27
Siltstone	0.21 - 0.41	0.35	0.01 - 0.33	0.12
Sand (fine)	0.25 - 0.53	0.43	0.01 - 0.46	0.33
Sand (medium)	-	-	0.16 - 0.46	0.32
Sand (coarse)	0.31 - 0.46	0.39	0.18 - 0.43	0.30
Gravel (fine)	0.25 - 0.38	0.34	0.13 - 0.40	0.28
Gravel (medium)	-	-	0.17 - 0.44	0.24
Gravel (coarse)	0.24 - 0.36	0.28	0.13 - 0.25	0.21
Silt	0.34 - 0.51	0.45	0.01 - 0.39	0.20
Clay	0.34 - 0.57	0.42	0.01 - 0.18	0.06
Limestone	0.07 - 0.56	0.30	~0 - 0.36	0.14
Wind-Laid Material				
Loess	-	-	0.14 - 0.22	0.18
Eolian sand	-	-	0.32 - 0.47	0.38
Tuff	-	-	0.02 - 0.47	0.21
Igneous Rock				
Weathered granite	0.34 - 0.57	0.45	-	-
Weathered gabbro	0.42 - 0.45	0.43	-	-
Basalt	0.03 - 0.35	0.17	-	-
Metamorphic Rock				
Schist	0.04 - 0.49	0.38	0.22 - 0.33	0.26

Source: Data from McWorter and Sunada (1977).

The distance from the ground surface to the water table, $D_{wt}(t)$, at time t is

$$D_{wt}(t) = C_d(t) + T(t) + \sum_{m=1}^{n+1} \Delta z_m \tag{E.28}$$

where

$C_d(t)$ = cover depth at time t (m),

$T(t)$ = thickness of contaminated zone at time t (m), and

Δz_m is as defined in Equations E.25 and E.26.

When the RESRAD code is being run, the code will check Equation E.28 for the accuracy of the input data. If Equation E.28 is not satisfied for $t = 0$ -- i.e., incorrect input data for $D_{wt}(0)$, $C_d(0)$, $T(0)$, or Δz_m -- the user will have to modify the input data before the code can be executed. In the current version of RESRAD, the water table is assumed to be below the contaminated zone, i.e., $D_{wt}(0) \geq C_d(0) + T(0)$.

E.3.1.2 Rise Time

When the well is located at the downgradient edge of the contaminated zone, as shown in Figure E.1 (i.e., if the ND model is used), the rise time is given by the formula

$$\begin{aligned} \delta t_i &= \tau_{\ell_i}, && \zeta \leq d_w,\ D_{wt}(t) \geq C_d(t) + T(t) \\ &= \tau_{\ell_i}/(\zeta/d_w), && \zeta > d_w,\ D_{wt}(t) \geq C_d(t) + T(t) \\ &= 0, && D_{wt}(t) < C_d(t) + T(t) \end{aligned} \tag{E.29}$$

