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TRANSMITTAL OF INFORMATION ON PARTITION COEFFICIENTS (KDS)

03/02/92

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DOE-FO/EPA
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LETTER

Department of Energy
Fernald Environmental Management Project
P.O. Box 398705
Cincinnati, Ohio 45239-8705
(513) 738-6357

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MAR 0 2 1992

DOE-1009-92

Mr. James A. Saric, Remedial Project Director
U. S. Environmental Protection Agency
Region V - 5HRE-8J
77 W. Jackson Boulevard
Chicago, Illinois 60604

Mr. Graham E. Mitchell, Project Manager
Ohio Environmental Protection Agency
40 South Main Street
Dayton, Ohio 45402-2086

Dear Mr. Saric and Mr. Mitchell:

TRANSMITTAL OF INFORMATION ON PARTITION COEFFICIENTS (K_ds)

In response to the request made by the United States Environmental Protection Agency's Biological Technical Assistance Group regarding information on partition coefficients (K_ds) used in fate and transport modeling, we are enclosing a copy of narrative, tables, and a list of references on this subject from the Risk Assessment Work Plan Addendum, February 1992. As a supplement to this information, also enclosed are copies of selected reference materials pertaining to the subject.

If you or your staff have any questions, please contact me at FTS 774-6159 or (513) 738-6159.

Sincerely,

Jack R. Craig
Fernald Remedial Action
Project Manager

FN:Craig

Enclosure: As Stated

Recycled and Recyclable

cc w/enc.:

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N. E. Hopson, ASI/IT
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J. D. Wood, ASI/IT

2909

RISK ASSESSMENT WORK PLAN ADDENDUM

FERNALD ENVIRONMENTAL MANAGEMENT PROJECT
FERNALD, OHIO

REMEDIAL INVESTIGATION and FEASIBILITY STUDY

February 1992

U.S. DEPARTMENT OF ENERGY
FERNALD OFFICE

DRAFT
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obtained from site RI/FS samples. Additional data may be obtained from porosities listed in published tables found in Morris and Johnson (1967), Driscoll (1986), and van der Leeden et al. (1990).

6.1.4.5 Vertical Seepage Velocity

The estimates of the flow parameters were used to calculate the seepage velocity for input into the vadose zone transport model. To determine whether flow was occurring as a saturated front, infiltration (q) was compared to the vertical hydraulic conductivity (K_v). If $q \geq K_v$, it is assumed that saturated conditions exist and velocity is calculated based upon the following formula:

$$V_{pw} = (K_v)(i)/n \qquad (6\text{-}3)$$

where

V_{pw} = Seepage velocity (m/s)
K_v = Vertical hydraulic conductivity (m/s)
i = Hydraulic gradient (m/m)
n = Porosity (unitless)

If $q < K_v$, it will be assumed that a seepage would not occur under saturated conditions and the following formula would then be used to calculate the seepage velocity:

$$V_{pw} = q/\theta \qquad (6\text{-}4)$$

where

q = Infiltration (m/s)
θ = Moisture content (unitless)

Based on the assumptions of steady-state moisture content, the selected K value, and using the best field data available for hydraulic gradient, the calculated seepage velocity will be conservative and tend to overestimate the rate of fluid movement.

6.1.4.6 Partition Coefficients

As contaminated leachate flows through a geologic formation, the individual contaminants may react with the solids in the formation in a variety of degrees and ways. This slows the transport of these contaminants. Partition coefficients, or "K_d's", are used to account for this phenomenon in the transport equation. A contaminant's K_d expresses the ratio of its concentration in the solid and liquid components in the groundwater flow system, at a given location in that system. The

use of K_d values assumes that a linear equilibrium relationship exists between the solid and solution phase concentrations of a contaminant.

Site-specific K_d values are currently available only for some mobile uranium compounds at the site. A literature search will be completed to determine appropriate K_d values for the remaining inorganic and radioactive constituents. Values found in the literature search will be carefully screened to select those values that will be derived under conditions that approximate those at the FEMP.

When parameter values derived from literature are used, it is imperative that K_d values from similar environments be considered. Similar soil types and water compositions should be used to generate the values. Criteria used to determine similarities in soil types include: pH, E_H, mean arithmetic particle diameter, total organic carbon (TOC), cation exchange capacity (CEC), and free ion oxides (FIO). This may prove difficult in terms of matching groundwater compositions because most studies use dilute acid solutions spiked with the compound of interest and do not represent natural conditions. However, these studies can provide an initial estimate of interaction between the contaminant and the solid matrix. The use of literature K_d values may result in retardation values that differ from site-specific conditions, and would result in uncertainty in the estimate of contaminant concentration at the receptor.

When a site-specific or literature-based K_d value is not available for a given organic chemical, its K_d value can be calculated, using an organic carbon partitioning coefficient, or "K_{oc}", the amount of carbon present in the soil matrix, and the size distribution of the matrix material in the vadose zone (Mills et al. 1985):

$$K_d = K_{oc}\,[0.2(1-f)x^s_{oc} + (f)(x^f_{oc})] \qquad (6\text{-}5)$$

where

K_d = Soil partitioning coefficient (mL/g)
K_{oc} = Organic carbon partitioning coefficient (mL/g)
f = Mass fraction of silt or clay (unitless)
x^s_{oc} = Organic carbon content of sand (unitless)
x^f_{oc} = Organic carbon content of silt-clay (unitless)

The numerical values for (f), (x^s_{oc}), and (x^f_{oc}) will be site-specific. The K_{oc} is the partition coefficient of a contaminant between water and a 100% organic carbon representing the organic material present in soil or sediment. Chemical-specific values for K_{oc} are available in the literature for many organic compounds. Additional K_{oc} values may be calculated using empirical

formulas relating the octanol-water partitioning coefficient (K_{ow}) to the K_{oc}. The K_{ow} (mL/mL) is the ratio of a contaminant's concentrations in a system containing water and octanol. The K_{ow}'s for the remaining constituents of concern are found in the U.S. EPA Water Engineering Research Laboratory Treatability Database, Cincinnati, Ohio supplemented by Howard (1990), Montgomery (1990 and 1991), and Verschueren (1983), if necessary. The formula (Mills et al. 1985) used to relate K_{ow} to K_{oc} is:

$$K_{oc} = (0.63)(K_{ow}) \qquad \textbf{(6-6)}$$

This approach of using empirical relationships assumes K_d is problem-specific because it depends on both the chemical modeled and the soil type, while K_{oc} is a property only of the chemical modeled. [While this approach is generally acceptable, Cleary et al. (1991) present laboratory evidence for five volatile organic compounds in eight different soils which shows K_{oc} is not a fixed property of the chemical in question.] Their study raises questions on the standard use of K_{oc} values. However, the standard approach given by Equation 6-6 appears to be reasonable given the lack of site-specific data.

Estimated K_d values for the major contaminants at the FEMP have been determined and are presented in Tables 6-3 and 6-4. Chemical forms of these radionuclides and metals generally have significant effects on partitioning coefficients and will be evaluated along with site-specific analytical data. Radioactive decay products (progeny) of the radionuclides at the FEMP may not have the same partitioning coefficients as the parent. The impact of such differences on fate and transport modeling results will be evaluated. These estimates of K_d values are acceptable for screening purposes, and conservative transport assessment.

The partitioning coefficient may also used to derive a retardation factor (Rf). Though the K_d/Rf formulation of the reaction term of the transport equation has numerous assumptions and uncertainties associated with it, it nevertheless provides a practical means of incorporating the reaction process into transport models.

6.1.4.7 Effects of Radioactive Decay and Biodegradation

Nuclear, chemical, and biological processes play major roles in the fate of some contaminants, and are thus an important aspect of all fate and transport modeling. For example, concentrations of both radioactive isotopes and organic compounds remaining in the environment for long periods would be greatly overestimated without accounting for the effects of radioactive decay and biodegradation. Therefore, information about radioactive decay and environmental degradation is used in several of the transport models.

TABLE 6-3
PARTITIONING COEFFICIENTS FOR RADIONUCLIDES AND INORGANICS AT THE FEMP[a]

Constituents	Vadose Layer 1 (Clay)		Vadose Layer 2 (Sand & Gravel)	
	K_d (ml/g)	Reference	K_d (ml/g)	Reference
Radionuclides				
Ac-227	2.40E+03	Sheppard & Thibault 1990	4.50E+02	Sheppard & Thibault 1990
Cs-137	1.81E+03	Sheppard et. al. 1984	1.37E+03	Sheppard et. al. 1984
Np-237	5.50E+01	Sheppard & Thibault 1990	5.00E+00	Sheppard & Thibault 1990
Pa-231	2.70E+03	Sheppard & Thibault 1990	5.50E+02	Sheppard & Thibault 1990
Pb-210	3.00E+03	Gerritse et. al. 1982	3.80E+01	Raj and Zachara 1984
Pu-238	1.70E+03	Glover et. al. 1976	1.00E+02	Glover et. al. 1976
Pu-239/240	1.70E+03	Glover et. al. 1976	1.00E+02	Glover et. al. 1976
Ra-224	6.96E+02	Gillham et. al. 1981	1.06E+02	Sheppard et. al. 1984
Ra-226	6.96E+02	Gillham et. al. 1981	1.06E+02	Sheppard et. al. 1984
Ra-228	6.96E+02	Gillham et. al. 1981	1.06E+02	Sheppard et. al. 1984
Ru-106	8.00E+02	Sheppard & Thibault 1990	5.50E+01	Sheppard & Thibault 1990
Sr-90	1.00E+01	Sheppard et. al. 1984	2.50E+00	Sheppard et. al. 1984
Tc-99	1.18E-01	Sheppard et. al. 1984	7.00E-02	Sheppard et. al. 1984
Th-228	5.80E+03	Sheppard & Thibault 1990	3.20E+03	Sheppard & Thibault 1990
Th-230	5.80E+03	Sheppard & Thibault 1990	3.20E+03	Sheppard & Thibault 1990
Th-232	5.80E+03	Sheppard & Thibault 1990	3.20E+03	Sheppard & Thibault 1990
U-234	1.80E+00	DOE 1989	1.48E+00	DOE 1989
U-235/236	1.80E+00	DOE 1989	1.48E+00	DOE 1989
U-238	1.80E+00	DOE 1989	1.48E+00	DOE 1989

TABLE 6-3
(Continued)

Constituents	Vadose Layer 1 (Clay)		Vadose Layer 2 (Sand and Gravel)	
	Kd (ml/g)	Reference	Kd (ml/g)	Reference
Inorganics				
Aluminum	1.50E+03	Baes and Sharp 1984	1.5E+03	Baes and Sharp 1984
Arsenic	2.00E+02	Baes and Sharp 1984	2.00E+02	Baes and Sharp 1984
Antimony	2.50E+02	Sheppard & Thibault 1990	4.50E+01	Sheppard & Thibault 1990
Barium	1.14E+03	Gillham et. al. 1981	2.00E+01	Sheppard et. al. 1984
Beryllium	1.30E+03	Sheppard & Thibault 1990	2.50E+02	Sheppard and Thibault 1990
Cadmium	5.00E+02	Gerritse et. al. 1982	1.20E+01	Raj and Zachara 1984
Calcium	5.00E+01	Sheppard and Thibault 1990	5.00E+00	Sheppard and Thibault 1990
Chromium	1.50E+03	Sheppard and Thibault 1990	7.00E+01	Sheppard and Thibault 1990
Cobalt	5.50E+02	Sheppard and Thibault 1990	6.00E+01	Sheppard and Thibault 1990
Copper	1.25E+02	Gerritse et. al. 1982	3.50E+01	Baes and Sharp 1984
Iron	1.65E+02	Sheppard and Thibault 1990	2.20E+02	Sheppard and Thibault 1990
Lead	3.00E+03	Gerritse et. al. 1982	3.80E+01	Raj and Zachara 1984
Magnesium	4.50E+00	Baes and Sharp 1984	4.50E+00	Baes and Sharp 1984
Manganese	1.80E+02	Sheppard and Thibault 1990	5.0E+01	Sheppard and Thibault 1990
Mercury	1.00E+01	Baes and Sharp 1984	1.00E+01	Baes and Sharp 1984
Molybdenum	9.00E+01	Sheppard and Thibault 1990	1.0E+01	Sheppard and Thibault 1990
Nickel	6.50E+02	Sheppard and Thibault 1990	4.00E+02	Sheppard and Thibault 1990
Potassium	7.50E+01	Sheppard and Thibault 1990	1.50E+01	Sheppard and Thibault 1990
Selenium	7.40E+02	Sheppard and Thibault 1990	1.50E+02	Sheppard and Thibault 1990
Silver	1.80E+02	Sheppard and Thibault 1990	9.00E+01	Sheppard and Thibault 1990
Sodium	1.00E+02	Baes and Sharp 1984	1.00E+02	Baes and Sharp 1984
Thallium	1.50E+03	Baes and Sharp 1984	1.50E+03	Baes and Sharp 1984
Vanadium	1.00E+03	Baes and Sharp 1984	2.00E+02	Gerritse et. al. 1982
Zinc	2.40E+03	Sheppard and Thibault 1990	2.00E+02	Sheppard and Thibault 1990

TABLE 6-4
PARTITIONING COEFFICIENTS FOR ORGANIC COMPOUNDS AT THE FEMP[a]

Constituents	K_{ow}[b] (mL/mL)	K_{oc} (mL/g)	K_d[c] Vadose 1 (mL/g)	K_d[c] Vadose 2 (mL/g)
1,1-Dichloroethane	6.17E+01	3.89E+01	1.18E+00	5.10E-01
1,1-Dichloroethene	3.02E+01	1.90E+01	5.78E-01	2.50E-01
1,1,2-Trichloro-1,2,2-trifluorethane[d]	1.48E+02	9.32E+01	2.83E+00	1.22E+00
1,1,1-Trichloroethane	2.95E+02	1.86E+02	5.65E+00	2.44E+00
1,1,2,2-Tetrachloroethane	2.46E+02	1.55E+02	4.70E+00	2.03E+00
1,2-Dichloroethene[e]	1.23E+02	7.75E+01	2.36E+00	1.02E+00
2-Butanone	1.81E+00	1.14E+00	3.47E-02	1.50E-02
2-Methylnaphthalene	7.24E+03	4.56E+03	1.39E+02	5.98E+01
2-Methyl phenol	8.91E+01	5.61E+01	1.71E+00	7.36E-01
2-Propanol	6.90E-01	4.35E-01	1.32E-02	5.70E-03
2,4-Dimethyl phenol	2.63E+02	1.66E+02	5.04E+00	2.17E+00
4-Methyl-2-Pentanone[e]	1.23E+01	7.75E+00	2.36E-01	1.02E-01
4-Methyl phenol[e]	7.94E+01	5.00E+01	1.52E+00	6.57E-01
Acenaphthene	8.32E+03	5.24E+03	1.59E+02	6.88E+01
Acetone	5.70E-01	3.59E-01	1.09E-02	4.71E-03
Anthracene	2.80E+04	1.76E+04	5.36E+02	2.31E+02
Aroclor-1016	2.40E+04	1.51E+04	4.60E+02	1.98E+02
Aroclor-1242	1.29E+04	8.11E+03	2.47E+02	1.06E+02
Aroclor-1248	5.62E+05	3.54E+05	1.08E+04	4.65E+03
Aroclor-1254	1.07E+06	6.75E+05	2.05E+04	8.86E+03
Aroclor-1260	1.29E+06	8.13E+05	2.47E+04	1.07E+04
Benzene	1.35E+02	8.51E+01	2.59E+00	1.12E+00
Benzo(a)anthracene	4.00E+05	2.52E+05	7.66E+03	3.31E+03
Benzo(a)pyrene	9.55E+05	6.02E+05	1.83E+04	7.89E+03
Benzo(b)fluoranthene	3.72E+06	2.34E+06	7.11E+04	3.07E+04
Benzo(g,h,i)perylene	1.70E+07	1.07E+07	3.25E+05	1.40E+05
Benzo(k)fluoranthene	6.92E+06	4.36E+06	1.32E+05	5.72E+04

TABLE 6-4
(Continued)

Constituents	K_{ow}[b] (mL/mL)	K_{oc} (mL/g)	K_d[c] Vadose 1 (mL/g)	K_d[c] Vadose 2 (mL/g)
Benzoic acid	7.41E+01	4.67E+01	1.42E+00	6.13E-01
Beta-BHC	6.31E+03	3.98E+03	1.21E+02	5.22E+01
Bis(2-ethylhexyl)phthalate	2.00E+05	1.26E+05	3.83E+03	1.65E+03
Butylbenzyl phthalate	6.03E+04	3.80E+04	1.15E+03	4.98E+02
Carbondisulfide	1.45E+02	9.14E+01	2.78E+00	1.20E+00
Carbon tetrachloride	5.37E+02	3.38E+02	1.03E+01	4.44E+00
Chloroform	9.33E+01	5.88E+01	1.79E+00	7.71E-01
Chlordane	6.03E+02	3.80E+02	1.15E+01	4.98E+00
Chrysene	4.00E+05	2.52E+05	7.66E+03	3.31E+03
cis-1,2-Dichloroethene[e]	3.02E+01	1.90E+01	5.78E-01	2.50E-01
Cyanide	2.24E+00	1.41E+00	4.29E-02	1.85E-02
DDT	1.55E+06	9.77E+05	2.97E+04	1.28E+04
Dibenzofuran	1.32E+04	8.32E+03	2.53E+02	1.09E+02
Dibenzo(a,h) anthracene	9.33E+05	5.88E+05	1.79E+04	7.71E+03
Di-n-butyl phthalate	1.58E+05	9.95E+04	3.03E+03	1.31E+03
Di-n-octyl phthalate	1.58E+09	9.95E+08	3.03E+07	1.31E+07
Ethylbenzene	1.40E+03	8.82E+02	2.68E+01	1.16E+01
Ethyl parathion[e]	5.75E+03	3.63E+03	1.10E+02	4.76E+01
Fluoranthene	2.14E+05	1.35E+05	4.09E+03	1.77E+03
Fluorene	1.50E+04	9.45E+03	2.87E+02	1.24E+02
Idenol(1,2,3-cd)pyrene	4.57E+07	2.88E+07	8.75E+05	3.78E+05
Methyl parathion	1.10E+02	6.93E+01	2.11E+00	9.09E-01
Methylene chloride	1.78E+01	1.12E+01	3.41E-01	1.47E-01
Naphthalene	2.30E+03	1.45E+03	4.40E+01	1.90E+01
N-Nitrosodiphenyl amine	1.35E+03	8.51E+02	2.59E+01	1.12E+01
Pentachlorophenol	1.02E+05	6.43E+04	1.95E+03	8.43E+02
Phenol	2.88E+01	1.81E+01	5.52E-01	2.38E-01
Phenanthrene	2.90E+04	1.83E+04	5.55E+02	2.40E+02

TABLE 6-4
(Continued)

Constituents	K_{ow}[b] (mL/mL)	K_{oc} (mL/g)	K_d[c] Vadose 1 (mL/g)	K_d[c] Vadose 2 (mL/g)	
Pyrene	1.51E+05	9.54E+04	2.90E+03	1.25E+03	1
Tetrachloroethene	3.39E+02	2.13E+02	6.49E+00	2.80E+00	2
Toluene	4.90E+02	3.09E+02	9.38E+00	4.05E+00	3
Trichloroethene	3.39E+02	2.13E+02	6.49E+00	2.80E+00	4
Total Xylenes	1.10E+03	6.93E+02	2.11E+01	9.09E+00	5
Vinyl Chloride	3.98E+00	2.51E+00	7.62E-02	3.29E-02	6

a This table presents default values, which are subject to change, based on FEMP-site site-specific data. 7
b K_{ow} taken from EPA Treatability Database (1990). 8
c Calculated by Equation 6-5. 9
d K_{ow} data are not available in the EPA Treatability Database (1990). K_{ow} data from Howard (1990). 10
e K_{ow} data are not available in the EPA Treatability Database (1990). K_{ow} data from Montgomery et al. (1990). 11

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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
(708)972-5569
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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 ith 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 ith 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 ith 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 ith 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 ith principal radionuclide transported through the pqr^{th} pathway, and $W_{ir}(t)$ is the concentration in water (pCi/L) at time t of the ith principal radionuclide transported through the rth 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 rth water pathway segment (g/L) -- $W_{ir}(t)$ is the concentration in water (pCi/L) at time t of the ith principal radionuclide transported through the rth 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 ith 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):

$$\dot{R}_i(t) = L_i \, \rho_b^{(cz)} \, A \, T(t) \, S_i(t) \qquad \text{(E.2)}$$

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 ith 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, $\theta^{(cz)}_{sat}$, and the saturation ratio of the contaminated zone, $R^{(cz)}_{sat}$. The saturated water content is the water content when the soil material is saturated. Hence, θ_{sat}

[redacted]

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).

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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.

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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).

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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 ith 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 ith 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.

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TABLE E.3 Typical Average Distribution Coefficients for Various Elements in Sand, Soils, and Clays

Element	Average Distribution Coefficient,[a] K_d (cm^3/g) Soils and Clays	Sand[b]	Geometric Standard Deviation	Element	Average Distribution Coefficient,[a] K_d (cm^3/g) Soils and Clays	Sand[b]	Geometric Standard Deviation
As	3	0.3	1.8	Hg	100	10	-
Ba	50	5	-	Ni	1,000	100	-
Cd	7	0 7	2.4	Pu	2,000	200	10
Ce	1,000	100	3.7	Ra[d]	70	7	7
Cs[c]	500	80	2.4	Se	3	0.3	1.9
Cr (+6)	4	0.4	9.0	Ag	100	10	3.7
Co	1,000	100	-	Sr[d,e]	30	3	7.4
Cu	20	2	3.0	Th	60,000	6,000	4.5
F	0	0	-	Tritium	0	0	-
Fe	1,000	100	-	U	50	5	3.6
La	1,000	100	-	Va	1,000	100	-
Pb	100	10	5.5	Zn	20	2	6.7
Li	500	50	-	Zr	1,000	100	-
Mn	200	20	15				

[a]Data for As, Cd, Ce, Cr, Co, Cu, Pb, Mn, Pu, Se, Ag, Sr, Th, U, and Zn from Baes and Sharp (1983); the values of K_d are the geometric means of the literature data (see also Gilbert et al. [1983, pp. 3-57 to 3-60]). Data for other elements from Nuclear Safety Associates (1980), except as noted. All values have been rounded to one significant figure.

[b]The values are taken to be 10% of the values of soils and clays (Nucl. Saf. Assoc. 1980), except for cesium.

[c]From Isherwood (1981), using soil and quartz sand data, rounded to one significant digit.

[d]Sources: U.S. Nuclear Regulatory Commission (1980); Gee et al. (1980).

[e]The corresponding values from Isherwood (1981) for Sr are 73 for soils, 2.5 for quartz sand, and 24 for other sands.

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TABLE E.4 Distribution Coefficients for Strontium and Cesium[a]

Conditions	K_d (cm^3/g)	
	Strontium	Cesium
Basalt, 32-80 mesh	16-140	790-9,500
Basalt, 0.5-4 mm, 300 ppm TDS	39-280	220-1,200
Basalt, 0.5-4 mm, sea water	1.1	6.5
Basalt-fractured in-situ measurement	3	
Sand, quartz - pH 7.7	1.7-3.8	22-310
Sands	13-43	100
Carbonate, greater than 4 mm	0.19	14
Dolomite, 4,000 ppm TDS	5-14	
Granite, greater than 4 mm	1.7	34
Granodiorite, 100-200 mesh	4-9	8-9
Granodiorite, 0.5-1 mm	11-23	1,000-1,800
Hanford sediments	50	300
Tuff	45-4,000	800-18,000
Soils	19-280	190-1,000
Shaley siltstone, greater than 4 mm	8	310
Shaley siltstone, greater than 4 mm	1.4	100
Alluvium, 0.5-4 mm	48-2,400	120-3,200
Salt, greater than 4 mm saturated brine	0.19	0.027

[a]All values have been rounded to two significant figures.

Source: Data from Isherwood (1981).

TABLE E.5 Distribution Coefficients for Thorium and Uranium

Conditions	K_d (cm^3/g)
Thorium	
Silt loam, Ca-saturated clay, pH 6.5	160,000
Montmorillonite, Ca-saturated clay, pH 6.5	400,000
Clay soil, 5 mM $Ca(NO_3)_2$, pH 6.5	160,000
Medium sand, pH 8.15	40-130
Very fine sand, pH 8.15	310-470
Silt/clay, pH 8.15	270-10,000
Schist soil, 1 g/L Th, pH 3.2	8
Schist soil, 0.1 g/L Th	603.2
Illite, 1 g/L Th, pH 3.2	120
Illite, 0.1 g/L Th, pH 3.2	1,000
Illite, 0.1 g/L Th, pH > 6	<100,000
Uranium	
Silt loam, U(VI), Ca-saturated, pH 6.5	62,000
Clay soil, U(VI), 5 mM $Ca(NO_3)_2$, pH 6.5	4,400
Clay soil, 1 ppm UO^{+2}, pH 5.5	300
Clay soil, 1 ppm UO^{+2}, pH 10	2,000
Clay soil, 1 ppm UO^{+2}, pH 12	270
Dolomite, 100-325 mesh, brine, pH 6.9	4.5
Limestone, 100-170 mesh, brine, pH 6.9	2.9

Source: Data from Isherwood (1981).

[redacted]

TABLE E.6 Distribution Coefficients for Several Radionuclides

Element	pH	K_d (cm^3/g)	Source	Element	pH	K_d (cm^3/g)	Source
Uranium	2	0	A	Lead	2.2	1,850	B
	8	100			7.7	10,000	
	10	600					
	13	50			4-9	99[c]	C
	2.2	1.3	B	Strontium	2	0.1	E
	7.7	23,000			3	6	
					7	500	
	4-9	45[a]	C		10	170	
Thorium	2	500	A		4-9	27[d]	C
	5	3,000					
	7	50,000		Cesium	3	45	E
	13	50			5	150	
					8	905	
	2.2	1.2	B		10	650	
	7.7	80,000					
					4-9	1,100[e]	C
	4-9	60,000[b]	C				
				Plutonium	2	150	E
Radium	2	0	D		5	250	
	4	12			7	8,500	
	6	60			11	1,000	
	7	100					
					4-9	1,800[f]	C
	2.2	13	B				
	7.7	2,400		Deuterium	All	0	

[a]Geometric mean (GM) of values ranging from 11 to 4,400, with a geometric standard deviation (GSD) of 3.7.

[b]GM of values ranging from 2,000 to 510,000, with a GSD of 4.5.

[c]GM of values ranging from 4.5 to 7,600, with a GSD of 5.5.

[d]GM of values ranging from 0.15 to 3,300, with a GSD of 7.4.

[e]GM of values ranging from 10 to 52,000, with a GSD of 6.7.

[f]GM of values ranging from 11 to 300,000, with a GSD of 10.

Sources: A = Rancon (1973); B = Gee et al. (1980); C = Baes and Sharp (1983); D = U.S. Nuclear Regulatory Commission (1980); E = Staley et al. (1979).

[redacted] 32

and the point-of-use concentration $W_i(t)$ in terms of a nuclide water-transport function $G_i(t,t')$ that is valid for any hydrogeological system.

Let $\delta W_{i,k}(t)$ be the concentration at time t at the point of use due to release of a pulse of unit concentration for a time interval $\delta t'$ at time $t'_k = k\delta t'$ [i.e., for a source pulse where k is an integer, $\delta W^s_{i,k}(t') = 1$ when $k\delta t' \le t' < (k+1)\delta t'$ and $\delta W^s_{i,k}(t') = 0$ otherwise]. The point-of-use concentration may then be expressed as the sum of the contributions from the unit pulses multiplied by the source concentration for each pulse:

$$W_i(t) = \sum_k W^s_i(t'_k) \times \delta W_{i,k}(t) \tag{E.9}$$

If the duration $\delta t'$ of each unit source pulse is sufficiently small ($\delta t' \to 0$), the magnitude of each point-of-use pulse will be proportional to the source-pulse duration and one can write

$$\delta W_{i,k}(t) = G_i(t,t'_k)\ \delta t' \tag{E.10}$$

where $G_i(t,t'_k)$ is, by definition, the nuclide water-transport function. In the limit as $\delta t' \to 0$, Equation E.9 becomes

$$W_i(t) = \int_0^t G_i(t,t')\ W^s_i(t')\ dt' \tag{E.11}$$

Using Equation E.11, one obtains the following general formula for the water/soil concentration ratio:

$$WSR_i(t) = \frac{\rho_b^{(cz)}}{\theta^{(cz)} S_i(t)} \int_0^t G_i(t,t')\ Q_i(t')\ S_i(t')\ dt' \tag{E.12}$$

where*

$WSR_i(t) = W_i(t)/S_i(t)$ = water/soil concentration ratio for the ith principal radionuclide at time t, the time of use (kg/m^3),

$W_i(t)$ = average concentration of the ith principal radionuclide in water at the end of the water pathway segment at the point and time of use (Ci/m^3),

$S_i(t)$ = average concentration of the ith radionuclide in the contaminated zone at the time of use, including contributions from ingrowth from other principal radionuclides and the decrease due to removal by radioactive decay and leaching (Ci/kg),

$\rho_b^{(cz)}$ = bulk density of the contaminated zone (kg/m^3),

$\theta^{(cz)}$ = volumetric water content of the contaminated zone (dimensionless),

$G_i(t,t')$ = nuclide water-transport function for the ith principal radionuclide (1/yr),

t = time at which water is used for irrigation or drinking (yr), and

t' = time at which water leaves the contaminated zone (yr).

The volumetric water content $\theta^{(cz)}$ is discussed in Section E.1. The source term $Q_i(t')$ in Equation E.12 is the leaching ratio, defined as the

*Note that some quantities are expressed in different units in different formulas, e.g., $WSR_i(t)$ is expressed in g/L in Equation E.1 and kg/m^3 in Equation E.5. Units of meters, kilograms, and curies are used for intermediate calculations. Units of centimeters, grams, and picocuries are generally used for input and output data or quantities that are more easily interpreted and compared when expressed in terms of these units. Conversion factors for activity concentrations and densities are 1 Ci/kg = 10^9 pC[redacted] 1 Ci/m^3 = 1×10^6 pCi/cm^3, and 1 kg/m^3 = 1 g/L = 1×10^{-3} g/cm^3.

water/soil concentration ratio for the ith principal radionuclide in the contaminated effluent water from the contaminated zone:

$$Q_i(t') = \theta^{(cz)} W_i^s(t')/[\rho_b^{(cz)} S_i(t')] \qquad (E.13)$$

where

$W_i^s(t')$ = average volume concentration of the ith 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 ith 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,

$$W_i^s(t') = \dot{R}_i(t')/IA. \qquad (E.14)$$

Substituting Equations E.2 and E.14 into Equation E.13 yields

$$Q_i(t') = \frac{1}{R_{d_i}^{(cz)}} . \qquad (E.15)$$

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)

$= \left[\sum_j S_j(0)\, ID_{ji}(t)\right] / \left[\sum_j S_j(0)\, ID_{ji}(t')\right]$, and

$g_i(t - t')$ = ingrowth- and decay-independent nuclide water-transport function for the ith 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 ith 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

[redacted]

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 ith 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

[redacted]

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

$$
\begin{aligned}
g_i(t) &= 0, && t \le \Delta t_i \\
&= g_i, && \Delta t_i < t < \Delta t_i + \delta t_i \\
&= 0, && \Delta t_i + \delta t_i \le t
\end{aligned}
\qquad \text{(E.21)}
$$

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 ith 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 ith 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

$$WSR_i(t) = 0, \qquad t \le \Delta t_i$$

$$= \frac{\rho_b f_i}{\theta^{(cz)} R_{d_i}^{(cz)} \delta t_i L_i} \left[1 - \exp\left[- \left(L_i(t - \Delta t_i)\right)\right]\right], \qquad \Delta t_i < t \le \Delta t_i + \delta t_i$$

$$= \frac{\rho_b f_i}{\theta^{(cz)} R_{d_i}^{(cz)} \delta t_i L_i} \left[\exp\left[- L_i(t - \Delta t_i - \delta t_i)\right] - \exp\left[- \left(L_i(t - \Delta t_i)\right)\right]\right], \qquad \Delta t_i + \delta t_i < t \qquad \text{(E.22)}$$

where the various parameters are as defined before.

If the rise time is negligibly small ($\delta t_i = 0$), Equation E.22 reduces to

$$WSR_i(t) = 0, \qquad t \le \Delta t_i$$

$$= \frac{\rho_b^{(cz)} f_i}{\theta^{(cz)} R_{d_i}^{(cz)}} \exp\left[- \left(L_i(t - \Delta t_i)\right)\right], \qquad \Delta t_i < t \qquad \text{(E.23)}$$

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

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ORNL-5786

Health and Safety Research Division

A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture

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seen in the systematic variability of our B_v estimates (Figs. 2.27 and 2.28) and F_m estimates (Figs 2.29 and 2.30). However, experimental determination of F_f for elements of atomic number greater than 82 would be preferable to our present approach, if available.

2.4 The Distribution Coefficient, K_d

The distribution coefficient, K_d, is the ratio of elemental concentration in soil to that in water in a soil-water system at equilibrium. In general, K_d is measured in terms of gram weights of soil and milliliter volumes of water. In TERRA the distribution coefficient is used in the following equation to determine a location-specific leaching constant for elemental removal from a given soil depth,

$$\lambda_l = \frac{P+I-E}{\theta d[1+(\frac{\rho}{\theta}K_d)]}, \tag{7}$$

where

P = annual average total precipitation (cm),
E = annual average evapotranspiration (cm),
I = annual average irrigation (cm),
d = depth of soil layer from which leaching occurs (cm),
ρ = soil bulk density (g/cm^3),
θ = volumetric water content of the soil [mL(= cm^3)/cm^3), and
K_d = the distribution coefficient (mL/g).

Default estimates of K_d used in the TERRA code are presented in Fig. 2.31. The mantissa of these values has been rounded off to the nearest 0.5 decimal place as for the other element specific transport parameters. The values for magnesium, potassium, calcium, manganese, iron, cobalt, copper, zinc, strontium, yttrium, molybdenum, technetium, ruthenium, cesium, lead, polonium, cerium, thorium, uranium, neptunium, plutonium, americium, and curium were determined through a review of the K_d literature. The estimates for the remaining elements were determined by a correlation of K_d with B_v. Because of the inherent uncertainities in estimates of K_d for various materials, a brief discussion of the parameter and its determination is appropriate.

2.4.1 Variability in K_d

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 batch methods, although other methods have been employed to measure distributions of chemical forms[147] or distribution among soil fractions.[148] 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

$$\frac{V_l}{V_w} = \frac{1}{1+\frac{\rho}{\theta}K_d}, \tag{8}$$

where

V_l = the velocity of the migrating material (determined from the 50% breakthrough curve) and
V_w = the velocity of the water.

In the batch method, soil and water are shaken with the material for a period of time until equilibirum 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.[149] 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, a 1% overestimate of the soil concentration (95.95 μg in soil) yields a K_d of 237 mL/g, or approximately a 25% overestimate of K_d. The relative error in K_d estimate from a given percent error in measurement of soil concentration increases rapidly with increasing K_d (Fig. 2.32). 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 to K_d estimates for materials which are highly soluble or insoluble.

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[150] of lead absorption by 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 was found. At pH > 7.0, lead 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 plutonium and curium;[151–153] ruthenium, yttrium, zirconium, niobium, and cerium;[154] arsenic and selenium;[155,156] and manganese, iron, zinc, cobalt, copper, cadmium, and calcium.[157–159]

Another source of variation in K_d is the time factor involved with its determination. Batch-type K_d determinations are usually made over a period of a few to several hours until equilibrium is achieved or assumed. If equilibrium does not occur within this short time period, some error is introduced. Errors from nonequilibrium K_d determinations made after 24 hours, however, are relatively insignificant.[151,152,160] 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.[23] found that sorption of Tc-99 by low organic soils tended to significantly increase over a 5-6 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, and CdS have been reported by Cole,[161] and microbial influences on the solubility of transuranics has also been suggested by Wildung and Garland.[162] 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 2.13). Because of the variation of K_d with soil pH, an analysis of 222 agricultural soils[163,164] was 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 which were measured in soils outside of the pH range of 4.5 to 9. The K_d determinations used to generate Table 2.13 represent a diversity of 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

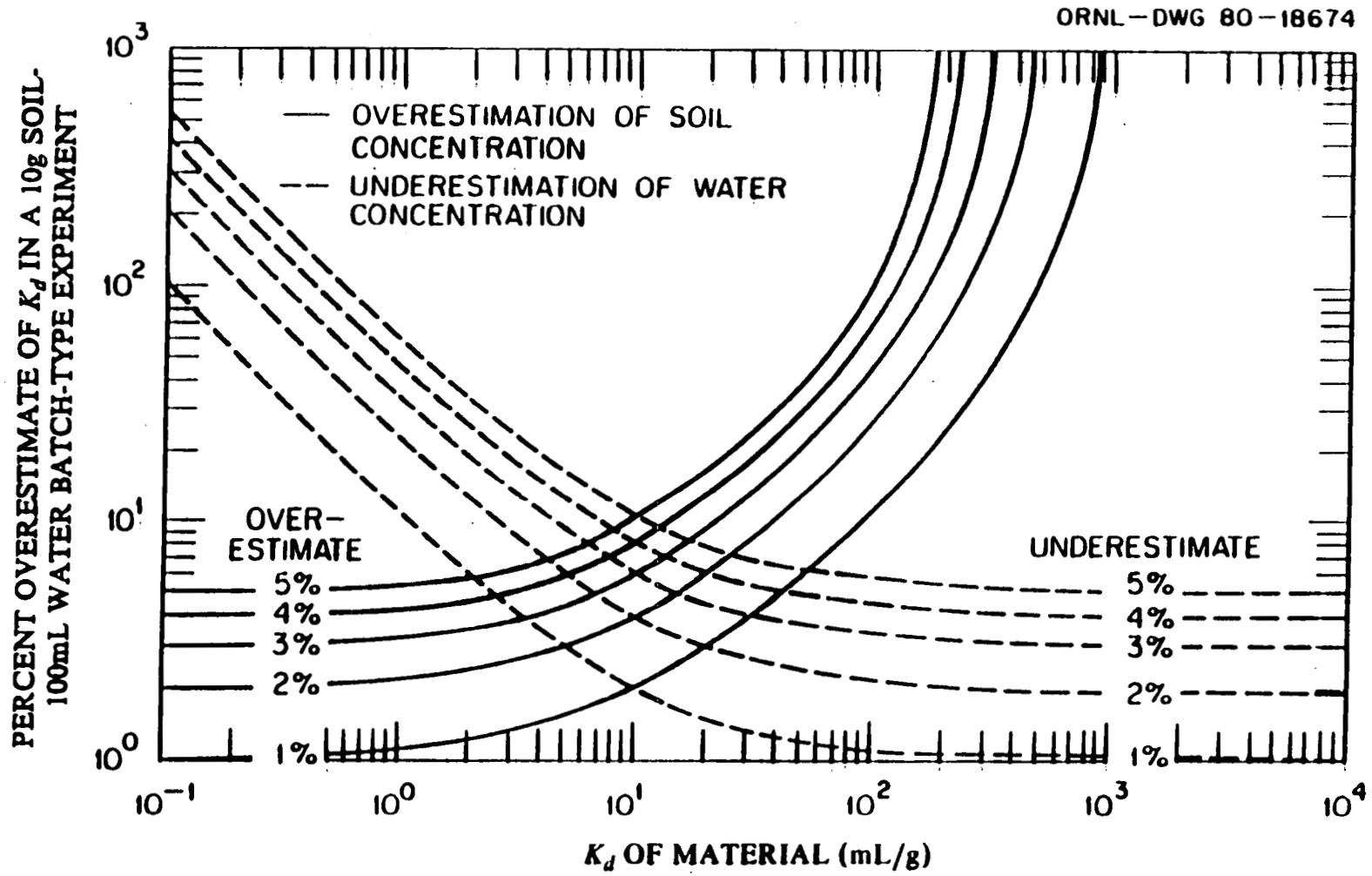

Figure 2.32. Percent error in K_d estimation from one to five percent overestimates of soil concentration or underestimates of water concentration in a 10g-100mL batch-type K_d experiment.