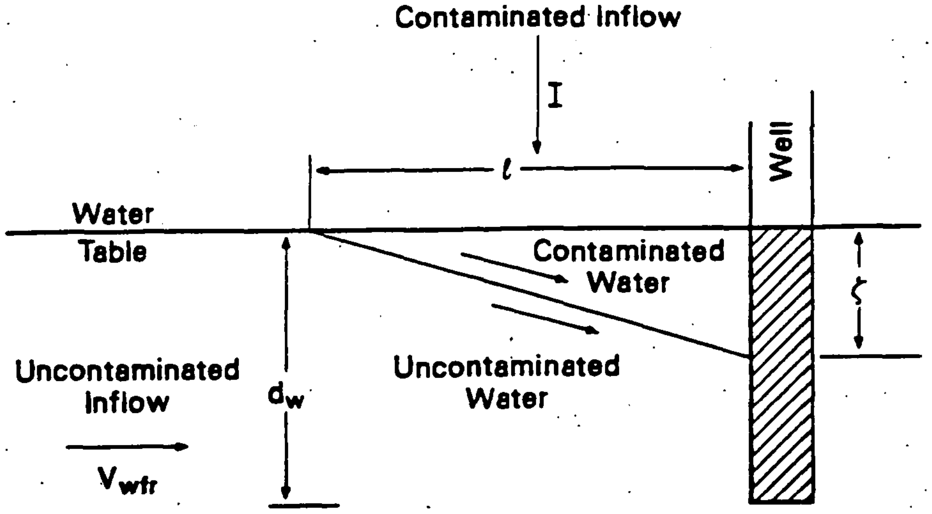


FIGURE E.1 Dilution of Contaminated Inflow by Uncontaminated Inflow in the Nondispersion Model for a Well Adjacent to the Contaminated Area

where

$\zeta = (I/V_{wfr})\ell$ = distance from water table to lower boundary of contamination in aquifer at the downgradient edge of the contaminated zone (m),

I = infiltration rate (m/yr),

J_x = hydraulic gradient (0.02, dimensionless),

$K_s^{(sz)}$ = saturated hydraulic conductivity of saturated zone (100 m/yr),

ℓ = length of contaminated zone parallel to the hydraulic gradient (maximum distance from upgradient edge to downgradient edge parallel to the hydraulic gradient, 100 m),

d_w = distance of well intake below the water table (10 m),

$\tau_{\ell_i} = p_e^{(sz)} R_{d_i}^{(sz)} \ell / V_{wfr}$ = time for the i^{th} principal radionuclide to be transported from the upgradient edge to the downgradient edge of the contaminated zone (yr),

$R_{d_i}^{(sz)}$ = retardation factor for the ith principal radionuclide ..

saturated zone (dimensionless), and

$p_e^{(sz)}$ = effective porosity of aquifer (0.2, dimensionless).

Equation E.29 is used only in the ND model. Representative porosities are given in Tables E.2 and E.7. The infiltration rate is given by Equation E.4. Representative saturated hydraulic conductivities are given in Table E.2. Representative distribution coefficients are given in Tables E.3 through E.6.

If the well is in the center of the contaminated zone (i.e., if the MB model is used), the rise time is assumed to be zero. That is, for the MB model,

$$\delta t_i = 0 \ . \qquad (E.30)$$

E.3.1.3 Dilution Factor

It is assumed that the water flow is vertically downward from the bottom of the contaminated zone to the water table; hence, no dilution will occur in the unsaturated zone. The dilution in the saturated zone is estimated using the conservative assumption that the dispersivity is zero.

The dilution factor for the MB model is a radionuclide-independent factor given by the formula

$$\begin{aligned} f_i &= A\,I/U_w, & A\,I &< U_w \\ &= 1, & A\,I &\geq U_w \end{aligned} \qquad (E.31)$$

where

A = area of contaminated zone (m^2),

I = infiltration rate (m/yr), and

U_w = annual volume of water withdrawn from well for use by a single individual (150 m^3/yr).

The infiltration rate is calculated by Equation E.4.

For a well located at the downgradient edge of the contaminated zone, the dilution factor for the ND model is given by the formula

$$\begin{aligned} f_i &= \zeta/d_w, & \zeta &< d_w \\ &= 1, & \zeta &\geq d_w, \end{aligned} \qquad (E.32)$$

where the parameters are as defined for Equation E.29.

The dilution factors for a well that is located away from the edge of the contaminated zone can be significantly smaller than dilution factors estimated by Equations E.31 and E.32 for a well located in or immediately adjacent to the contaminated zone. The one-dimensional models used to obtain Equations E.31 and E.32 cannot be used to estimate this reduction; a two- or three-dimensional model, such as the FEWA and FEMA codes developed by Yeh and Huff (1983, 1985), must be used to estimate a more realistic dilution factor. In addition, the assumption of no dispersivity may be unnecessarily conservative if the well is not located very close to the contaminated zone.

E.3.2 Surface Water Pathway

The surface water pathway will consist of an on-site groundwater pathway segment that extends to the edge of the contaminated zone, an off-site groundwater pathway segment that extends from the edge of the contaminated zone to a location where surface seepage occurs, and a surface water segment in which the contaminated groundwater is mixed with uncontaminated surface water. Contamination of the surface water can also occur as a consequence of erosion as contaminated soil is deposited in a nearby streambed or pond. This pathway is not included in the current version of RESRAD.