[redacted]

Table 2.13. Estimates of the distribution of K_d for various elements in agricultural soils of pH 4.5 to 9.0

Element	# Obs.	μ[a]	σ[b]	exp(μ)[c]	Observed range	References
				mL/g		
Mg	58	1.5	0.40	4.6	1.6 to 13.5	165, 166
K	10	1.7	0.49	5.6	2.0 to 9.0	165
Ca	10	1.4	0.78	4.1	1.2 to 9.8	165
Mn	45	4.2	2.5	65	0.2 to 10,000	149, 158, 167, 168
Fe	30	3.2	2.0	25	1.4 to 1,000	149, 158, 167, 169
Co	57	3.9	1.1	47	0.2 to 3,800	149, 158, 160, 167, 169–171
Cu	55	3.6	0.97	35	1.4 to 333	157, 158
Zn	146	3.6	1.8	38	0.1 to 8,000	149, 157–159, 167
Sr	218	3.6	1.6	37	0.15 to 3,300	149, 152, 154, 160, 167, 169, 171–180
Y	2	6.2	1.7	510	160 to 1,640	154
Mo	17	2.9	2.2	18	0.37 to 400	149
Tc	24	−3.4	1.1	0.033	0.0029 to 0.28	23
Ru	17	5.9	0.75	350	48 to 1,000	154, 160
Ag	16	3.8	1.5	46	10 to 1,000	149, 167
Cd	28	1.9	0.86	6.4	1.26 to 26.8	157
Cs	135	6.9	1.8	1000	10 to 52,000	149, 160, 167, 169, 171, 173, 175, 177, 178, 180–183
Ce	16	6.7	0.54	840	58 to 6,000	154, 160
Pb	125	6.0	2.1	400	4.5 to 7640	150, 184
Po	6	6.3	0.65	520	196 to 1,063	184
Th	17	12	0.57	150,000	2,000 to 510,000	185–187
U	24	6.1	2.5	450	10.5 to 4,400	185–187
Np	44	3.4	2.5	29	0.16 to 929	148, 186, 188, 189
Pu	40	8.4	2.4	4500	11 to 300,000	151, 152–154, 177, 182, 186, 187, 189
Am	46	6.5	2.4	680	1.0 to 47,230	148, 188–190
Cm	31	7.6	1.6	1,900	99.3 to 51,900	148, 153, 189

[a]The mean of the logarithms of the observed values.
[b]The standard deviation of the logarithms of the observed values.
[c]Geometric mean (50% cumulative probability).

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 cesium and strontium (Fig. 2.33), a lognormal distribution for K_d has been assumed for all elements. Thus, the median value of the assumed lognormal distribution is used as a best estimate default K_d for TERRA (except for lead, and technetium where judgement was exercised). However, if the distribution of K_d computed for cesium and strontium are typical, then K_d may vary by as much as three orders of magnitude in soils of pH 4.5 to 9.0. Such variation in K_d is greater than or equal to the variation in B_v observed for cesium, strontium, and plutonium (Figs. 2.3, 2.7, and 2.22) and suggests the advisability of using site-specific values when available.

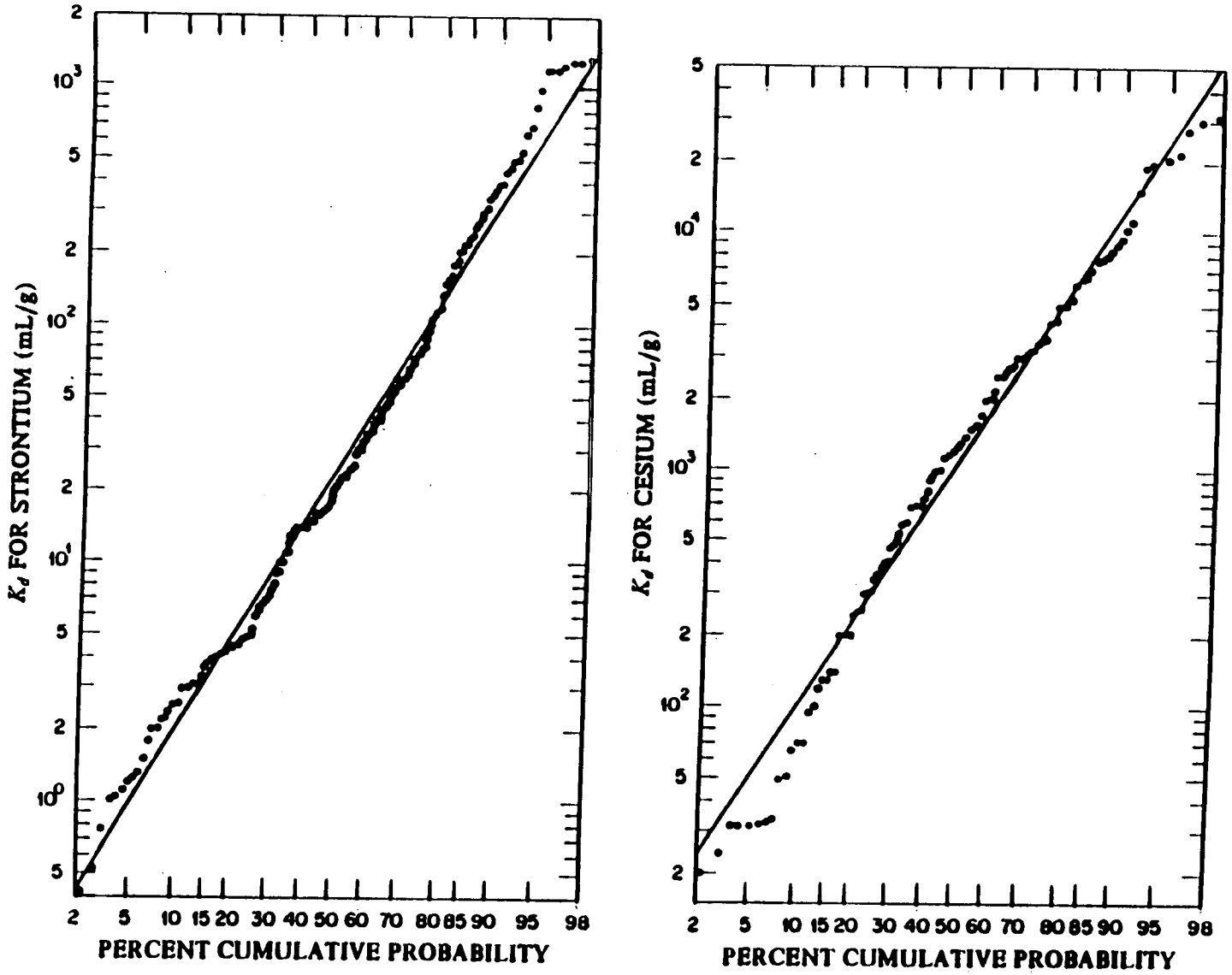

Figure 2.33. Lognormal probability plots of K_d for cesium and strontium in soils of pH 4.5 to 9 based on available references.

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2.4.2 Estimates of K_d based on default B_v values

Although K_d estimates for the 23 above-mentioned elements are subject to great uncertainty, they are based on values reported in the literature. No references are immediately available for the remaining elements of the periodic table. In order to provide a default estimate for these elements, an alternative method is used. In 1979, Van Dorp, Eleveld, and Frissel[191] proposed a model for estimation of the soil-plant concentration factor. Their approach was to calculate the solubility of a nuclide in soil water, its ability to transfer across root membranes, and its upward movement with the transpiration stream. They reasoned that measured values of K_d, root selectivity coefficient (S), and transpiration coefficient (T_c) would allow them to predict the soil-plant concentration factor from soil-radionuclide concentration. Their model has not become generally used or accepted for dose calculations, but their implied dependency of B_v on K_d is the basis of our approach for estimating default K_d estimates in lieu of experimental determinations.

Our approach is to presume that the default K_d estimates for elements in Sect. 2.4.1 and their corresponding B_v estimates represent a wide variety of soils and plants. Therefore, a single default estimate for B_v and K_d will reflect soils, plants, and experimental conditions which are "averaged" or "generalized." Thus, any relationship observed between K_d and B_v may be used to predict "average" or "generalized" K_d estimates from our default B_v estimates.

Figure 2.34 shows the correlation found between B_v and K_d. It should be noted that the B_v estimates in Fig. 2.34 are the geometric means determined directly through analysis of reviewed literature, and not necessarily the default values from Fig. 2.1. Technetium is an example. The technetium B_v of 89 is the geometric mean of the geometric means of references 23, 107, 122, and 123. It was felt that although the short-term plant uptake studies represented in references 23, 107, and 122 were inappropriate for long-term B_v estimates, they were appropriately associated with the short-term K_d determinations for technetium (because B_v decreases and K_d increases with time). Thus, these two short-term parameters were used in the definition of the B_v-K_d relationship. However, in Fig. 2.31 we used our best estimate of technetium B_v and the regression equation

$$K_d = \exp(2.38 - 0.89(\ln B_v)) \tag{9}$$

to determine our best estimate of technetium K_d of 1.5. In addition to technetium the K_d default estimates for elements not mentioned in Sect. 2.4.1 were determined via Eq. (9) and the best estimate B_v default values in Fig. 2.1.

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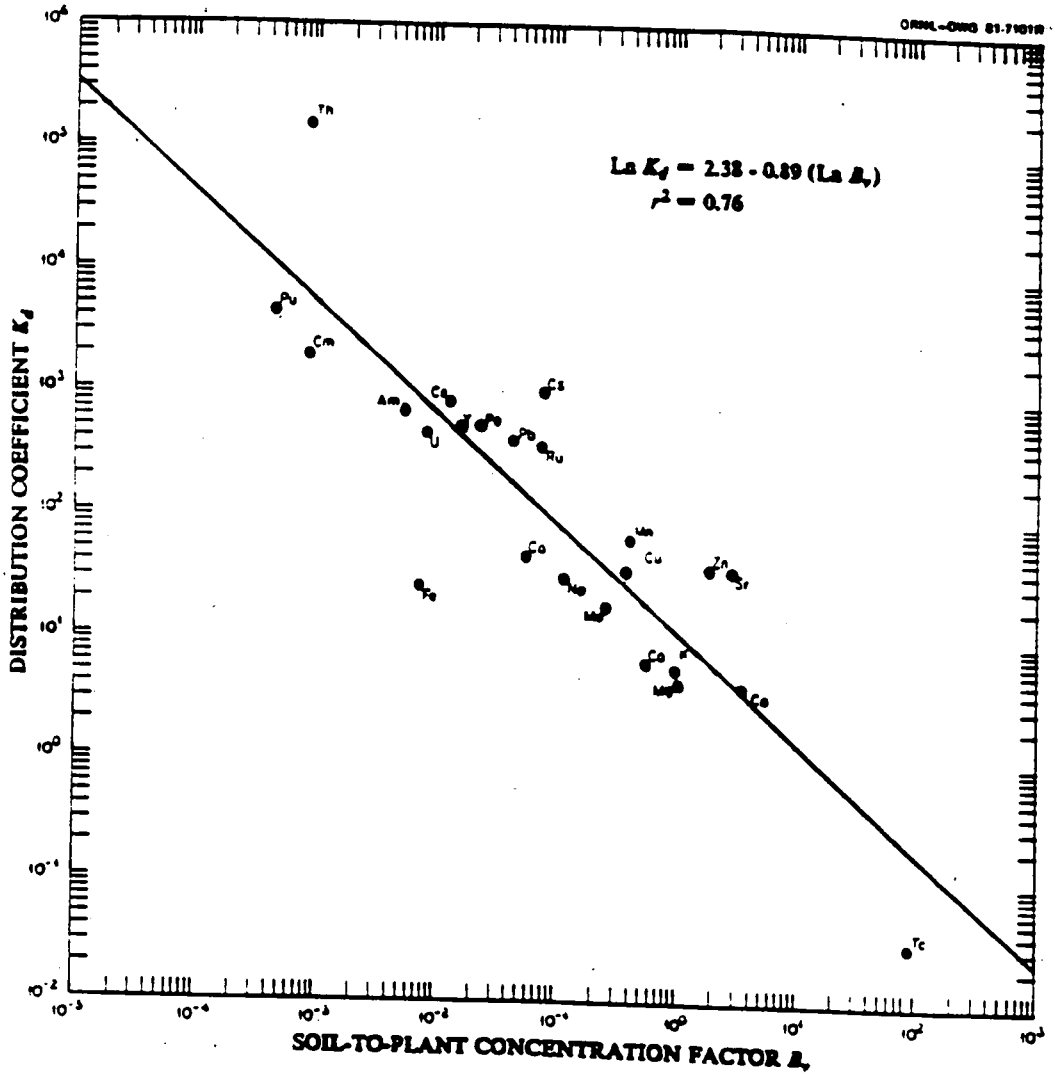

Figure 2.34. Correlation between B_v and K_d based on geometric means of available reference geometric means.

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SOIL NUCLIDE DISTRIBUTION COEFFICIENTS AND THEIR STATISTICAL DISTRIBUTIONS

COEFFICIENTS DE REPARTITION DES NUCLIDES DANS LE SOL ET LEUR REPARTITION STATISTIQUE

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Etablissement de recherches nucléaires de Whiteshell

Pinawa, Manitoba R0E 1L0

December 1984 décembre

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Whiteshell Nuclear Research Establishment
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1984 December

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par

Marsha I. Sheppard, Donald I. Beals, Denis H. Thibault
et Patrick O'Connor

RÉSUMÉ

Les évaluations écologiques de l'évacuation des déchets de combustible nucléaire dans les formations de roche plutonique nécessitent une analyse de la migration des nuclides qui passent de l'enceinte d'évacuation à la biosphère. Pour l'analyse de la migration des nuclides à travers l'enceinte d'évacuation, les matériaux tampons et de remblayage, la roche plutonique et les morts-terrains consolidés et non consolidés, par l'entremise de l'eau souterraine, on se sert de modèles nécessitant des coefficients de répartition (K_d) pour décrire l'interaction des nuclides et des matériaux géologiques et artificiels. Ce rapport présente des coefficients de répartition dans le sol particuliers à certains éléments et leur répartition du point de vue statistique, à partir d'une étude bibliographique en détail. Les éléments radioactifs considérés furent les suivants: actinium, américium, bismuth, calcium, carbone, cérium, césium, iode, plomb, molybdène, neptunium, nickel, niobium, palladium, plutonium, polonium, protactinium, radium, samarium, sélénium, argent, strontium, technétium, terbium, thorium, étain, uranium et zirconium. Les éléments stables considérés furent les suivants: antimoine, bore, cadmium, tellure et zinc. Lorsque la disponibilité des données le permet, les coefficients de répartition et leur répartition sont indiqués pour les sols sabloneux, limoneux, argileux et organiques. L'utilisation de nos valeurs est recommandée pour les évaluations effectuées dans le cadre du programme canadien de gestion des déchets de combustible nucléaire.

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ABSTRACT

Environmental assessments of the disposal of nuclear fuel waste in plutonic rock formations require analysis of the migration of nuclides from the disposal vault to the biosphere. Analyses of nuclide migration via groundwater through the disposal vault, the buffer and backfill, the plutonic rock, and the consolidated and unconsolidated overburden use models requiring distribution coefficients (K_d) to describe the interaction of the nuclides with the geological and man-made materials. This report presents element-specific soil distribution coefficients and their statistical distributions, based on a detailed survey of the literature. Radioactive elements considered were actinium, americium, bismuth, calcium, carbon, cerium, cesium, iodine, lead, molybdenum, neptunium, nickel, niobium, palladium, plutonium, polonium, protactinium, radium, samarium, selenium, silver, strontium, technetium, terbium, thorium, tin, uranium and zirconium. Stable elements considered were antimony, boron, cadmium, tellurium and zinc. Where sufficient data were available, distribution coefficients and their distributions are given for sand, silt, clay and organic soils. Our values are recommended for use in assessments for the Canadian Nuclear Fuel Waste Management Program.

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1. INTRODUCTION

Canada has selected geological containment in a vault deep in plutonic rock in the Precambrian Shield as the most promising method for disposal of its nuclear fuel waste (Boulton, 1978). A stable granitic pluton will most likely be the host rock.

Assessment of the integrity of geological containment requires pathways analysis to determine the travel time from the vault to the biosphere of all the nuclides associated with the waste (Mehta, 1982). The travel time and the predicted nuclide concentrations in the biosphere will depend upon the interaction of the nuclides with their surroundings as they migrate from the vault. Traditionally, this interaction has been described using a distribution coefficient, K_d, for rock, unconsolidated regolith and soil (Wuschke et al., 1981). The objective of this report is to document these K_d values, separating them according to the major soil types found on the Precambrian Shield. These parameter values are required for the soil model in the assessment code used in the Canadian Nuclear Fuel Waste Management Program.

Further, since the assessment code is stochastic, the distributions of the K_d values are also needed. Preliminary work with the K_d values indicates that they are lognormally, as opposed to normally, distributed. The lognormal distribution parameters ($\log_{10}$) are reported here. These parameters directly represent the data presented where two or more values were found, and have not been adjusted toward conservatism for assessment purposes. Sections 4, 5 and 6 list the soil K_d values, and their appropriate distributions, for the actinides, the radionuclides produced from nuclear fission and the stable nuclides, respectively, that are expected to be present in 100-year cooled nuclear fuel (Mehta, 1982). A reference list is included for each nuclide.

2. DISTRIBUTION COEFFICIENT, K_d

The processes of solute migration pertinent to radionuclide migration in soil and unconsolidated geological materials have been discussed and reviewed extensively (Wheeler, 1976; Onishi et al., 1981; Miller, 1983; Gillham and Cherry, 1979; and Arnold et al., 1982). Many computer models have been developed to predict nuclide migration through soil (Murali and Aylmore, 1981; Yeh and Ward, 1981; Oster, 1982; Miller, 1983; Wong et al., 1983; van Genuchten, 1978; Duguid and Reeves, 1976; and Sheppard, 1981). These models vary in their complexity and purpose. The simplest model of the solute transport process, expressed in one-dimensional form, is

$$\frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial x^2} - \bar{V} \frac{\partial C}{\partial x} \qquad (1)$$

where C is the solute concentration in solution, i.e., mass of solute per unit volume of soil ($g.cm^{-3}$),

t is time (s),

D is the dispersion coefficient ($cm^2.s^{-1}$),

x is the space coordinate (cm), and

$\bar{V}$ is the average linear pore-water velocity ($cm.s^{-1}$).

Since Equation (1) does not account for the interaction of the solute and the solid phase, the distribution coefficent, K_d, has been introduced to describe this interaction. The distribution coefficient is defined as the concentration of solute in the adsorbed phase (mass of solute per unit mass of soil) divided by the concentration of solute in the solution phase (mass of solute per unit volume of soil pore water). The units of K_d are usually mL/g. The K_d value for each nuclide represents the partitioning of the solute between the solid and solution phases and is applicable to equilibrium reactions, such as ion exchange.

Typical radionuclide interactions with soil include other geochemical processes, such as precipitation, coprecipitation, hydrous metal oxide

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complexation, organic matter complexation, colloid formation, and microbial effects. Empirically determined K_d values may or may not include these processes.

Ion exchange is one of the most common mechanisms of radionuclide adsorption on geological materials (Ames and Rai, 1978). Thus, the K_d value depends upon several factors, including the cation exchange capacity (CEC), and the species and concentration of both the ion being exchanged and the competing ions. If the nuclide is present in smaller concentrations than the competing ions, then the K_d value will be independent of the concentration of the nuclide, and it will be constant if all other factors remain constant (Johnston and Gillham, 1980).

To incorporate the K_d concept into the solute transport process described by Equation (1), the dispersion coefficient (D) and the pore-water velocity coefficient ($\bar{V}$) become the effective dispersion coefficient (D') and velocity coefficient ($\bar{V}'$), respectively, where

$$D' = \frac{D}{R}$$
$$\bar{V}' = \frac{\bar{V}}{R} \qquad (2)$$

and R is the retardation factor, defined as

$$R = 1 + \frac{\rho_b}{n} K_d \qquad (3)$$

where ρ_b is the bulk density of the soil (g/cm^3)
n is the porosity (cm^3/cm^3), and
K_d is the distribution coefficient (mL/g).

The K_d concept is restricted to equilibrium reactions in which the concentrations in the solution and solid phases are related. K_d was initially defined by Mayer and Tompkins (1947) as

$$K_d = \left(\frac{C_o}{C} - 1\right) \frac{V}{M} \qquad (4)$$

where C_o is the tracer concentration in the solution before adding the sorbent,

C is the tracer concentration in the liquid phase of a sorbent-water suspension,

V is the volume of liquid, and

M is the mass of solid.

Despite the fact that the K_d concept strictly applies only to simple cation-exchange, K_d values are reported that describe more complex reactions. This is in response to the need for input to simple migration models. This report does not review the soil chemistry of the nuclides considered; this has been done adequately elsewhere (Jenson, 1980; Johnston and Gillham, 1980; Allard et al., 1977; Friedman, 1976, Swedish Nuclear Fuel Supply Co. Ltd., 1983; Ames and Rai, 1978). The report does list all of the K_d values by predominant soil type for the Precambrian Shield (sand, silt, clay and organic (Beals, 1984)) and includes other pertinent information found in the literature. Table 1 lists K_d distribution parameter estimates for some nuclides combining all soils, as reported by Baes and Sharp (1981).

TABLE 1

ESTIMATES OF DISTRIBUTIONS OF K_d* VALUES

Nuclide	μ**	σ***	exp(μ)+ (mL/g)	K_d Range (mL/g)
Am	2.9	1.3	810	1.0 to 47 000
Ce	3.0	0.6	1100	58 to 6000
Cs	3.0	0.8	1100	10 to 52 000
Np	1.0	1.0	11	0.16 to 929
Pb	2.0	0.7	99	4.5 to 7600
Po	2.7	0.3	540	200 to 1100
Pu	3.3	1.0	1800	11 to 300 000
Sr	1.4	0.9	27	0.37 to 400
Tc	-1.5	0.5	0.03	0.003 to 0.28
Th	4.8	0.6	60 000	2000 to 510 000
U	1.6	0.6	45	11 to 4400

* From Baes and Sharp (1981)

** Mean of the logarithm (to base 10) of K_d

*** Standard deviation of the logarithm (to base 10) of K_d

\+ Median value of K_d with a 0.5 cumulative probability

[redacted]

3. DISTRIBUTION COEFFICIENTS FOR THE ACTINIDES

3.1 ACTINIUM

Nothing was found in the literature on K_d values or the soil chemistry of actinium. We recommend using the K_d values for americium because of their chemical similarity.

3.2 AMERICIUM

Americium has been studied extensively because of weapons testing in the 1950s. The summary on americium geochemistry presented by Johnston and Gillham (1980) indicates that

(1) the most stable form of americium in aqueous solutions is Am^{3+};

(2) the soil sorption of americium is correlated to cation exchange capacity, clay content, and concentration and type of the competing ions in solution, indicating that the principal retardation mechanism is ion exchange;

(3) at high K_d values, americium adsorption is sensitive to the concentration of americium in solution.

Table 2 lists the K_d values reported by various investigators and includes soil information (texture, pH, competing ions, etc.) pertinent to the sorption data. The recommended K_d value means, standard deviations, ranges and distribution parameters by soil type for americium, based on Table 2, are given in Table 3.

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TABLE 2

K_d VALUES FOR AMERICIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH* Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	fine sandy loam			2.4	-	5.3 (4.39)	-	15	1.65	-	9.635×10^3	Malbis (Louisiana)	Mishita et al. 1979
	fine sandy loam			2.4	-	5.3 (5.71)	-	15	1.65	-	8.063×10^3	Malbis (Louisiana)	Mishita et al. 1979
	fine sandy loam			5.7	-	5.0 (4.58)	-	15	1.52	-	1.549×10^3	Lyman (Maine)	Mishita et al. 1979
	fine sandy loam			5.7	-	5.0 (6.17)	-	15	1.52	-	1.82×10^2	Lyman (Maine)	Mishita et al. 1979
	light loam			8.4	-	6.0 (5.31)	-	15	5.29	-	2.187×10^4	Alkan (California)	Mishita et al. 1979
	light loam			8.4	-	6.0 (6.72)	-	15	5.29	-	1.066×10^4	Alkan (California)	Mishita et al. 1979
	coarse sand			-	-	7-8	-	-	-	Na (90% sat. solution)	4×10^2	(Netherlands)	Beusten & Verhark, 1977
	76.0	21.2	2.8	0.41	-	8.1	-	5.94	-	-	7.14×10^2	(Richland, Washington)	Amos & Mal, 1978
	91.2	7.8	1.0	1.19	-	4.0	-	2.01	-	-	4.76×10^2	Puquay (Barnwell, SC) 0-5 cm	Amos & Mal, 1978
	91.6	5.4	3.0	0.99	-	6.7	-	1.79	-	-	4.17×10^2	Puquay (Barnwell, SC) 5-15 cm	Amos & Mal, 1978
	94.6	1.6	3.8	0.21	-	5.2	-	0.69	-	-	2.49×10^2	Puquay (Barnwell, SC) 15-30 cm	Amos & Mal, 1978
	65.2	29.0	5.8	0.45	-	8.1	-	6.14	-	-	1.23×10^2	Hanford A	Amos & Mal, 1978
	83.6	12.6	3.8	0.17	-	8.4	-	4.95	-	-	8.33×10^2	Hanford B	Amos & Mal, 1978
	42.6	39.4	18.0	0.60	-	8.6	-	15.04	-	-	3.92×10^3	Idaho A	Amos & Mal, 1978
	60.4	19.4	20.2	0.18	-	8.4	-	10.44	-	-	4.35×10^4	Idaho B	Amos & Mal, 1978
	83.4	8.8	7.8	0.16	-	8.4	-	6.38	-	-	3.7×10^4	Idaho C	Amos & Mal, 1978
	49.2	28.4	22.4	0.98	-	7.7	-	18.36	-	-	1.09×10^4	Idaho D	Amos & Mal, 1978
	44.0	20.0	36.0	2.4	0.4	5.7	0.41	20.9	-	-	$2.5\times10^3 \pm 210^+$	Colorado A (Rocky Flats)	Glover et al., 1976
	64.0	14.0	22.0	3.4	0.3	5.6	0.52	17.5	-	-	$6.0\times10^2 \pm 24^+$	Colorado B (Sugar Loaf)	Glover et al., 1976
	44.0	24.0	32.0	0.2	7.9	8.3	0.43	13.8	-	-	$3.0\times10^2 \pm 10^+$	Idaho B	Glover et al., 1976
	66.0	11.0†	23.0	0.3	5.2	8.0	0.47	8.2	-	-	$8.2\times10^2 \pm 43^+$	Idaho C	Glover et al., 1976
	38.0	32.0	30.0	0.1	0.0	7.5	0.45	17.5	-	-	$1.0\times10^4 \pm 1.5\times10^{4+}$	Idaho D	Glover et al., 1976
	74.0	12.0	14.0	0.3	0.6	8.0	0.43	6.4	-	-	$1.2\times10^2 \pm 7^+$	Washington A (Hanford)	Glover et al., 1976
	74.0	12.0	14.0	0.1	0.0	8.2	0.44	5.8	-	-	$2.3\times10^2 \pm 5^+$	Washington B (Hanford)	Glover et al., 1976
	78.0	2.0	20.0	0.7	0.2	5.4	0.54	2.9	-	-	$8.2\times10^1 \pm 1^{+**}$	S. Carolina (Barnwell)	Glover et al., 1976
	48.0	34.0†	18.0	0.7	0.2	6.4	0.49	7.0	-	-	$4.0\times10^2 \pm 11^+$	New Mexico (Los Alamos)	Glover et al., 1976
	82.0	9.0	9.0	0.6	0.7	4.8	0.57	3.8	-	-	$3.9\times10^2 \pm 20^+$	Arkansas B	Glover et al., 1976
Silt	silty clay loam			2.8		5.9 (5.41)	-	20	1.29		2.98×10^4	Sharpsburg (Iowa)	Mishita et al., 1979
	silty clay loam			2.8		5.9 (6.56)	-	20	1.29		1.728×10^4	Sharpsburg (Iowa)	Mishita et al., 1979
	loam			2.5		6.7 (6.12)	-	25	2.41		2.387×10^4	Yolo (California)	Mishita et al., 1979
	loam			2.5		6.7 (6.98)	-	25	2.41		2.021×10^4	Yolo (California)	Mishita et al., 1979
	16.0	50.0	34.0	0.8	1.72	7.8	0.44	15.5	-	-	$5.9\times10^3 \pm 230^+$	Idaho A	Glover et al., 1976
	9.0	54.0	37.0	2.3	0.6	2.3	0.57	16.2	-	-	$1.8\times10^{3+}$	Arkansas C	Glover et al., 1976
	31.0	53.0	16.0	3.6	0.7	3.6	0.56	17.4	-	-	$1.4\times10^3 \pm 190^+$	Illinois	Glover et al., 1976
Clay				0.6		7.8 (7.12)		30	1.20		3.563×10^4	Maltsville	Mishita et al., 1979
				0.6		7.8 (8.04)		30	1.20		4.723×10^4	Maltsville	Mishita et al., 1979
	clay			-		7-8	-	-	-	Na (90% sat. solution)	5×10^4	(Netherlands)	Beusten & Verhark, 1977
	5.0	31.0	64.0	0.7	2.4	7.9	0.42	29.6	-	-	$5.2\times10^3 \pm 970^+$	Colorado C (Rocky Flats)	Glover et al., 1976
	32.0	32.0	36.0	1.0	0.0	4.8	0.49	20.5	-	-	$2.6\times10^3 \pm 430^+$	Tennessee (Oak Ridge)	Glover et al., 1976
	32.0	32.0	36.0	2.7	0.0	5.4	0.45	16.0	-	-	$9.2\times10^2 \pm 79^+$	New York (West Valley)	Glover et al., 1976
	10.0	34.0	56.0	3.2	0.9	6.2	0.57	34.4	-	-	$2.9\times10^3 \pm 1800^+$	Arkansas A	Glover et al., 1976
	abyssal red clay			-	-	2.7	-	-	-	0.68 mol/L NaCl	25.1	- -	Erickson, 1980
	abyssal red clay			-	-	6.9	-	-	-	0.68 mol/L NaCl	4.0×10^5	- -	Erickson, 1980
Organic	organic			40.8		7.2 (7.14)	-	60	1.57	-	7.266×10^3	Egbert	Mishita et al., 1979
	organic			40.8		7.2 (7.54)	-	60	1.57	-	5.529×10^3	Egbert	Mishita et al., 1979

* When the value is bracketed, it is the pH of the extract.

** S. Carolina subsoil K_d values for americium w.r.t. calcium and sodium as competing ions over two orders of magnitude are reported in Routson et al., 1975.

\+ K_d value determined with initial americium concentration of 10^{-10} mol/L.

† Sand, silt and clay percentages exceed 100% in original report.

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TABLE 3

K_d FOR AMERICIUM

Soil Type	$\bar{K}_d^*$ (mL/g)	S.D.+	n	K_d Range (mL/g)	Lognormal Distribution≠ μ	σ
Sand	6.146×10^3	1.1159×10^4	27	82 to 4.35×10^4	3.105	0.8172
Silt	1.4351×10^4	1.1282×10^4	7	1.6×10^3 to 2.98×10^4	3.946	0.5382
Clay	6.0501×10^4	1.29×10^5	9	25.1 to 4.0×10^5	3.832	1.23
Organic	6.398×10^3	1.228×10^3	2	5.529×10^3 to 7.266×10^3	3.802	0.0839

* $\bar{K}_d$ = mean of K_d values

\+ S.D. = standard deviation of K_d values

≠ Base 10 logarithms here and in all subsequent tables.

Baes and Sharp (1981) suggested a mean value of 2.9 for the $\log_{10} K_d$ for americium, combining all soil types, and a corresponding standard deviation of 1.3 (see Table 1). Allard et al. (1977) reported K_d values for clay/mud of 2×10^2 to 1.6×10^4 mL/g and for granite of 5.0×10^3 to 1.6×10^4 mL/g. Vandergraaf (1982) recommended a range of 1×10^3 to 2×10^4 mL/g for granite. The values for granite should be similar to those for coarse-textured soil (sand).

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3.3 BISMUTH

Nothing was found in the literature on soil K_d values for bismuth, but bismuth should behave similarly to polonium because of their proximity in the Periodic Table.

3.4 LEAD

Lead is a heavy-metal cation of general environmental concern in most industrial areas. Consequently, considerable information exists about its environmental behaviour (Gerritse et al., 1982; Wolf et al., 1977; Soldatini et al., 1976; Abd-Elfattah and Wada, 1981). Unfortunately, not much K_d information is available (see Table 4).

The recommended K_d value means, standard deviations, ranges and distribution parameters for lead by soil type, based on Table 4, are given in Table 5. [redacted]

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* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

TABLE 4

K_d VALUES FOR LEAD : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	-	-	0	3.5	-	4.5-5.0	-	22	-	$[Ca^{2+}]$ = 0-0.015 mol/L	2.8×10^2	Soil C	Garritse et al., 1982
	-	-	0	3.5	-	4.5-5.0	-	22	-	$[Ca^{2+}]$ = 0-0.015 mol/L	1.3×10^3	Soil C	Garritse et al., 1982
	-	-	20	2.5	-	7.5-8.0	-	16	-	$[Ca^{2+}]$ = 0-0.015 mol/L	3.5×10^3	Soil D	Garritse et al., 1982
Organic	unpolluted organic soil			90	-	4.5	-	-	-	-	2.52×10^4	Soil A	Garritse et al., 1982
	unpolluted peat			>90	-	4-5	-	-	-	$[Ca^{2+}]$ = 0-0.015 mol/L	1.8×10^2	Peat A	Garritse et al., 1982
	unpolluted peat			>90	-	4-5	-	-	-	$[Ca^{2+}]$ = 0-0.015 mol/L	6.3×10^4	Peat A	Garritse et al., 1982
	polluted peat			>90	-	6.2	-	-	-	-	2.34×10^4	Soil B	Garritse et al., 1982
	sphagnum peat			-	-	4-5	-	-	-	-	6×10^4	- -	Wolf et al., 1977
	sphagnum peat			-	-	4-5	-	-	-	0.025 meq Ca^{2+}/mL	2×10^2	- -	Wolf et al., 1977

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[redacted]

TABLE 5

K_d FOR LEAD

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	1693	1646	3	280 to 3500	3.035	0.5527
Organic	2.7845×10^4	2.6024×10^4	4	180 to 6.3×10^4	3.954	1.150

Baes and Sharp (1981) suggested a mean value of 2.0 for the $\log_{10} K_d$ for lead, with a corresponding standard deviation of 0.7 (see Table 1). Lead should behave similarly to polonium and K_d values for polonium can be found in Section 3.6.

Lead References

Abd-Elfattah, A. and K. Wada. 1981. Adsorption of lead, copper, zinc, cobalt and cadmium by soils that differ in cation-exchange materials. J. Soil Sci. 32, 271-283.

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Wolf, A., K. Bunzl, F. Dietl and W.F. Schmidt. 1977. Effect of calcium ions on the absorption of lead (2+), copper (2+), cadmium (2+), and zinc (2+) by humic substances. Chemosphere 6, 207-213.

3.5 NEPTUNIUM

The summary on neptunium geochemistry presented by Johnston and Gillham (1980) indicates that [redacted]

(1) neptunium should exist as Np^{5+} in the form of NpO_2^+ in an oxidizing soil environment; however, it is not evident whether NpO_2^+ is also the dominant species under reducing conditions;

(2) neptunium colloids have been reported in some soil-solution experiments (Sheppard et al., 1976) and absent in others (Routson et al., 1977).

The recommended K_d value means, standard deviations, ranges and distribution parameters for neptunium by soil type, based on Table 6, are given in Table 7.

TABLE 7

K_d FOR NEPTUNIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	37.6	94.57	17	0.16 to 390	0.6782	0.9728
Silt	47.41	35.88	6	1.27 to 95	1.426	0.6925
Clay	1327.	1529.	4	41 to 3200	2.619	0.9222
Organic	857.5	101.1	2	786 to 929	2.932	0.0513

Baes and Sharp (1981) suggested a mean value of 1.0 for the $\log_{10}K_d$ for neptunium, with a corresponding standard deviation of 1.0 (see Table 1). Allard et al. (1977) reported K_d ranges of 10 to 16 mL/g for clay/mud and 25 to 50 mL/g for granites. Vandergraaf (1982) recommended a K_d of 40 to 100 mL/g for granites.

Neptunium References

Allard, B., H. Kipatsi and J. Rydberg. 1977. Sorption of long-lived radionuclides in clay and rock. Part 1. Determinations of Distribution Cofficients. KBS Technical Report 55.