The derived water transport parameters for the surface water pathway are identified by primes — $\Delta t_i'$, $\delta t_i'$, and f_i' -- in order to distinguish them from the corresponding groundwater pathway parameters — Δt_i, δt_i, and f_i.

E.3.2.1 Breakthrough Time

The breakthrough time for the surface water pathway is assumed to be the same as that for the groundwater pathway, i.e.,

$$\Delta t_i' = \Delta t_i \tag{E.33}$$

where Δt_i is calculated using Equations E.24 through E.27.

E.3.2.2 Rise Time

The rise time for the surface water pathway is assumed to be the same as that used in the ND model for the groundwater pathway, i.e.,

$$\delta t_i' = \delta t_i \tag{E.34}$$

where δt_i is given by Equation E.29.

E.3.2.3 Dilution Factor

The dilution factor for the surface water pathway is based on the assumption that the surface water is a pond and (1) the inflow and outflow for the pond are in steady-state equilibrium and (2) the annual inflow of radioactivity into the pond is equal to the annual quantity of radioactivity leached from the contaminated zone. With these assumptions, the dilution factor is the ratio of the annual volume of water that percolates through the contaminated zone to the annual total inflow of water into the pond. If, in addition, it is assumed that the infiltrating water flow is vertically downward, the dilution factor is given by the formula

$$f_i' = A/A_w \qquad \text{(E.35)}$$

where

A = area of contaminated zone (10^4 m^2) and
A_w = area of watershed (10^6 m^2).

E.4 REFERENCES

Baes, C.F., and R.D. Sharp, 1983, *A Proposal for Estimation of Soil Leaching and Leaching Constants for Use in Assessment Models,* Journal of Environmental Quality, *12*(1):17-28.

Clapp, R.B., and G.M. Hornberger, 1978, *Empirical Equations for Some Soil Hydraulic Properties,* Water Resources Research, *14*:601-604.

Gee, G.W., et al., 1980, *Interaction of Uranium Mill Tailings Leachate with Soils and Clay Liners,* NUREG/CR-1494; PNL-3381, prepared by Pacific Northwest Laboratory, Richland, Wash., for Office of Nuclear Regulatory Research, U.S. Nuclear Regulatory Commission, Washington, D.C.

Geraghty, J.J., et al., 1973, *Water Atlas of the United States,* Water Information Center, Port Washington, N.Y.

Gilbert, T.L., et al., 1983, *Pathways Analysis and Radiation Dose Estimates for Radioactive Residues at Formerly Utilized MED/AEC Sites,* ORO-832 (Rev.), prepared by Division of Environmental Impact Studies, Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations, Oak Ridge, Tenn. (March 1983 [reprinted with corrections: January 1984]).

Gray, D.M. (ed.), 1970, *Handbook on the Principles of Hydrology,* Water Information Center, Inc., Port Washington, N.Y.

Isherwood, D., 1981, *Geoscience Data Base Handbook for Modeling a Nuclear Waste Repository,* NUREG/CR-0912, Vols. 1 and 2, U.S. Nuclear Regulatory Commission (Jan.).

McWorter, D.B., and D.K. Sunada, 1977, *Ground-Water Hydrology and Hydraulics,* Water Resources Publications, Fort Collins, Colo. (as cited in Codell and Duguid [1983]).

Nuclear Safety Associates, Inc., 1980, *Comparison of Alternatives for Long-Term Management of High-Level Radioactive Waste at the Western New York Nuclear Service Center,* prepared by Nuclear Safety Associates, Inc., Bethesda, Md., for Argonne National Laboratory, Argonne, Ill., Appendix IIIC (Sept.).

Rancon, D., 1973, *The Behavior in Underground Environments of Uranium and Thorium Discharge by the Nuclear Industry,* in Proceedings, Environmental Behavior of Radionuclides Released in the Nuclear Industry, International Atomic Energy Agency, Vienna, Austria, pp. 333-346.