TABLE 6

K_d VALUES FOR NEPTUNIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH* Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	fine sandy loam			2.4	-	5.3 (4.08)	-	15	1.65	-	3	Malbis (Louisiana)	Nishita et al., 1979
	fine sandy loam			2.4	-	5.3 (5.57)	-	15	1.65	-	18	Malbis (Louisiana)	Nishita et al., 1979
	fine sandy loam			5.7	-	5.0 (4.42)	-	15	1.52	-	3	Lyman (Maine)	Nishita et al., 1979
	fine sandy loam			5.7	-	5.0 (6.06)	-	15	1.52	-	32	Lyman (Maine)	Nishita et al., 1979
	light loam			8.4	-	6.0 (5.56)	-	15	5.29	-	26	Alkali (California)	Nishita et al., 1979
	light loam			8.4	-	6.0 (6.57)	-	15	5.29	-	108	Alkali (California)	Nishita et al., 1979
	sand			-		2.5 - 3.1	-	-	-	0.002 mol/L Ca	2.37	Burbank (Washington)	Routson et al., 1977
	sand			-		2.5 - 3.1	-	-	-	0.2 mol/L Ca	0.36	Burbank (Washington)	Routson et al., 1977
	sand			-		2.5 - 3.1	-	-	-	0.015 mol/L Na	3.9	Burbank (Washington)	Routson et al., 1977
	sand			-		2.5 - 3.1	-	-	-	3.0 mol/L Na	3.2	Burbank (Washington)	Routson et al., 1977
	sandy clay			-		2.5 - 3.1	-	-	-	0.002 mol/L Ca	0.25	South Carolina	Routson et al., 1977
	sandy clay			-		2.5 - 3.1	-	-	-	0.2 mol/L Ca	0.16	South Carolina	Routson et al., 1977
	sandy clay			-		2.5 - 3.1	-	-	-	0.015 mol/L Na	0.7	South Carolina	Routson et al., 1977
	sandy clay			-		2.5 - 3.1	-	-	-	3.0 mol/L Na	0.4	South Carolina	Routson et al., 1977
	76.0	21.2	2.8	0.43	-	8.1	-	5.94	-	-	15.4	Burbank (Richland, Washington)	Amme & Rai, 1978
	94.6	1.6	3.8	0.21	-	5.2	-	0.69	-	-	32.4	Fuquay (5-50 cm)	Amme & Rai, 1978
	sand			0.39	-	8.1	0 sat.	-	-	-	390 ± 16	N.E. Irish Sea Sediment	Fowler & Aston, 1982
Silt	silty clay loam			2.8	-	5.9 (5.83)	-	20	1.29	-	35	Sharpsburg (Iowa)	Nishita et al., 1979
	silty clay loam			2.8	-	5.9 (6.85)	-	20	1.29	-	95	Sharpsburg (Iowa)	Nishita et al., 1979
	loam			2.5	-	6.7 (6.13)	-	25	2.41	-	52	Yolo (California)	Nishita et al., 1979
	loam			2.5	-	6.7 (6.83)	-	25	2.41	-	81	Yolo (California)	Nishita et al., 1979
	12.6	65.8	21.6	3.61	-	5.3	-	16.88	-	-	1.27	Muscatine	Amme & Rai, 1978
	32.0	55.0	12.0	0.84	-	6.5	-	10.76	-	-	20.2	Ritzville	Amme & Rai, 1978
Clay	clay			0.6	-	7.8 (7.29)	-	30	1.20	-	41	Holtsville	Nishita et al., 1979
	clay			0.6	-	7.8 (8.28)	-	30	1.20	-	117	Holtsville	Nishita et al., 1979
	clay			0.86	-	8.1	0 sat.	-	-	-	1950 ± 310	NW Mediterranean sea sediment	Fowler & Aston, 1982
	clay			0.29	-	8.1	0 sat.	-	-	-	3200 ± 890	NE Atlantic sea sediment	Fowler & Aston, 1982
Organic	organic			40.8	-	7.2 (6.24)	-	60	1.57	-	786	Egbert	Nishita et al., 1979
	organic			40.8	-	7.2 (7.25)	-	60	1.57	-	929	Egbert	Nishita et al., 1979

* When value is bracketed it is extract pH.

Ames, L.L. and D. Rai. 1978. Radionuclide interactions with soil and rock media, Volume 1. U.S. Environmental Portection Agency Report, EPA 520/6-78-007.

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Vandergraaf, T.T. 1982. A compilation of sorption coefficients for radionuclides on granites and granitic rocks. Atomic Energy of Canada Limited Technical Record, TR-120*.

3.6 POLONIUM

Polonium information in the literature is extremely scarce. Bismuth should behave similarly to polonium, and also lead and polonium should behave similarly. All of the K_d information on polonium reported here (see Table 8) comes from one research program (Hansen, 1970; Hansen and Watters, 1971). The chemical form of natural polonium in soils, resulting from the decay of radium, may be similar to that of selenium (Hansen, 1970). Tellurium is also a member of Group VIA of the Periodic Table and may behave similarly to polonium. Polonium compounds with +2 and +4 oxidation states have been reported, with the preferred oxidation state being +4 (Hansen, 1970). Polonium in air is generally found as polonium dioxide (PoO_2)

* Unrestricted unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

TABLE 8

K_d VALUES FOR POLONIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	42.0	32.0	26.0	6.1	-	7.6	-	22.8	-	-	120 ± 5*	Nunn silty clay loam (A11) (Colorado)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	203 ± 22	Nunn silty clay loam (A12) (Colorado)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	310 ± 41	Nunn silty clay loam (B_1) (Colorado)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	766 ± 148	Nunn silty clay loam (B_2t) (Colorado)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	1213 ± 186	Nunn silty clay loam (B_3Ca) (Colorado)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	643 ± 85	Nunn silty clay loam (C Ca) (Colorado)	Hansen & Watters, 1971
	40	31	39	-	-	7.9	-	-	-	-	723 ± 83	Dismal silty clay loam (C) (Iowa)	Hansen & Watters, 1971
	45	42	13	2.3	-	6.5	-	5.4	-	-	192 ± 26	Lapeer loam (A_p) (Wisconsin)	Hansen & Watters, 1971
	54	22	24	-	-	6.7	-	-	-	-	206 ± 11	Lapeer loam (B_{21}) (Wisconsin)	Hansen & Watters, 1971
	59	20	21	-	-	5.5	-	-	-	-	508 ± 34	Lapeer loam (B_{22}) (Wisconsin)	Hansen & Watters, 1971
	62	15	23	-	-	5.7	-	-	-	-	814 ± 42	Lapeer loam (B_3) (Wisconsin)	Hansen & Watters, 1971
	72	18	10	-	-	7.8	-	-	-	-	275 ± 9	Lapeer loam (C_1) (Wisconsin)	Hansen & Watters, 1971
	95	0	5	-	-	5.9	-	3.0	-	-	26 ± 2	Adamsville (A_1) (Florida)	Hansen & Watters, 1971
	84	6	10	-	-	5.4	-	2.6	-	-	35 ± 3	Blanton (A_1) (Florida)	Hansen & Watters, 1971
	95	2	3	-	-	5.5	-	1.8	-	-	25 ± 2	Lakeland (A_1) (Florida)	Hansen & Watters, 1971
	97	1	2	-	-	5.5	-	1.5	-	-	17 ± 1	Leon (A_1) (Florida)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	15 ± 0.6	Leon (A_2) (Florida)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	55 ± 17	Leon (Bh) (Florida)	Hansen & Watters, 1971
	-	-	-	-	-	-	-	-	-	-	77 ± 29	Leon (C) (Florida)	Hansen & Watters, 1971
	96	2	2	-	-	5.6	-	4.6	-	-	17 ± 1	Ruskin (A_1) (Florida)	Hansen & Watters, 1971
	57	30	13	-	-	5.5	-	-	-	-	13 ± 2	Darling gravelly sandy loam (B_2) (Colorado)	Hansen & Watters, 1971
	74	17	8	-	-	5.7	-	-	-	-	30 ± 7	Darling gravelly sandy loam (B_{2ir}) (Colorado)	Hansen & Watters, 1971
	80	16	4	-	-	6.0	-	-	-	-	66 ± 9	Darling gravelly sandy loam (C_1) (Colorado)	Hansen & Watters, 1971
	74	22	4	-	-	6.0	-	-	-	-	75 ± 8	Darling gravelly sandy loam (C_2) (Colorado)	Hansen & Watters, 1971
	49	39	12	4.0	-	6.6	-	16.8	-	-	254 ± 22	Cogebic sandy loam (A_2) (Wisconsin)	Hansen & Watters, 1971
	57	30	13	-	-	5.5	-	-	-	-	371 ± 36	Cogebic sandy loam (Bir) (Wisconsin)	Hansen & Watters, 1971
	47	29	24	-	-	6.8	-	-	-	-	137 ± 5	Cogebic sandy loam (Birh) (Wisconsin)	Hansen & Watters, 1971

(continued...)

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TABLE B (Concluded)

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	68	27	5	-	-	5.9	-	-	-	-	242 ± 25	Gogebic sandy loam (B3) (Wisconsin)	Hansen & Mattera, 1971
	46	44	10	2.4	-	6.8	-	5.1	-	-	227 ± 20	Onaway fine sandy loam (Ap) (Wisconsin)	Hansen & Mattera, 1971
	46	43	11	-	-	6.9	-	-	-	-	412 ± 150	Onaway fine sandy loam (Birh) (Wisconsin)	Hansen & Mattera, 1971
	67	20	13	-	-	8.2	-	-	-	-	2248 ± 1200	Onaway fine sandy loam (C_1) (Wisconsin)	Hansen & Mattera, 1971
	67	20	13	-	-	8.4	-	-	-	-	7020 ± 3600	Onaway fine sandy loam (C_2) (Wisconsin)	Hansen & Mattera, 1971
	82	5	13	-	-	6.3	-	2.7	-	-	76 ± 11	Amite (A_1) Alabama	Hansen & Mattera, 1971
	91	0	9	-	-	5.0	-	1.9	-	-	188 ± 15	Independence (A_1) (Alabama)	Hansen & Mattera, 1971
	63	27	10	-	-	5.6	-	3.4	-	-	49 ± 3	Wickham (A_1) (Alabama)	Hansen & Mattera, 1971
Silt	11	68	21	3.8	-	5.8	-	25.2	-	-	1030 ± 49	Dinsdale silty clay loam (A) (Iowa)	Hansen & Mattera, 1971
	17	55	28	-	-	5.6	-	-	-	-	976 ± 127	Dinsdale silty clay loam (B) (Iowa)	Hansen & Mattera, 1971
	3	73	24	4.5	-	5.5	-	28.4	-	-	1136 ± 118	Muscatine silty clay loam (A) (Iowa)	Hansen & Mattera, 1971
	10	61	29	-	-	5.9	-	-	-	-	968 ± 32	Muscatine silty clay loam (B) (Iowa)	Hansen & Mattera, 1971
	11	65	24	-	-	7.8	-	-	-	-	1830 ± 210	Muscatine silty clay loam (C) (Iowa)	Hansen & Mattera, 1971
	10	80	10	2.1	-	5.9	-	11.2	-	-	970 ± 160	Fayette silt loam (Ap) (Wisconsin)	Hansen & Mattera, 1971
	8	71	21	-	-	6.2	-	-	-	-	122 ± 3	Fayette silt loam (B_1) (Wisconsin)	Hansen & Mattera, 1971
	5	66	29	-	-	6.1	-	-	-	-	92 ± 3	Fayette silt loam (B_{21}) (Wisconsin)	Hansen & Mattera, 1971
	2	66	32	-	-	5.6	-	-	-	-	597 ± 55	Fayette silt loam (B_{22}) (Wisconsin)	Hansen & Mattera, 1971
	5	65	30	-	-	5.3	-	-	-	-	80 ± 2	Fayette silt loam (B_{23}) (Wisconsin)	Hansen & Mattera, 1971
	5	66	29	-	-	5.5	-	-	-	-	772 ± 29	Fayette silt loam (C_1) (Wisconsin)	Hansen & Mattera, 1971
	33	52	15	3.8	-	5.1	-	28.9	-	-	24 ± 1	Darling gravelly sandy loam (A_1) (Colorado)	Hansen & Mattera, 1971
	27	55	18	-	-	5.5	-	16.4	-	-	405 ± 28	Congaree (A_1) (Alabama)	Hansen & Mattera, 1971

* All error terms in this table are standard error of the mean (S.E.)

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(Hansen, 1970). Hansen also reported that, for pH values of 1 to 5, 7 to 8 and 12 to 14, polonium exists mainly in dissolved forms, while for pH values of 6 to 7 and 10 to 11, most of the polonium exists as a colloid. Thus, in the acidic organic and acidic sandy soils of the Precambrian Shield, polonium may exist only in the dissolved form.

The recommended K_d value means, standard deviations, ranges and distribution parameters for polonium by soil type, based on Table 8, are given in Table 9.

TABLE 9

K_d FOR POLONIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	504.2	1215	35	13 to 7020	2.188	0.6574
Silt	692.5	535.7	13	24 to 1830	2.607	0.5789

Baes and Sharp (1981) suggested a mean value of 2.7 for the $\log_{10}K_d$ for polonium, with a corresponding standard deviation of 0.3 (see Table 1). The value for μ is higher, but the range of K_d values is narrower, than recommended for lead.

Polonium References

Baes III, C.F. and R.D. Sharp. 1981. Predicting radionuclide leaching from root zone soil for assessment applications. Oak Ridge National Laboratory Report, CONF-810606-44.

Hansen, W.R. 1970. Polonium-210 in soils and plants. Special Report on U.S. Atomic Energy Commission Contract No. AT(11-1)-1733, COO-1733-11.

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3.7 PLUTONIUM

Plutonium, like americium, has been studied extensively because of weapons testing in the 1950s. The summary on plutonium geochemistry presented by Johnston and Gillham (1980) indicates that

(1) Pu^{4+} is considered the most probable oxidation state in the environment because of reduction of Pu^{6+} to Pu^{4+} by organic materials; reduction of Pu^{4+} to Pu^{3+} could occur at pH < 6 under anaerobic conditions;

(2) plutonium adsorption is a function of oxidation state (Pu^{6+} is adsorbed less than Pu^{4+}), organic matter content and solution pH;

(3) K_d values for plutonium reported in the literature were often obtained without knowledge of the oxidation state, and caution must be used in interpreting results that use these K_d values.

Most K_d values in the literature apply to aerobic conditions. The recommended K_d value means, standard deviations, ranges and distribution parameters for plutonium by soil type, based on Table 10, are given in Table 11.

TABLE 11

K_d FOR PLUTONIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	1.041×10^3	1.568×10^3	19	33 to 6.865×10^3	2.663	0.5964
Silt	1.3871×10^4	3.0836×10^4	8	230 to 9.0×10^4	3.474	0.7906
Clay	4.2842×10^4	6.8934×10^4	13	316 to 1.9×10^5	3.706	1.047
Organic	2.2902×10^4	2.8181×10^4	4	1.655×10^3 to 6.2×10^4	3.970	0.7469

Baes and Sharp (1981) suggested a mean value of 3.3 for the $\log_{10} K_d$ for plutonium, with a corresponding standard deviation of 1.0 (see

[redacted]

TABLE 10

K_d VALUES FOR PLUTONIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH* Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	fine sandy loam			2.4	-	5.3 (4.08)	-	15	1.65	-	8.5×10^2	Malbis (Louisiana)	Nishita et al., 1979
	fine sandy loam			2.4	-	5.3 (5.57)	-	15	1.65	-	1.515×10^3	Malbis (Louisiana)	Nishita et al., 1979
	fine sandy loam			5.7	-	5.0 (4.42)	-	15	1.52	-	9.58×10^2	Lyman (Maine)	Nishita et al., 1979
	fine sandy loam			5.7	-	5.0 (6.06)	-	15	1.52	-	3.3×10^1	Lyman (Maine)	Nishita et al., 1979
	light loam			8.4	-	6.0 (5.56)	-	15	5.29	-	6.85×10^3 **	Aiken (California)	Nishita et al., 1979
	light loam			8.4	-	6.0 (6.57)	-	15	5.29	-	1.352×10^3	Aiken (California)	Nishita et al., 1979
	coarse sand			-	-	7-8	-	-	-	Na (90% sat. solution)	2×10^2	(Netherlands)	Hamstra & Verhoek, 1977
	44.0	20.0	36.0	2.4	0.4	5.7	0.41	20.0	-	-	$2.2\times10^1 \pm 460$ ♦	Colorado A (Rocky Flats)	Glover et al., 1976
	64.0	14.0	22.0	3.4	0.3	5.6	0.52	17.5	-	-	$2.0\times10^2 \pm 26$ ♦	Colorado B (Sugar Loaf)	Glover et al., 1976
	44.0	24.0	32.0	0.2	7.9	8.3	0.43	13.8	-	-	$3.2\times10^2 \pm 26$ ♦	Idaho B	Glover et al., 1976
	66.0	11.0?	23.0	0.3	5.2	8.0	0.47	8.2	-	-	$6.9\times10^2 \pm 110$ ♦	Idaho C	Glover et al., 1976
	38.0	32.0	30.0	0.1	0.0	7.5	0.45	17.5	-	-	$2.1\times10^3 \pm 640$ ♦	Idaho D	Glover et al., 1976
	74.0	12.0	8.0	0.3	0.6	8.0	0.43	6.4	-	-	$1.0\times10^2 \pm 7$ ♦	Washington A (Hanford)	Glover et al., 1976
	74.0	12.0	14.0	0.1	0.0	8.2	0.44	5.8	-	-	$4.3\times10^2 \pm 27$ ♦	Washington B (Hanford)	Glover et al., 1976
	78.0	2.0	20.0	0.7	0.2	5.4	0.54	2.9	-	-	$2.8\times10^2 \pm 5$ ♦	S. Carolina (Barnwell)	Glover et al., 1976
	48.0	34.0?	18.0	0.7	0.2	6.4	0.49	7.0	-	-	$1.0\times10^2 \pm 5$ ♦	New Mexico (Los Alamos)	Glover et al., 1976
	82.0	9.0	9.0	0.6	0.7	4.8	0.57	3.8	-	-	$8.0\times10^1 \pm 3$ ♦	Arkansas B	Glover et al., 1976
	subsoil sand			-	2.0	8.6 (6.5)	-	5.0	-	-	1.314×10^3	Hanford	Rhodes, 1957
	subsoil sand			-	2.0	8.6 (9.3)	-	5.0	-	-	2.0×10^2	Hanford	Rhodes, 1957
Silt	silty clay loam			2.8	-	5.9 (5.83)	-	20	1.29	-	6.302×10^3	Sharpsburg (Iowa)	Nishita et al., 1979
	silty clay loam			2.8	-	5.9 (6.85)	-	20	1.29	-	3.024×10^3	Sharpsburg (Iowa)	Nishita et al., 1979
	loam			2.5	-	6.7 (6.13)	-	25	2.41	-	4.938×10^3	Yolo (California)	Nishita et al., 1979
	loam			2.5	-	6.7 (6.83)	-	25	2.41	-	4.341×10^3	Yolo (California)	Nishita et al., 1979
	16.0	50.0	34.0	0.8	17.2	7.8	0.44	15.5	-	-	$1.7\times10^3 \pm 70$ ♦	Idaho A	Glover et al., 1976
	9.0	54.9	37.0	2.3	0.6	2.3	0.57	16.2	-	-	$4.3\times10^2 \pm 23$ ♦	Arkansas C	Glover et al., 1976
	31.0	53.0	16.0	3.6	0.7	3.6	0.56	17.4	-	-	$2.3\times10^2 \pm 10$ ♦	Illinois	Glover et al., 1976
	silt suspended in seawater				-	-	-	-	-	-	9×10^4		Pillai & Mathew, 1975
Clay	clay			0.6	-	7.8 (7.29)	-	30	1.20	-	7.44×10^2	Holtsville	Nishita et al., 1979
	clay			0.6	-	7.8 (8.28)	-	30	1.20	-	3.61×10^2	Holtsville	Nishita et al., 1979
	clay			-	-	7-8	-	-	-	Na (90% sat. solution)	1×10^4	(Netherlands)	Hamstra & Verhoek, 1977
	5.0	31.0	64.0	0.7	2.4	7.9	0.42	29.6	-	-	$1.9\times10^3 \pm 110$ ♦	Colorado C (Rocky Flats)	Glover et al., 1976
	32.0	32.0	36.0	1.0	0.0	4.8	0.49	20.5	-	-	$2.6\times10^3 \pm 640$ ♦	Tennessee (Oak Ridge)	Glover et al., 1976
	32.0	32.0	36.0	2.7	0.0	5.4	0.45	16.0	-	-	$8.1\times10^2 \pm 130$ ♦	New York (West Valley)	Glover et al., 1976
	10.0	34.0	56.0	3.2	0.9	6.2	0.57	34.4	-	-	$7.1\times10^2 \pm 36$ ♦	Arkansas A	Glover et al., 1976
	treated clay			-	-	4.0	-	-	-	5 mol/L Ca^{2+} $(Ca\ NO_3)_2$	1.9×10^5	20% Pu(VI)	Bondietti & Reynolds, 1976
	Ca-sat. soil clay fraction				-	6.5	-	-	-	5 mol/L Ca^{2+} $(Ca\ NO_3)_2$	1.04×10^5	^{239}Pu(IV)	Bondietti et al., 1975
	Ca-sat. soil clay fraction				-	6.5	-	-	-	5 mol/L Ca^{2+} $(Ca\ NO_3)_2$	1.68×10^5	^{239}Pu(IV)	Bondietti et al., 1975
	Ca-sat. soil clay fraction				-	6.5	-	-	-	5 mol/L Ca^{2+} (Ca-acetate)	7.5×10^4	^{239}Pu(VI)	Bondietti et al., 1975
	abyssal red clay			-	-	2.7	-	-	-	0.68 mol/L NaCl	3.16×10^2	--	Erickson, 1980
	abyssal red clay			-	-	6.9	-	-	-	0.68 M NaCl solution	2.5×10^3	--	Erickson, 1980
Organic	organic			40.8	-	7.2 (6.24)	-	60	1.57	-	2.951×10^3	Egbert	Nishita et al., 1979
	organic			40.8	-	7.2 (7.25)	-	60	1.57	-	1.655×10^3	Egbert	Nishita et al., 1979
	bone charcoal			-	-	7.0	-	-	-	-	6.2×10^4	--	Tamura, 1972
	coconut charcoal			-	-	7.0	-	-	-	-	2.5×10^4	--	Tamura, 1972

* When the value is bracketed, it is pH of the extract.

** Nishita et al., 1978 report plutonium K_d values w.r.t. pH for this soil.

♦ K_d value determined with initial plutonium concentration of 10^{-8} mol/L (data for 10^{-7} and 10^{-6} mol/L can be found in the reference).

? Sand, silt and clay percentages exceeded 100% in original report.

Table 1). Allard et al. (1977) reported that the plutonium K_d range for both granite and clay/mud is 6.3×10^1 to 1.6×10^2 mL/g. Vandergraaf (1982) recommended a K_d value range for plutonium of 2.8×10^2 to 2.0×10^3 mL/g for granite.

Plutonium References

Allard, B., H. Kipatsi and J. Rydberg. 1977. Sorption of long-lived radionuclides in clay and rock. Part 1. Determination of distribution coefficients. KBS Technical Report 55.

Baes III, C.F. and R.D. Sharp. 1981. Predicting radionuclide leaching from root zone soil for assessment applications. Oak Ridge National Laboratory Report, CONF-810606-44.

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Vandergraaf, T.T. 1982. A compilation of sorption coefficients for radionuclides on granites and granitic rocks. Atomic Energy of Canada Limited Technical Record, TR-120*.

3.8 PROTACTINIUM

Nothing on K_d values or soil chemistry for protactinium was found in the literature. We suggest that K_d values for thorium or uranium be used, or even some combination of the values for these elements, such as ½(Th+U).

3.9 RADIUM

The interaction of radium with geological materials and soils, and the environmental behaviour of radium have been documented by Gillham et al. (1981b); Nathwani and Phillips (1979), and Sheppard (1980), respectively. The K_d values for radium vary from 50 to 1000 mL/g (Gillham et al., 1981). Johnston and Gillham (1980) summarized the information relevant to K_d as follows:

(1) Radium is present as Ra^{2+} in the pH range 4 to 8, and does not readily form complex species.

(2) Radium can be expected to coprecipitate with $BaSO_4$, carbonates and ferric hydroxides.

(3) Cation exchange is an important adsorption mechanism, since K_d values have been correlated to cation exchange capacity (CEC).

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* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

The recommended K_d value means, standard deviations, ranges and distribution parameters for radium by soil type, based on Table 12, are given in Table 13.

TABLE 13

K_d FOR RADIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	1.0435×10^4	2.0845×10^4	3	106 to 3.8×10^4	3.402	1.289
Silt	3.0×10^5	4.3566×10^5	4	2.0×10^4 to 9.5×10^5	5.10	0.6862
Clay	1.5637×10^4	1.7216×10^4	8	696 to 5.6×10^4	3.961	0.5522

The K_d values for strontium may be used as a guide because of the chemical similarity of radium and strontium. Baes and Sharp (1981) suggested a mean value of 1.4 for the $\log_{10} K_d$ for strontium, with a corresponding standard deviation of 0.9 (see Table 1). Allard et al. (1977) reported a K_d range for radium of 40 to 79 mL/g for clay/mud and 63 to 100 mL/g for granite. Vandergraaf (1982) recommended a K_d range of 5 to 5000 mL/g for granite. Since no data were found for organic soils, the radium K_d value for clay, or the strontium K_d value for organic soil, is recommended.

Radium References

Allard, B., H. Kipatsi and J. Rydberg. 1977. Adsorption of long-lived radionuclides in clay and bedrock. Part 1. Determination of distribution coefficients. KBS Technical Report 55.

Baes III, C.F. and R.D. Sharp. 1981. Predicting radionuclide leaching from root zone soil for assessment applications. Oak Ridge National Laboratory Report, CONF-810606-44.

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TABLE 12

K_d VALUES FOR RADIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	93.0	5.0	2.0	0.05	40.8	7.8 ($CaCl_2$)	-	1.19	-	Initial Ra conc. before soil contact 1.6×10^{-5} mg/L	106 ± 16	Leamington (6) medium sand	Gillham et al., 1981b
	91.1	6.8	1.3	3.1	-	5.2	-	10.9	-	Ca^{2+} 0.05 mol/L	4×10^3	St. Thomas	Nathwani & Phillips, 1979
	91.1	6.8	1.3	3.1	-	5.2	-	10.9	-	no Ca	3.8×10^4	St. Thomas	Nathwani & Phillips, 1979
Silt	35.0	36.0	29.0	0.41	33.8	8.5 ($CaCl_2$)	-	8.32	-	Initial Ra conc. before soil contact 4×10^{-5} mg/L	1262 ± 370	WNRE+(2) clay loam	Gillham et al., 1981b
	6.7	47.9	45.4	16.2	-	5.4	-	34.7	-	Ca^{2+} 0.05 mol/L	1.1×10^5	Wendover	Nathwani & Phillips, 1979
	6.7	47.9	45.4	16.2	-	5.4	-	34.7	-	no Ca	9.5×10^5	Wendover	Nathwani & Phillips, 1979
	43.7	48.9	7.4	1.0	-	4.3	-	10.4	-	Ca^{2+} 0.05 mol/L	2.0×10^4	Grimsby	Nathwani & Phillips, 1979
	43.7	48.9	7.4	1.0	-	4.3	-	10.4	-	no Ca	1.2×10^5	Grimsby	Nathwani & Phillips, 1979
Clay	31.0	34.0	35.0	0.81	5.2	7.8 ($CaCl_2$)	-	31.48	-	Initial Ra conc. before soil contact 3.7×10^{-5} mg/L	696 ± 185	Alberta clay loam	Gillham et al., 1981b
	clay			-	-	7.55?	-	-	-	Na^+ 288 mg/L Ca^{2+} 75 mg/L	5.6×10^4	clay, mud	Allard et al., 1977
	clay sediment				-	-	-	-	-	-	$13.3 \times 10^{3*}$	clay sediment (Pacific)	Cochran & Krishnaswami, 1980
	clay sediment				-	-	-	-	-	-	$10.5 \times 10^{3*}$	clay sediment (Pacific)	Cochran & Krishnaswami, 1980
	clay sediment				-	-	-	-	-	-	$8.0 \times 10^{3*}$	clay sediment (Pacific)	Cochran & Krishnaswami, 1980
	clay sediment				-	-	-	-	-	-	$4.3 \times 10^{3*}$	clay sediment (Pacific)	Cochran & Krishnaswami, 1980
	clay sediment				-	-	-	-	-	-	$14.9 \times 10^{3*}$	clay sediment (Pacific)	Cochran & Krishnaswami, 1980
	clay sediment				-	-	-	-	-	-	$17.4 \times 10^{3*}$	clay sediment (Pacific)	Cochran & Krishnaswami, 1980

+ WNRE - Whiteshell Nuclear Research Establishment, Pinawa, Manitoba

* Minimum values reported and data based on desorption of deep sea clays.

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3.10 THORIUM

Little information is available in the literature on thorium interactions in the environment; however, two brief reviews of thorium chemistry are available (Rancon, 1973; Sheppard, 1980). Johnston and Gillham (1980) summarized the information relevant to K_d as follows:

(1) The primary thorium adsorption mechanism is ion exchange.

(2) In non-calcareous soils, thorium adsorption is extremely sensitive to initial thorium solution concentrations. In organic materials, increased pH causes increased humic acid solubility and thorium complexation, resulting in lower K_d values. In calcareous soils, K_d values are high (> 10 mL/g), regardless of pH or thorium concentration, because of the buffering capacity of the soil and the precipitation of $Th(OH)_4$.

(3) K_d values are generally high, (> 10^3 mL/g), in dilute solutions, indicating limited thorium migration.

The recommended K_d value means, standard deviations, ranges and distribution parameters for thorium by soil type, based on Table 14, are given in Table 15.

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* Unrestricted, unpublished report, available from SDDO, Atomic Energy Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

TABLE 14

K_d VALUES FOR THORIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	45 (% SiO_2)	-	30	< 1	25 (% carbonate)	7.0	-	-	-	Th Conc. (1 g/L)	1.5×10^5	Cadarache sediment	Rançon, 1973
Clay	40 (% SiO_2)	-	60	0	0 (% carbonate)	3.2	-	-	-	Th Conc. (1 g/L)	8	clay schist	Rançon, 1973
	40 (% SiO_2)	-	60	0	0 (% carbonate)	4.8	-	-	-	Th Conc. (0.1 g/L)	1×10^5	clay schist	Rançon, 1973
Organic	5 (% SiO_2)	-	12	60	23 (% carbonate)	6.7	-	-	-	Th Conc. (1 g/L)	8×10^4	river peat	Rançon, 1973
	5 (% SiO_2)	-	12	60	23 (% carbonate)	7.4	-	-	-	Th Conc. (0.1 g/L)	1.5×10^4	river peat	Rançon, 1973

[redacted]

TABLE 15

K_d FOR THORIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Clay	5.0×10^4	7.0710×10^4	2	8.0 to 1.0×10^5	2.95	2.90
Organic	4.75×10^4	4.5962×10^4	2	1.5×10^4 to 8.0×10^4	4.54	0.5141

Baes and Sharp (1981) suggested a mean value of 4.8 for the $\log_{10} K_d$ for thorium, with a corresponding standard deviation of 0.6 (see Table 1). Allard et al. (1977) reported K_d ranges from 40 to 316 mL/g and 500 to 1260 mL/g for clay/mud and granite, respectively. Vandergraaf (1982) recommended a K_d value for thorium of 850 mL/g for granite.

Thorium References

Allard, B., H. Kipatsi and J. Rydberg. 1977. Sorption of long-lived radionuclides in clay and rock. Part 1. Determination of Distribution Coefficients. KBS Technical Report 55.

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* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

[redacted]

3.11 URANIUM

Several reviews of uranium chemistry exist (Harmsen and de Haan, 1980; Borovec, 1981; Sheppard, 1980), but few K_d values have been reported in the literature. The summary on uranium geochemistry presented by Johnston and Gillham (1980) indicates that

(1) in oxidizing environments, U^{6+} compounds are stable and can precipitate, whereas U^{4+} is stable in a reducing environment and would precipitate as UO_2; thus the oxidation-reduction status is important;

(2) soluble uranium (U^{6+}) can be adsorbed or reduced by organic matter; if U^{6+} is reduced to U^{4+}, precipitation can occur;

(3) UO_2^{2+} can be adsorbed by clay minerals by cation exchange, but may also form complexes with anions such as carbonate or phosphate.

Borovec (1981) indicated that K_d values for uranium for clay minerals range from 50 to 1000 mL/g and for peat from 10^4 to 10^6 mL/g. The recommended K_d value means, standard deviations, ranges and distribution parameters for uranium by soil type, based on Table 16, are given in Table 17.

TABLE 17

K_d FOR URANIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	8.065	11.22	2	0.13 to 16.0	0.159	1.478
Clay	2.6349×10^5	4.5597×10^5	3	200 to 7.9×10^5	3.543	2.040

Baes and Sharp (1981) suggested a mean value of 1.6 for the $\log_{10} K_d$ for uranium, with a corresponding standard deviation of 0.6 (see

TABLE 16

K_d VALUES FOR URANIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH* Saturated Paste	E (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	45 (% SiO_2)	-	30	< 1	25 (% carbonate)	< 7	-	-	-	-	16	Cadarache sediment	Rançon, 1973
	sandy soil			-	-	-	-	-	-	4.3 μg CO_3^{2-}/mL	0.13	- -	Tamamoto et al., 1973
Silt	alluvial soil			-	-	-	-	-	-	4.3 μg CO_3^{2-}/mL	0.25	- -	Tamamoto et al., 1973
Clay	40 (% SiO_2)	-	60	0	0 (% carbonate)	< 7	-	-	-	-	270	altered schist	Rançon, 1973
	abyssal red clay			-	-	2.8	-	-	-	0.68 mol/L NaCl	200	- -	Erickson, 1980
	abyssal red clay			-	-	7.1	-	-	-	0.68 mol/L NaCl	7.9×10^5	- -	Erickson, 1980
Organic	5 (% SiO_2)	-	12	60	23 (% carbonate)	< 7	-	-	-	-	33	organic peat	Rançon, 1973

Table 1). Allard et al. (1977) reported K_d ranges from 2.5 to 20 mL/g for clay/mud and 4 to 13 mL/g for granites. Vandergraaf (1982) recommended a K_d range of 0.4 to 10 mL/g for granites.

Uranium References

Allard, B., H. Kipatsi and J. Rydberg. 1977. Sorption of long-lived radionuclides in clay and rock. Part 1. Determination of Distribution Coefficients. KBS Technical Report 55.

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* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

4. DISTRIBUTION COEFFICIENTS FOR FISSION PRODUCTS

4.1 CALCIUM

K_d values for calcium reported by Graham (1973) and Graham and Silva (1979) vary from 1×10^{-3} to 9.8 mL/g; however, there is some confusion about the units. Wong et al. (1983) reported K_d values of 1117 and 1900 mL/g for a sand and a muck soil, respectively. It is recommended that the K_d values for strontium be used for calcium (see Section 4.11).

Calcium References

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Wong, K.V., S. Sengupta, D. Dasgupta, E.L. Daly, Jr., N. Nemerow and H.P. Gerrish. 1983. Heavy metal migration in soil-leachate systems. Biocycle 24, 30-33.

4.2 CARBON

Allard et al. (1981) studied the sorption of $H^{14}CO_3^-$ on some solids using the batch technique. The sorption of ^{14}C was generally low, but appeared to increase with increasing calcium content of the solid. Retardation factors of up to 3 (i.e., three times slower transport of ^{14}C than of water) were measured for calcite. Concrete will probably retain most of the ^{14}C, and a retardation factor >10 might be expected for a bentonite-quartz mixture ($K_d = 2.2 \times 10^{-6}$ mL/g). Owing to the paucity of information, a conservative retardation factor of 1, or a K_d of 0 mL/g, is recommended.

[redacted]

Carbon Reference

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4.3 CESIUM

The work of Gillham et al. (1981a) is the most extensive on K_d values for cesium for Canadian soils. Their study showed that for 15 Canadian soils, K_d values for cesium ranged from 1×10^2 to 2×10^4 mL/g, but there was no significant correlation between the K_d value and measured soil properties such as CEC, major cation concentration, clay mineral composition, organic matter content and pH. In more than half of the samples, however, the K_d values were related significantly to the natural exchangeable cesium content of the soil, and this must be accounted for in sorption studies.

The recommended K_d value means, standard deviations, ranges and distribution parameters for cesium, based on Table 18, are given in Table 19.