Staley, G.B., G.P. Turi, and D.L. Schreiber, 1979, *Radionuclide Migration from Low-Level Waste: A Generic Overview,* in M.W. Carter et al. (eds.), Management of Low-Level Radioactive Waste, Pergamon Press, New York, pp. 1041-1072.

U.S. Nuclear Regulatory Commission, 1980, *Final Generic Environmental Impact Statement on Uranium Milling -- Vol. 1, Summary and Text; Vol. 2, Appendices A-F; Vol. 3, Appendices G-V,* NUREG-0706, Office of Nuclear Material Safety and Safeguards, Washington, D.C.

Yeh, G.T., and D.D. Huff, 1983, *FEWA: A Finite Element Model of Water Flow through Aquifers,* ORNL-5976, Oak Ridge National Laboratory, Oak Ridge, Tenn.

Yeh, G.T., and D.D. Huff, 1985, *FEMA: A Finite Element Model of Material Transport through Aquifers,* ORNL-6063, Oak Ridge National Laboratory, Oak Ridge, Tenn.

Yu, C., 1987, *Modeling of Low-Level-Waste Disposal for Environmental Impact Analysis,* in Waste Management '87, Proceedings of the Symposium, Tucson, Ariz., March 1-5, pp. 385-390.

APPENDIX F

SOURCE FACTORS

APPENDIX F: SOURCE FACTORS

The time dependence of dose/source ratios due to radioactive ingrowth and decay and infiltration leaching is taken into account by introducing source factors, $SF_{ij}(t)$. Radionuclides are divided into two groups: those with half-lives longer than 1 year (principal radionuclides) and those with half-lives of 1 year or less (associated radionuclides) (see Table 3.1). It is assumed that the associated radionuclides are in secular equilibrium with their principal radionuclide and that the leach rate of the associated radionuclides is the same as the leach rate of their principal radionuclide. Hence, only the source factors for the principal radionuclides need to be calculated.

Let $S_{ii}(0)$ be the initial concentration of the i^{th} principal radionuclide and let $S_{ij}(t)$ be the concentration at time t of the j^{th} principal radionuclide that results from ingrowth from the initial inventory of the i^{th} principal radionuclide. $S_{ii}(t)$ is the concentration of the undecayed portion that is available for leaching of the i^{th} principal radionuclide. [$S_{ii}(t)$, which does not include any contribution from ingrowth, should be distinguished from $S_i(t) = \sum_j S_{ji}(t)$, which does include the contribution from ingrowth.] $S_{ij}(0) = 0$ unless $i = j$, and $S_{ij}(t) = 0$ unless $i = j$ or the j^{th} principal radionuclide is a decay product of the i^{th} principal radionuclide. The source factor is defined as

$$SF_{ij}(t) = S_{ij}(t)/S_i(0) \qquad (F.1)$$

The initial values of the source factors are $SF_{ij}(0) = 0$ unless $i = j$, $SF_{ii}(0) = 1$. For $t > 0$, $SF_{ij}(t) = 0$ unless $i = j$ or the j^{th} principal radionuclide is a decay product of the i^{th} principal radionuclide. When $i = j$, the inequality $1 > SF_{ii}(t) > 0$ will be satisfied for $t > 0$.

The source factor is a correction factor for the source term (radionuclide concentration in the contaminated zone), which includes the contribution from ingrowth and the removal due to radioactive decay and leaching. If leaching is negligible, the source factor reduces to a factor that accounts for ingrowth and decay only. This factor is called the ingrowth-and-decay factor, $ID_{ij}(t)$.

Formulas for calculating source factors and ingrowth-and-decay factors are given below. Because all of the branching radionuclides are associated radionuclides and the longest half-life is 18.7 days (see Table 3.1), the branching radionuclides are assumed to be in equilibrium with their principal radionuclide. Thus, branching need not be considered in deriving source factors and ingrowth-and-decay factors.

The equation for ingrowth and decay and leaching of a nonbranching chain, assuming a first-order ion-exchange leaching (see Appendix E), is

$$dA_k/dt = \lambda_k A_{k-1} - (\lambda_k + L_k) A_k \tag{F.2}$$

where

A_k = activity concentration of the k^{th} radionuclide in the chain (pCi/g),

$\lambda_k = \ln(2)/T_{\frac{1}{2},k}$ = radioactive decay constant of the k^{th} radionuclide (yr^{-1}),

$T_{\frac{1}{2},k}$ = radioactive decay half-life of the k^{th} radionuclide (yr), and

L_k = leach rate of the k^{th} radionuclide (yr^{-1}); the radionuclide leach rate is defined in Appendix E (Equation E.3).

The solution to Equation F.2 is

$$A_k(t) = \sum_{\ell=0}^{k} a_{k\ell} \times \exp[-(\lambda_\ell + L_\ell)t] \qquad (F.3)$$

where

$a_{00} = A_0(0)$;

$a_{k\ell} = [\lambda_k/(\lambda_k + L_k - \lambda_\ell - L_\ell)]\, a_{k-1,\ell}$, $\ell = 0,1, \ldots, k-1$; and

$a_{kk} = A_k(0) - \sum_{\ell=0}^{k-1} a_{k\ell}$.

The source factor $SF_{ij}(t)$ can be calculated as

$$SF_{ij}(t) = A_{\ell(ij)}(t)/A_0(0) \qquad (F.4)$$

where $\ell(ij)$ is the sequence number for the j^{th} principal decay product of the i^{th} principal radionuclide, numbered from the i^{th} principal radionuclide [$\ell(ii) \equiv 0$]. The ingrowth-and-decay factor $ID_{ij}(t)$ is the source factor $SF_{ij}(t)$ when the leach rates L_ℓ and L_k are set to zero.

DISTRIBUTION FOR ANL/ES-160; DOE/CH/8901

Internal

N. Beskid
C. Boggs
L.E. Boing
S.Y. Chen
J. Devgun
A.J. Dvorak
F.F. Gorup (2)
L.J. Habegger
K.J. Hong
L.D. Jensen
M.J. Jusko
K.S. Macal (5)
B.J. Micklich
G.D. Mosho
J.M. Peterson
M.J. Robinet
T.G. Surles
C. Yu (120)
A.J. Zielen
ANL Contract Copy
ANL Libraries
ANL Patent Department
ANL Technical Publications Services (3)

External

U.S. Department of Energy Office of Scientific and Technical Information (10)
Manager, U.S. Department of Energy, Chicago Operations Office
V. Adams, U.S. Department of Energy, Oak Ridge, Tenn.
A.J. Adduci, U.S. Department of Energy, Oakland, Calif.
H.W. Arrowsmith, Scientific Ecology Group, Inc., Oak Ridge, Tenn.
R. Atkin, U.S. Department of Energy, Oak Ridge, Tenn.
T. Aud, Babcock & Wilcox, Lynchburg, Va.
J.C. Baker, Jacobs Engineering Group, Inc., Pasadena, Calif.
E.J. Bentz, Jr., E.J. Bentz & Associates, Inc., Springfield, Va.
J. Berger, Oak Ridge Associated Universities, Oak Ridge, Tenn.
H. Brandon, Baker/TSA Inc., Coraopolis, Pa.
T. Brazley, U.S. Department of Energy, Germantown, Md.
H. Burkholder, Pacific Northwest Laboratory, Richland, Wash.
T.W. Burwinkle, Oak Ridge National Laboratory, Oak Ridge, Tenn.
M. Butler, National Energy Software Center, Argonne, Ill. (10)
C. Collantes, U.S. Department of Energy, Richland, Wash.
W. Cottrell, Oak Ridge National Laboratory, Oak Ridge, Tenn.
F.P. Crimi, General Electric Co., Shippingport, Pa.
J.C. Dehmel, Dehmel, Inc., Larchmont, N.Y.
W.F. Dennison, Rockwell International Corp., Canoga Park, Calif.
K.F. Eckerman, Oak Ridge National Laboratory, Oak Ridge, Tenn.
R. Eckhart, University of Cincinnati, Cincinnati, Ohio
K. Eger, General Electric Co., Shippingport, Pa.
J.P. Englert, Dames & Moore, Orchard Park, N.Y.
T. Frangos, U.S. Department of Energy, Germantown, Md.
J.R. Frazier, International Technologies, Oak Ridge, Tenn.
R. Freeberg, U.S. Department of Energy, Richland, Wash.
R. Garde, Los Alamos National Laboratory, Los Alamos, N.M.
R. Gardewing, Westinghouse Materials Company of Ohio, Cincinnati
J.H. Gibbons, Office of Technology Assessment, U.S. Congress
T.L. Gilbert, Chicago Center for Religion and Science, Chicago, Ill. (3)