TABLE 19

K_d FOR CESIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	2163	3226	24	10 to 1.0×10^4	2.668	0.9332
Silt	1.1395×10^4	7899	20	650 to 3.0×10^4	3.912	0.4227
Clay	8379	1.3613×10^4	5	65 to 3.15×10^4	2.945	1.216

Baes and Sharp (1981) suggested a mean value of 3.0 for the $\log_{10} K_d$ for cesium, with a corresponding standard deviation of 0.8 (see Table 1). Allard et al. (1977) reported K_d ranges from 6 to 32 mL/g and 32

TABLE 18

K_d VALUES FOR CESIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	100	-	-	0.03	41.3	8.3 ($CaCl_2$)	-	1.4	-	see ref.	1.19×10^2	Soil #4 (WNRE)	Gillham et al., 1981a
	93	5	2	0.05	40.8	7.8 ($CaCl_2$)	-	1.2	-	see ref.	1.37×10^3	Soil #6 (Leamington)	Gillham et al., 1981a
	96	4	0	0.51	0	6.3 ($CaCl_2$)	-	1.1	-	see ref.	7.4×10^3	Soil #7 (CRNL)+	Gillham et al., 1981a
	52	45	3	0.08	0	5.0 ($CaCl_2$)	-	1.6	-	see ref.	1.0×10^4	Soil #8 (North Bay)	Gillham et al., 1981a
	59	24	17	0.40	0	6.5 ($CaCl_2$)	-	1.9	-	see ref.	1.0×10^3	Soil #10 (WNRE)	Gillham et al., 1981a
	62	31	7	0.38	18.3	7.6 ($CaCl_2$)	-	2.2	-	see ref.	1.0×10^4	Soil #11 (WNRE)	Gillham et al., 1981a
	52	39	9	0.33	43.4	8.0 ($CaCl_2$)	-	0.7	-	see ref.	1.5×10^2	Soil #12 (BNPD)+	Gillham et al., 1981a
	96	2	2	0.30	11.1	8.0 ($CaCl_2$)	-	0.4	-	see ref.	5.0×10^2	Soil #13 (C.F.B. Borden)	Gillham et al., 1981a
	60	22	18	2.05	7.1	7.8 ($CaCl_2$)	-	21.2	-	see ref.	1.0×10^2	Soil #16 (Alberta)	Gillham et al., 1981a
	87	9	4	0.10	0.07	8.21 ($CaCl_2$)	-	5.0		see ref.	1.5×10^3 ± 320	Sediment B (Solution 1)	Serne et al., 1978
	-	-	-	-	-	-	-	-	-	SiO_2	18	Iron & silty sands	Tyuchovitz, 1981
	94	6 (<0.074 mm)		-	-	-	-	-	-	-	9.5×10^2	Composite soil	Schwalz, 1972
		river sand		-	-	7-8	-	-	-	9% NaCl solution	10	River sand	Hamstra & Verkerk, 1977
		subsoil sand		-	2	8.6	-	5	-	4 mol/L NaCl	16.4	Hanford subsoil	Rhodes, 1957
		clinoptilolite		-	-	-	-	-	-	-	4.2×10^3	Clinoptilolite (Idaho)	Wildung & Rhodes, 1963
		Barbark soil		-	-	-	-	-	-	groundwater	9.0×10^3	Barbark soil	Rajek & Ames, 1968
		Barbark soil		-	-	-	-	-	-	3 mol/L $NaNO_3$	4.66×10^2	Barbark soil	Rajek & Ames, 1968
		Barbark soil		-	-	-	-	-	-	0.5 mol/L NaCl	1.09×10^3	Barbark soil	Rajek & Ames, 1968
		Barbark soil		-	-	-	-	-	-	0.25 mol/L $CaCl_2$	5.21×10^3	Barbark soil	Rajek & Ames, 1968
	84	13	3	0.16	2.8	-	-	5.1	0.63	0.2 mol/L NaCl see ref.	2.40×10^3	Barbark sand (average profile)	Routson, 1973
	63	32	5	0.21	1.36	-	-	5.3	1.02	0.2 mol/L NaCl see ref.	3.51×10^3	Ephrata sand (average profile)	Routson, 1973
		sand		-	-	7.0	-	-	-	-	98*	Four mile creek	Zelazny et al., 1978
		sand		-	-	7.0	-	-	-	-	84*	Pen branch	Zelazny et al., 1978
		sand		-	-	7.0	-	-	-	-	40*	Par Pond	Zelazny et al., 1978
Silt	36	35	29	0.43	33.6	8.1 ($CaCl_2$)	-	8.4	-	see ref.	1.78×10^4	Soil #1 (WNRE)	Gillham et al., 1981a
	34	35	31	0.40	34.1	8.1 ($CaCl_2$)	-	8.6	-	see ref.	1.84×10^4	Soil #3 (WNRE)	Gillham et al., 1981a
	28	41	31	1.27	21.1	7.7 ($CaCl_2$)	-	5.9	-	see ref.	1.81×10^3	Soil #5 (Leamington)	Gillham et al., 1981a
	12	55	33	0.35	0	6.7 ($CaCl_2$)	-	10.2	-	see ref.	2.0×10^4	Soil #9 (North Bay)	Gillham et al., 1981a
	34	34	32	0.85	5.1	7.7 ($CaCl_2$)	-	32.7	-	see ref.	1.0×10^4	Soil #14 (Alberta)	Gillham et al., 1981a
	45	44	11	0.14	1.4	8.83 ($CaCl_2$)	-	12.0	-	see ref.	1.35×10^4 ± 4741	Sediment A (Solution 1)	Serne et al., 1978
		alluvial silt loam (Ap)		-	-	-	-	-	-	-	3×10^4	Captina silt loam (Ap)	Rogowski & Tamura, 1965
		medium loam		-	-	-	-	-	-	0.1 mol/L HNO_3	6.5×10^2	Podzolic soil	Alekseikin, 1965
	31	69	0	-	-	-	-	2.6	-	-	5.32×10^3	alluvial soil (Cadarache)	Rançon, 1972
	38	62	0	-	-	-	-	2.7	-	-	9.55×10^3	alluvial soil (Cadarache)	Rançon, 1972
	18	66	16	-	-	-	-	6.3	-	-	1.04×10^4	Vindobonian sed. (Cadarache)	Rançon, 1972
	40	45	15	-	-	-	-	1.8	-	-	1.14×10^4	Vindobonian sed. (Cadarache)	Rançon, 1972
	34	52	14	-	-	-	-	4.9	-	-	7.3×10^3	Vindobonian sed. (Cadarache)	Rançon, 1972
	45	47	8	-	-	-	-	1.5	-	-	6.2×10^3	Vindobonian sed. (Cadarache)	Rançon, 1972
	7	92	1	-	-	-	-	4.2	-	-	2.07×10^4	sandy-clay sed. (Durance R.)	Rançon, 1972
	18	71	11	-	-	-	-	3.5	-	-	1.52×10^4	sandy-clay sed. (Durance R.)	Rançon, 1972
	3	96	1	-	-	-	-	5.2	-	-	2.0×10^4	sandy-clay sed. (Durance R.)	Rançon, 1972
		silty clay, (9+)		-	-	-	-	-	-	-	3.0×10^3	silty clay (Idaho)	Wildung & Rhodes, 1963
		silty clay, PC-2		-	-	-	-	-	-	-	2.7×10^3	silty clay (Idaho)	Wildung & Rhodes, 1963
	44	50	6	0.23	3.8	-	-	11.0	1.21	0.2 mol/L NaCl	3.96×10^3	Ritzville silt (avg. profile)	Routson, 1973
Clay	31	34	35	0.81	5.2	7.8 ($CaCl_2$)	-	31.5	-	see ref.	1.0×10^4	Soil #15 (Alberta)	Gillham et al., 1981a
	10% < 0.02 mm			-	-	-	-	35	-	-	3.15×10^4	very fine suspended sediments (Durance River)	Rançon, 1972
		clay		-	-	7-8	-	-	-	9% NaCl solution	2×10^2	clay	Hamstra & Verkerk, 1977
	Savannah River sed. (<100 μm)			-		-	-	-	-	NaCl	65 (σ=8)	Savannah River sediments	Elprince et al., 1977
	Savannah River sed. (<100 μm)			-		7.0	-	-	-	-	130*	Savannah River clay	Zelazny et al., 1978

* CRNL - Chalk River Nuclear Laboratories, Chalk River, Ontario

\+ BNPD - Bruce Nuclear Power Development

* Also data for sorption versus pH, see reference.

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to 794 mL/g for clay/mud and granite, respectively. Vandergraaf (1982) recommended a K_d range of 40 to 1000 mL/g for granite.

Cesium References

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Zelazny, L.W., D.C. Martens, A.M. El-Prince and C.I. Rich. 1978. Effect of temperature and hydroxy-aluminum interlayers on cesium selectivity and fixation in river suspensions and soils. ORO-4851-2.

4.4 IODINE

The most extensive study of iodine adsorption on soil was that of Wildung et al. (1974). Johnston and Gillham (1980) have summarized the known soil chemistry of iodine as follows:

(1) the most stable form of iodine in both oxidizing and reducing environments is iodide, I^-. Because the predominant iodine species is an anion, ion exchange would not be important in soil adsorption, particularly at neutral or high pH values.

(2) organic matter appears to be a significant factor in iodine adsorption.

(3) K_d values for iodine range from 0.1 to 50 mL/g, depending on the form of the iodine and the pH of the solution. The maximum K_d value would be obtained for I^- at a pH of 4 to 6.

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* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

Since no K_d values for specific soils were found in the literature, the multiple-regression equations (based on iodide (I^-) and methyl iodide (CH_3I) interactions with 22 soils) reported by Wildung et al. (1974) were applied to soils already described in this report. The K_d values were calculated only for soils whose properties were within the range of the soils used to generate the multiple-regression equations. The equations for iodide K_d and methyl iodide K_d differ, and are

$$K_{d_{iodide}} = 0.33\ X_1 + 0.09\ X_3 \qquad (5)$$

$$K_{d_{\substack{methyl\\iodide}}} = 0.027\ X_2 + 0.10\ X_3 \qquad (6)$$

where X_1 is the silt content (range for equation development is 17.6 to 58.0%),

X_2 is the clay content (range for equation development is 3.8 to 46.6%), and

X_3 is the organic carbon content (range for equation development is 0.23 to 28.8%).

The recommended K_d value means, standard deviations, ranges and distribution parameters by soil type (using Equation (6) since it will predict the lowest K_d values because methyl iodide is more highly mobile), based on Table 20, are given in Table 21.

TABLE 21

K_d FOR IODINE

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	0.5514	0.3595	7	0.2 to 1.210	-0.3404	0.2929
Silt	0.9145	0.3201	11	0.18 to 1.50	-7.99×10^{-2}	0.2351
Clay	1.293	0.3697	4	1.03 to 1.83	9.952×10^{-2}	0.1140

TABLE 20

K_d VALUES FOR IODINE : LITERATURE SURVEY SUMMARY

Soil Type	% Silt	% Clay	% Organic	Calculated K_d (mL/g)	Soil Location or Description	Reference
Sand	31	7	0.38	0.23	Soil #11 (WNRE)	Gillham et al., 1981a
	39	9	0.33	0.28	Soil #12 (BNPD)	Gillham et al., 1981a
	22	18	2.05	0.69	Soil #16 (Alberta)	Gillham et al., 1981a
	29	5.8	0.45	0.20	Hanford A	Ames & Rai, 1978
	39.4	18.0	0.60	0.55	Idaho A	Ames & Rai, 1978
	28.4	22.4	0.98	0.70	Idaho D	Glover et al., 1976
	20.0	36.0	2.4	1.21	Colorado A	Glover et al., 1976
Silt	50	34	0.8	1.00	Idaho A	Glover et al., 1976
	54	37	2.3	1.23	Arkansas C	Glover et al., 1976
	53	16	3.6	0.79	Illinois	Glover et al., 1976
	35	29	0.43	0.83	Soil #1 (WNRE)	Gillham et al., 1981a
	36	29	0.41	0.82	Soil #2 (WNRE)	Gillham et al., 1981a
	35	31	0.40	0.88	Soil #3 (WNRE)	Gillham et al., 1981a
	41	31	1.27	0.96	Soil #5 (Lemington)	Gillham et al., 1981a
	55	33	0.35	0.93	Soil #9 (North Bay)	Gillham et al., 1981a
	34	32	0.85	0.94	Soil #14 (Alberta)	Gillham et al., 1981a
	-	29.1	7.1	1.50	Brookston silt (average profile)	Juo & Barber, 1970
	50	6	0.23	0.18	Ritzville silt	Routson, 1973
Clay	34	35	0.81	1.03	Soil #15 (Alberta)	Gillham et al., 1981a
	32	36	1.0	1.07	Tennessee (Oak Ridge)	Glover et al., 1976
	32	36	2.7	1.24	New York (West Valley)	Glover et al., 1976
	34	56	3.2	1.83	Arkansas A	Glover et al., 1976

The K_d range reported for methyl iodide in soil was 0.1 to 3.1 (Wildung et al., 1974). The multiple-regression equations were developed for mineral soils. A single value of > 30 mL/g for charcoal is pertinent to organic soil (Nowak, 1981). Vandergraaf (1982) recommended a K_d value for iodine of 0 mL/g, until a cationic species of iodine is identified.

Iodine References

Ames, L.L. and D. Rai. 1978. Radionuclide interactions with soil and rock media, Volume 1. U.S. Environmental Protection Agency Report, EPA 520/6-78-007.

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Vandergraaf, T.T. 1982. A compilation of sorption coefficients for radionuclides on granites and granitic rocks. Atomic Energy of Canada Limited Technical Record, TR-120*.

Wildung, R.E., R.C. Routson, R.J. Serne and T.R. Garland. 1974. Pertechnetate, iodide, and methyl iodide retention by surface soils. BNWL-SA-5195.

4.5 MOLYBDENUM

No data were found in the literature for molybdenum; however, because of its position in the Periodic Table, the K_d values for technetium can be used for molybdenum.

4.6 NICKEL

Little information exists for soil adsorption of nickel. Swanson (1981) reported a range of K_d values for nickel of 5.2×10^1 to 1.2×10^4 mL/g for a selected size fraction of Hanford soil (75 to 150 μm) with a soil-to-solution ratio of 0.010 g/mL. Contradictory results and the use of organic complexants in the Swanson experiments indicate that values for specific samples should not be used.

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Gerritse et al. (1982) suggested that the K_d value range for nickel is 1×10^2 to 1×10^3 mL/g. They reported two sandy mineral soil values of 6×10^1 and 3.4×10^2 mL/g and four peat soil values of 3.6×10^2, 6×10^2, 9.9×10^2 and 4.7×10^3 mL/g. Wong et al. (1983) reported K_d values of 604 and 1437 mL/g for a sand and a muck soil, respectively. The recommended values of μ and σ for the K_d distribution for nickel, based on this information and distribution information for the other nuclides, are given in Table 22.

TABLE 22

RECOMMENDED VALUES OF μ AND σ FOR NICKEL

Soil Type	Lognormal Distribution μ	σ
Sand	1.5	1.0
Silt	2.0	1.0
Clay	3.0	1.0
Organic	3.0	1.0

Nickel References

Gerritse, R.G., R. Vriesema, J.W. Dalenberg and H.P. De Roos. 1982. Effect of sewage sludge on trace element mobility in soils. J. Environ. Qual. 11, 359-364.

Swanson, J.L. 1981. Effect of organic complexants on the mobility of low-level waste radionuclides in soils: Status report. Pacific Northwest Laboratory Report, PNL-3927, UC-70.

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4.7 PALLADIUM

No specific information was found on soil adsorption of palladium. We suggest that the K_d values for nickel be used for palladium. Vandergraaf

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(1982) reported a range of 0 to 28 mL/g for the K_d of palladium for granite. He recommended that a K_d value of 11 mL/g be used.

Palladium Reference

Vandergraaf, T.T. 1982. A compilation of sorption coefficients for radionuclides on granites and granitic rocks. Atomic Energy of Canada Limited Technical Record, TR-120*.

4.8 RARE EARTHS - TERBIUM, SAMARIUM AND CERIUM

Terbium, samarium and cerium are fission products, and it is convenient to discuss these three rare-earth elements together because of their chemical similarity. Cerium was the only one of these elements for which data were found. Vandergraaf (1982) reported that the K_d value for cerium ranges from 250 to 5000 mL/g, and recommended a value of 1000 mL/g. Allard et al. (1977) reported a K_d range for cerium of 100 to 10 000 mL/g for clay/ mud and 1000 to 1.6×10^4 mL/g for granite. Baes and Sharp (1981) suggested a mean value of 3.0 for the $\log_{10}K_d$ for cerium, with a corresponding standard deviation of 0.6, and a K_d range of 58 to 6000 mL/g (see Table 1) for all soils. We recommend using the values of Baes and Sharp for all soil types.

Rare Earths References

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Vandergraaf, T.T. 1982. A compilation of sorption coefficients for radionuclides on granites and granitic rocks. Atomic Energy of Canada Limited Technical Record, TR-120*.

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* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

4.9 SELENIUM

Elsokkary (1980) reported selenium adsorption for three soils, which allowed the computation of K_d values of 1.6, 2.2 and 2.5 mL/g on a clay soil and two silty soils, respectively. Frost and Griffin (1977) reported a K_d value of ~ 50 mL/g for $HSeO_3^-$ adsorption on calcium-montmorillonite at a pH of 7. Singh et al. (1981) reported K_d values ranging from 3 to 73 mL/g for selenate adsorption on sandy soils. Since insufficient data are available, we suggest that the values for polonium be used for assessment purposes.

Selenium References

Elsokkary, I.H. 1980. Selenium distribution chemical fractionation and adsorption in some Egyptian alluvial and lacustrine soils. Z. Pflanzenernaehr. Bodenkd., 143, 74-83.

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4.10 SILVER

Little information exists for soil adsorption of silver and no specific soil K_d values were found. Consequently, it is suggested that information for copper be used for silver because of their proximity in the Periodic Table. Gerritse et al. (1982) suggested that the K_d value range for silver is 1×10^3 to 1×10^5 mL/g. They reported K_d values of 1.6×10^2 and 5.6×10^2 mL/g for copper on sandy mineral soil, and K_d values of 4.4×10^3, 1.7×10^4, 2.2×10^4 and 3.3×10^4 mL/g for copper on peaty soil. Values for copper also ranged from 5.5×10^4 to 1.2×10^5 mL/g for 0 to 0.9 meq Ca^{2+}/g dry peat. Wong et al. (1983) reported K_d values for copper of 206 and 197 mL/g for a sand and a muck soil, respectively. The recommended K_d distribution parameter values for silver, based on this information, on the distribution information for the other nuclides and on Table 23, are given in Table 24.

TABLE 23

K_d VALUES FOR SILVER : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	Eh (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	-	-	0	3.5	-	4.5-5.0	-	22	-	-	1.6×10^2	Soil C	Gerritse et al., 1982
	-	-	20	2.5	-	7.5-8.0	-	16	-	-	5.6×10^2	Soil D	Gerritse et al., 1982
	-	-	-	-	-	4.8	-	-	-	0.1 mol/L $CaCl_2$	2.7	Florida 1 - sand	Graham, 1973
	-	-	-	-	-	6.2	-	-	-	0.1 mol/L $CaCl_2$	33.0	Florida 2 - sand + organic matter	Graham, 1973
Silt	-	-	-	-	-	5.0	-	-	-	0.1 mol/L $CaCl_2$	28.0	Missouri 23	Graham, 1973
	-	-	-	-	-	7.4	-	-	-	0.1 mol/L $CaCl_2$	200	Missouri 24	Graham, 1973
	-	-	-	-	-	6.6	-	-	-	0.1 mol/L $CaCl_2$	333	Missouri 38	Graham, 1973
Organic	-	-	-	90	-	4.5	-	-	-	-	4.4×10^3	Soil A	Gerritse et al., 1982
	-	-	-	>90	-	4 to 5	-	-	-	-	2.2×10^4	Peat A	Gerritse et al., 1982
	-	-	-	>90	-	6	-	-	-	-	1.7×10^4	Peat B	Gerritse et al., 1982
	-	-	-	>90	-	6.2	-	-	-	-	3.3×10^4	Soil B	Gerritse et al., 1982

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TABLE 24

RECOMMENDED VALUES OF μ AND σ FOR SILVER

Soil Type	Lognormal Distribution μ	σ	n
Sand	1.726	0.9988	4
Silt	2.090	0.5678	3
Clay	4.0		
Organic	4.184	0.3792	4

Silver References

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Graham, E.R. 1973. Selective distribution and labile pools of micronutrient elements as factors affecting plant uptake. Soil Sci. Soc. Amer. Proc. 37, 70-74.

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4.11 STRONTIUM

The most extensive report on strontium K_d values for Canadian soils is the work of Gillham et al. (1981a). For 15 Canadian soils, the K_d values ranged from 2.5 to 1 x 10^2 mL/g. The study also showed that strontium "in some or possibly many circumstances would migrate at velocities smaller than the groundwater velocity but at velocities which nevertheless could be significant."

The recommended K_d value means, standard deviations, ranges and distribution parameters for strontium by soil type, based on Table 25, are given in Table 26.