G.N. Gnugno , U.S. Nuclear Regulatory Commission, Washington, D.C. (2)
M. Gonzalez, Lawrence Livermore National Laboratory, Livermore, Calif.
D. Goranson, U.S. Department of Energy, Richland, Wash.
R.J. Grandfield, U.S. Department of Energy, Miamisburg, Ohio
S. Green, Jacobs Engineering Group, Inc., St. Charles, Mo.
M. Haas, ENSA, Inc., Buffalo, N.Y.
W.R. Hansen, Los Alamos National Laboratory, Los Alamos, N.M.
J. Healey, Los Alamos National Laboratory, Los Alamos, N.M.
R. Hlavacek, M.K. Fergusen Company, St. Charles, Mo.
M. Hughes, Westinghouse Hanford Corporation, Richland, Wash.
N.E. Irizarry, Center for Energy and Environment Research, Mayaguez, P.R.
D.W. James, James LCE Associates, Minneapolis, Minn.
R. Jeager, EG&G Mound Applied Technologies, Miamisburg, Ohio
W.A. Jester, Pennsylvania State University, University Park
N.R. Johnson, Thermo Analytical, Inc., Albuquerque, N.M.
M.R. Jugan, U.S. Department of Energy, Oak Ridge, Tenn.
P. Kalb, Brookhaven National Laboratory, Upton, N.Y.
K. Karp, UNC Geotech, Grand Junction, Colo.
D.E. Kash, University of Oklahoma, Norman
W.E. Kennedy, Jr., Battelle Pacific Northwest Laboratory, Richland, Wash.
A. Kluk, U.S. Department of Energy, Germantown, Md.
D. Kocher, Oak Ridge National Laboratory, Oak Ridge, Tenn.
D. Kozlowski, U.S. Department of Energy, Germantown, Md.
K. Lawver, U.S. Department of Energy, St. Charles, Mo.
C. Lee, Applied Physics, Albuquerque, N.M.
B. Lewis, Northwestern University, Evanston, Ill.
G. Linsley, International Atomic Energy Agency, Vienna, Austria
D. MacFarlane, Los Alamos National Laboratory, Los Alamos, N.M.
C.E. Maggart, U.S. Department of Energy, Idaho Falls, Idaho
R. Marri, UNC Geotech, Grand Junction, Colo.
S. Marutzky, UNC Geotech, Grand Junction, Colo.
C. Massey, Oak Ridge National Laboratory, Oak Ridge, Tenn.
D. Mauer, Colorado Department of Health, Denver, Colo.
R. Meservey, EG&G Idaho, Inc., Idaho Falls, Idaho
M. Miller, Albuquerque, N.M.
B. Musgrave, Lawrence Livermore National Laboratory, Livermore, Calif.
S. Nair, Bechtel National, Inc., Oak Ridge, Tenn.
B.A. Napier, Battelle Pacific Northwest Laboratories, Richland, Wash.
J. Neff, U.S. Department of Energy, Columbus, Ohio
C. Nelson, U.S. Environmental Protection Agency, Washington, D.C.
R.R. Nelson, U.S. Department of Energy, St. Charles, Mo.
R.A. Nelson, Jacobs Engineering Group, Inc., St. Charles, Mo.
S. Neuder, U.S. Nuclear Regulatory Commission, Washington, D.C.
D. Padilla, U.S. Department of Energy, Los Alamos, N.M.
P.L. Piciulo, Ecology and Environment, Inc., Lancaster, N.Y.
J.W. Ray, Battelle Columbus Laboratories, Columbus, Ohio
C.J. Roberts, Dames & Moore, West Valley, N.Y.
R.C. Robertson, Bechtel National, Inc., Oak Ridge, Tenn.

J.C. Rodgers, Los Alamos National Laboratory, Los Alamos, N.M.
J. Russell, U.S. Environmental Protection Agency, Washington, D.C.
J.J. Schreiber, U.S. Department of Energy, Shippingport, Pa.
M. Scott, Louisiana State University, Baton Rouge, La.
V. Scovill, U.S. Nuclear Regulatory Commission Region IV, Denver, Colo. (2)
D. Seepo, U.S. Department of Energy, Schenectady, N.Y.
E. Shum, U.S. Nuclear Regulatory Commission, Washington, D.C.
L. Sims, Bechtel National, Inc., Oak Ridge, Tenn.
A.C. Smith, Ecotek, Inc., Erwin, Tenn.
L. Soholt, Los Alamos National Laboratory, Los Alamos, N.M.
J.K. Soldat, Battelle Pacific Northwest Laboratories, Richland, Wash.
A. Soong, U.S. Nuclear Regulatory Commission, Washington, D.C.
K.T. Stafford, Rockwell International Corp., Canoga Park, Calif.
M. Steinhouse, Battelle Columbus Laboratories, Columbus, Ohio
D. Swager, University of Cincinnati, Cincinnati, Ohio
J. Tarcza, Westinghouse - Hanford, Richland, Wash.
B. Thomas, International Technologies, Oak Ridge, Tenn.
J.C. Tseng, U.S. Department of Energy, Washington, D.C.
J. Turi, U.S. Department of Energy, Germantown, Md.
B. Walker, U.S. Department of Energy, Oak Ridge, Tenn.
A. Wallo III, U.S. Department of Energy, Germantown, Md. (15)
J.H.C Wang, Atomic Energy Council, Taipei, Taiwan
J. Williams, Bechtel National, Inc., Oak Ridge, Tenn.
D. Williamson, U.S. Department of Energy, Grand Junction, Colo.
B. Wissink, Health Physics Service, St. Paul, Minn.
D.E. Wood, Westinghouse Hanford Company, Richland, Wash.
Y.C. Yuan, Dames & Moore, West Valley, N.Y. (10)

Bibliography of Additional References

General References

Daniel, D.E., C.D.Shackelford, W.P.Liao, and H.M.Lijestrand (1991). Rate of Flow of Leachate through Clay Soil Liners. EPA/600/S2-91/021, U.S.Environmental Protection Agency, Risk Reduction Engineering Laboratory, Cincinnati, OH.

Eisenberg, N.A., L.D.Rickertsen, and C.Voss (1987). "Performance, Assessment, Site Characterization, and Sensitivity and Uncertainty Methods: Their Necessary Association for Licensing". In Proceedings of the Conference on Geostatistical, Sensitivity, and Uncertainty Methods for Ground-Water Flow and Radionuclide Transport Modeling San Francisco, CA, September 15-17, 1987, Pp. 9-38.

Gillham, R.W, and J.A.Cherry (1982). Contaminant Migration in Saturated Unconsolidated Geologic Deposits. Geologic Society of America Special Paper 189.

References Specific to Other Sites

Nicholas, J.R, and R.W.Healy (1988). Tritium Migration from a Low-Level Radioactive Waste Disposal Site Near Chicago, Illinois. U.S.Geological Survey Water-Supply Paper 2333.

Olimpio, J.C. (1984). Low-Level Radioactive-Waste Burial at the Palos Forest Preserve, Illinois: Geology and Hydrology of the Glacial Drift, as Related to the Migration of Tritium. U.S.Geological Survey Water-Supply Paper 2226.

SL-271
110832 01

00-1425

Formerly Utilized Sites Remedial Action Program (FUSRAP)

ADMINISTRATIVE RECORD

for the St. Louis Site, Missouri

U.S. Department of Energy

4.14-1070.2