TABLE 25

K_d VALUES FOR STRONTIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	100	-	-	0.03	41.3	8.3 ($CaCl_2$)	-	1.4	-	see ref.	2.0×10^1	Soil #4 (WNRE)	Gillham et al., 1981a
	93	5	2	0.05	40.8	7.8 ($CaCl_2$)	-	1.2	-	see ref.	2.5	Soil #6 (Lemington)	Gillham et al., 1981a
	96	4	0	0.51	0	6.3 ($CaCl_2$)	-	1.1	-	see ref.	2.0×10^1	Soil #7 (CRNL)	Gillham et al., 1981a
	52	45	3	0.08	0	5.0 ($CaCl_2$)	-	1.6	-	see ref.	1.0×10^2	Soil #8 (North Bay)	Gillham et al., 1981a
	59	24	17	0.40	0	6.5 ($CaCl_2$)	-	1.9	-	see ref.	2.5×10^1	Soil #10 (WNRE)	Gillham et al., 1981a
	62	31	7	0.38	18.3	7.6 ($CaCl_2$)	-	2.2	-	see ref.	5.0×10^1	Soil #11 (WNRE)	Gillham et al., 1981a
	96	2	2	0.30	11.1	8.0 ($CaCl_2$)	-	0.6	-	see ref.	1.0×10^1	Soil #13 (C.F.B. Borden)	Gillham et al., 1981a
	60	22	18	2.05	7.1	7.8 ($CaCl_2$)	-	21.2	-	see ref.	5.0×10^1	Soil #16 (Alberta)	Gillham et al., 1981a
	87	9	4	0.10	0.07	8.21 ($CaCl_2$)	-	5.0		see ref.	$1.14\times10^2 \pm 9$	Sediment 8 (Solution 1)	Serne et al., 1978
	94	6 (<0.074 mm)		-	-	-	-	-	-	-	2.4×10^1	Composite soil	Schmelz, 1972
		river sand		-	-	7-8	-	-	-	90% NaCl solution	2	River sand	Hamstra & Verburek, 1977
		subsoil sand		-	2	6	-	5	-	4 mol/L Na^+, 0.01 mol/L PO_4^-	1.2×10^1	Hanford subsoil	Rhodes, 1957
		subsoil sand		-	2	8	-	5	-		8.0×10^1	Hanford subsoil	Rhodes, 1957
		sand	20	2.9	-	6	-	19.2	-	3×10^{-3} mol/L $SrCl_2$	10	Sidell sand	Jao & Barber, 1970
		Burbank soil		-	-	-	-	-	-	groundwater	4.8×10^1	Burbank soil	Hajek & Amos, 1968
		Burbank soil		-	-	-	-	-	-	3 mol/L $NaNO_3$	2.1	Burbank soil	Hajek & Amos, 1968
		Burbank soil		-	-	-	-	-	-	0.5 mol/L NaCl	7.3	Burbank soil	Hajek & Amos, 1968
		Burbank soil		-	-	-	-	-	-	3 mol/L NaOAc	2.23	Burbank soil	Hajek & Amos, 1968
	84	13	3	0.16	2.8	-	-	5.1	0.63	0.2 mol/L NaCl see ref.	1.62×10^1	Burbank sand (average profile)	Routson, 1973
	63	32	5	0.21	1.36	-	-	5.3	1.02	0.2 mol/L NaCl see ref.	1.6×10^1	Ephrata sand (average profile)	Routson, 1973
		aquifer sand		-	-	5.5	0.42	0.25-0.9		see ref.*	1.42×10^1	Chalk River (CR) aquifer sand	Patterson & Spoel, 1981
		aquifer sand		-	-	5.5	0.42	0.25-0.9		see ref.*	9.2	Chalk River (RA) aquifer sand	Patterson & Spoel, 1981
		aquifer sand		-	-	5.5	0.42	0.25-0.9		see ref.*	7.8	Chalk River (Q) aquifer sand	Patterson & Spoel, 1981
		aquifer sand		-	-	5.5	0.42	0.25-0.9		see ref.*	1.67×10^1	Chalk River (GB) aquifer sand	Patterson & Spoel, 1981
		aquifer sand		-	-	5.5	0.42	0.25-0.9		see ref.*	1.13×10^1	Chalk River (K) aquifer sand	Patterson & Spoel, 1981
		aquifer sand		-	-	5.5	0.42	0.25-0.9		see ref.* see ref.	6.0	Chalk River (HA) aquifer sand	Patterson & Spoel, 1981

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TABLE 25 (Concluded)

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (ml/g)	Soil Location or Description	Reference
Silt	36	35	29	0.43	33.6	8.1 ($CaCl_2$)	-	8.4	-	see ref.	2.0×10^1	Soil #1 (WNRE)	Gillham et al., 1981a
	35	36	29	0.41	33.8	8.1 ($CaCl_2$)	-	8.3	-	see ref.	2.0×10^1	Soil #2 (WNRE)	Gillham et al., 1981a
	34	35	31	0.40	34.1	8.1 ($CaCl_2$)	-	8.6	-	see ref.	2.0×10^1	Soil #3 (WNRE)	Gillham et al., 1981a
	28	41	31	1.27	21.1	7.7 ($CaCl_2$)	-	5.9	-	see ref.	1.0×10^1	Soil #5 (Lemington)	Gillham et al., 1981a
	12	55	33	0.35	0	6.7 ($CaCl_2$)	-	10.2	-	see ref.	1.0×10^2	Soil #9 (North Bay)	Gillham et al., 1981a
	34	34	32	0.85	5.1	7.7 ($CaCl_2$)	-	32.7	-	see ref.	8.0	Soil #14 (Alberta)	Gillham et al., 1981a
	45	44	11	0.14	1.4	8.8) ($CaCl_2$)	-	12.0	-	see ref.	$1.12\times10^2 \pm 1$	Sediment A (Solution 1)	Serne et al., 1978
	medium loam (0.01 mm)		31.6	2.94	-	6.6	-	10.6	-	-	$3.0\times10^2 \pm 60$	low ash podzolic	Alekseekhin, 1965
	medium loam (0.01 mm)		41.6	1.28	-	8.4	-	12.2	-	-	$1.7\times10^2 \pm 30$	Serozem	Alekseekhin, 1965
	31	69	0	-	-	-	-	2.6	-	-	1.4×10^1	alluvial soil (Cadarache)	Rançon, 1972
	38	62	0	-	-	-	-	2.7	-	-	2.3×10^1	alluvial soil (Cadarache)	Rançon, 1972
	18	66	16	-	-	-	-	6.3	-	-	1.8×10^1	Vindobonian sed. (Cadarache)	Rançon, 1972
	40	45	15	-	-	-	-	1.8	-	-	1.6×10^1	Vindobonian sed. (Cadarache)	Rançon, 1972
	34	52	14	-	-	-	-	4.9	-	-	1.6×10^1	Vindobonian sed. (Cadarache)	Rançon, 1972
	45	47	8	-	-	-	-	1.5	-	-	1.4×10^1	Vindobonian sed. (Cadarache)	Rançon, 1972
	7	92	1	-	-	-	-	4.2	-	-	2.2×10^1	sandy-clay sed. (Durance R.)	Rançon, 1972
	18	71	11	-	-	-	-	3.5	-	-	1.6×10^1	sandy-clay sed. (Durance R.)	Rançon, 1972
	3	96	1	-	-	-	-	5.2	-	-	1.6×10^1	sandy-clay sed. (Durance R.)	Rançon, 1972
	silt		29.1	7.1	-	6	-	39.4	-	3×10^{-3} mol/L $SrCl_2$*	5.0×10^1	Brockston silt	Jun & Barber, 1970
	44	50	6	0.23	3.8	-	-	11.0	1.21	0.2 mol/L NaCl	2.47×10^1	Ritzville silt (avg. profile)	Routson, 1973
Clay	31	34	35	0.81	5.2	7.8 ($CaCl_2$)	-	31.5	-	see ref.	8	Soil #15 (Alberta)	Gillham et al., 1981a
	heavy loam 53.4% (0.01 mm)			2.04	-	6.6	-	26.1	-	-	$5.7\times10^2 \pm 80$	Chestnut	Alekseekhin, 1965
	heavy loam 46.6% (0.01 mm)			-	-	6.7	-	30.4	-	-	$1.15\times10^3 \pm 140$	Leached Chernozem	Alekseekhin, 1965
	heavy loam 67.0% (0.01 mm)			4.87	-	8.0	-	32.9	-	-	$4.3\times10^2 \pm 30$	Southern Chernozem	Alekseekhin, 1965
	clayey 60.7% (0.01 mm)			6.86	-	6.8	-	32.2	-	-	$4.9\times10^2 \pm 50$	Thick Chernozem	Alekseekhin, 1965
	100% (<0.02 mm)			-	-	-	-	35	-	-	4.7×10^1	very fine suspended sediment (Durance River)	Rançon, 1972
Organic	muck			49.8	-	7.0	-	70.0	-	3×10^{-3} mol/L $SrCl_2$	1.5×10^2	Muck	Jun & Barber, 1970

* Data available for competing cations sodium, potassium, magnesium, calcium, barium and hydrogen.

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TABLE 26

K_d FOR STRONTIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	26.02	30.21	26	2.0 to 114	1.162	0.4964
Silt	49.49	72.44	20	8.0 to 300	1.436	0.4254
Clay	449.2	415.7	6	8.0 to 1150	2.286	0.8239

Baes and Sharp (1981) suggested a mean value of 1.4 for the $\log_{10}K_d$ for strontium, with a corresponding standard deviation of 0.9 (see Table 1). K_d values for strontium determined for various pure clay minerals ranged from 0.2 to 9.0 mL/g (Wahlberg et al., 1965). K_d values determined for various minerals (including clay minerals) ranged from 0 mL/g for quartz to 2.1×10^3 mL/g for alumina (both at pH = 7.5) in a natural water solution and from 1 mL/g for quartz to 1.44×10^3 mL/g for alumina (both at pH = 7.0) in a 0.1 mol/L sodium nitrate solution (Tamura, 1972). Palmer et al. (1981) also reported extensive results for strontium sorption on pure clay, clay/silica and alumina/clay for various solution compositions and pH. Allard et al.(1977) reported a K_d range of 20 to 63 mL/g for clay/mud and a range of 3 to 16 mL/g for granite. Vandergraaf (1982) recommended a K_d range for strontium of 0.6 to 600 mL/g for granite.

Strontium References

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4.12 TECHNETIUM

The environmental behaviour of technetium has recently been reviewed (Turcotte, 1982), as has the chemistry of technetium (Paquette et al., 1980). Technetium migration is generally retarded under reducing conditions (i.e., in geological formations), where it is less soluble. Technetium, however, moves with the groundwater in aerated soils of low organic carbon content. Johnston and Gillham (1980) indicated that

(1) because the pertechnetate ion, TcO_4^-, is the most stable species of technetium in aqueous solutions, TcO_4^- will not be subject to ion exchange; hence, technetium will show little adsorption to soil;

(2) in soils with appreciable organic matter, Tc^{7+} may be reduced to Tc^{4+} and adsorbed.

The recommended K_d value means, standard deviations, ranges and distribution parameters for technetium by soil type, based on Table 27, are given in Table 28.

TABLE 28

K_d FOR TECHNETIUM

Soil Type	K_d (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	29.39	100.1	15	1.0×10^{-3} to 388	-1.148	1.565
Silt	1.426	3.869	8	1.0×10^{-3} to 11	-1.332	1.290
Organic	118.4	192.0	3	0.24 to 340	1.029	1.581

* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

TABLE 27

K_d VALUES FOR TECHNETIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	87	9	4	0.10	0.07	8.23	-	5.0	-	see ref.	0.07 ± 0.32	Sediment B (solution 1)	Serne et al., 1978
	87	9	4	0.10	0.07	8.2	oxidized	5.0	-	Na_3 Citrate Cit/Tc Molar Ratio = 0	0.32 (5×10^{-9} mol/L Tc)	Hanford soil	Franz et al., 1982
	87	9	4	0.10	0.07	10.1	reduced	5.0	-	Na_3 Citrate Cit/Tc Molar Ratio = 0	52 (5×10^{-9} mol/L Tc)	Hanford soil	Franz et al., 1982
	87	9	4	0.10	0.07	10.1	reduced	5.0	-	Na_3 Citrate Cit/Tc Molar Ratio = 1:1	388 (5×10^{-9} mol/L Tc)	Hanford soil	Franz et al., 1982
	53	35	12	3.0	-	5.4	-	15.2	1.1	-	0.155	Aquic Fragiudept (A_1-A_2)	Balogh & Grigal, 1980
	95	3	2	0.2	-	5.7	-	3.2	0.6	-	0.051	Alfic Udipsamment (B)	Balogh & Grigal, 1980
	69	19	22	2.8	-	6.0	-	20.4	0.7	-	0.078	Aquic Hapludoll (Ap)	Balogh & Grigal, 1980
	77	11	12	1.8	-	8.3	-	11.7	0.2	-	0.000	Aquic Haploboroll (Ap)	Balogh & Grigal, 1980
	59	4	37	-	<0.2	5.1	-	2.5	-	0.002 mol/L $NaHCO_3$	+0.019 ± 0.06	South Carolina subsoil	Routson et al., 1977
	59	4	37	-	<0.2	5.1	-	2.5	-	0.008 mol/L $NaHCO_3$	-0.052 ± 0.01*	South Carolina subsoil	Routson et al., 1977
	59	4	37	-	<0.2	5.1	-	2.5	-	0.020 mol/L $NaHCO_3$	-0.031 ± 0.01*	South Carolina subsoil	Routson et al., 1977
	59	4	37	-	<0.2	5.1	-	2.5	-	0.200 mol/L $NaHCO_3$	+0.010 ± 0.04	South Carolina subsoil	Routson et al., 1977
	45	44	11	0.14	-	8.8	-	-	-	-	0.04	unsaturated column	Gee & Campbell, 1980
	87	9	4	0.10	-	8.2	-	-	-	-	0.03	unsaturated column	Gee & Campbell, 1980
	coarse sand			-	5.2	8.4 (water)	-	1.2	1382 (μg/g)	see ref.	0.094	sand	Sheppard et al., 1983
Silt	45	44	11	0.14	1.4	8.93	-	12.0	-	see ref.	-2.77 ± 0.23*	Sediment A (solution 1)	Serne et al., 1978
	16	52	32	8.3	-	7.6	-	27.0	0.4	-	0.028	Cumulic Haplaquoll	Balogh & Grigal, 1980
	3	80	17	2.3	-	5.5	-	11.3	1.4	-	0.068	Typic Eutroboralf (A_2)	Balogh & Grigal, 1980
	9	64	27	5.4	-	7.7	-	16.9	0.1	-	0.118	Aeric Calciaquoll (A_1)	Balogh & Grigal, 1980
	9	57	34	11.0	-	7.7	-	36.4	0.5	-	0.118	Cumulic Haplaquoll (Ap)	Balogh & Grigal, 1980
	37	33	30	7.3	-	7.8	-	43.5	0.3	-	0.076	Typic Haplaquoll (Al)	Balogh & Grigal, 1980
	27	43	30	2.4	-	5.9	-	19.3	0.9	-	0.011	Aquic Hapludoll (Al)	Balogh & Grigal, 1980
	8	60	32	3.1	-	6.6	-	26.8	1.2	-	0.000	Udic Haploboroll (Ap)	Balogh & Grigal, 1980
Organic	charcoal			-	-	6.1 - 6.6	-	-	-	NaCl brine see ref.	340	activated "Nuchar"	Novak, 1981
	fen		3.0	23.3	-	4.5	-	46	0.55	0.01 or 0.05 mol/L $CaCl_2$	0.24	fen soil, Zuoen (Netherlands)	Mouny & Nyttenaere, 1981
	sphagnum peat			92	4.9	3.8 (water)	-	64.7	1000 (μg/g)	see ref.	15.0	sphagnum peat	Sheppard et al., 1983

* Negative values will be considered as a zero value.

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Baes and Sharp (1981) suggested a mean value of -1.5 for the $\log_{10}K_d$ for technetium, with a corresponding standard deviation of 0.5 (see Table 1). Mousny and Myttenaere (1981) investigated the effect of temperature on the soil adsorption of technetium and reported that, for seven soils investigated (including a podzol soil and a fen soil), K_d ranged from 0.007 to 0.234 mL/g. Baes and Sharp (1981) suggested the K_d range is 0.003 to 0.28 mL/g (see Table 1). Wildung et al. (1974) selected 22 surface soils with the following range of properties:

CEC (meq/100 g)	pH	Carbonate	Organic Carbon	Sand	Silt	Clay
		%				
5.5-90.0	3.6-8.9	0-6.5	0.23-28.8	14.1-73.1	17.6-58.0	3.8-46.6

and suggested that the K_d for technetium ranges from 0.007 to 2.8 mL/g. They also suggested a prediction equation of the form

$$K_d = 0.08\ X_3 - 0.09\ X_4 \qquad (7)$$

where X_3 is the organic carbon content,
X_4 is the pH.

Vandergraaf (1982) recommended a K_d range for technetium of 0 to 80 mL/g for granites.

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4.13 TIN

Gerritse et al. (1982) suggested that the K_d value for tin ranges from 1×10^2 to 1×10^6 mL/g. We recommend that the K_d distribution information for lead be used for tin.

* Unrestricted, unpublished report, available from SDDO, Atomic Energy of Canada Limited Research Company, Chalk River, Ontario K0J 1J0.

[redacted]

Tin Reference

Gerritse, R.G., R. Vriesema, J.W. Dalenberg and H.P. De Roos. 1982. Effect of sewage sludge on trace element mobility in soils. J. Environ. Qual. 11, 359-364.

4.14 ZIRCONIUM AND NIOBIUM

Rhodes (1957) reported K_d values ranging from 90 mL/g (pH = 6.0) to > 1980 mL/g (pH = 2.7, 3.5, 4.4, 8.4 and 9.3) for zirconium-niobium adsorption on Hanford subsoil. From this information, in the pH range of most interest for surface soil (pH = 5 to 8, a sandy soil), zirconium has an average K_d of 164 mL/g. This appears to agree well with the K_d for niobium of 210 mL/g recommended for granite (Vandergraaf, 1982). Vandergraaf also recommended a K_d range for zirconium of 1000 to 6000 mL/g for granite. Allard et al. (1977) reported a K_d range for zirconium of 50 to 1000 mL/g for clay/mud and 1250 to 6300 mL/g for granite.

Based on this information, the recommended mean of the lognormal distribution for zirconium and niobium is 2.5 with a standard deviation of 1.0. Because information is insufficient to break it down by soil type, one value is recommended for all soil types.

Zirconium and Niobium References

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5. DISTRIBUTION COEFFICIENTS FOR OTHER NUCLIDES

5.1 ANTIMONY

No specific soil K_d information was found for antimony. The K_d values for lead are recommended for antimony, because of its proximity to lead in the Periodic Table.

5.2 BORON

Little information was found on boron adsorption on soils; however, there is some indication that adsorption is influenced by soil texture and the presence of soluble salts and exchangeable cations (Gupta, 1980). Boron adsorption information reported by Keren and O'Connor (1982) for montmorillonite and illite indicated that the K_d value for boron for these pure clays could be as high as 20 mL/g. That work suggested that the K_d value for soils may be in the range 0 to 10 mL/g. We recommend a value of 1 mL/g for assessment purposes; the lognormal distribution parameter values cannot be given.

Boron References

Gupta, I.C. 1980. Equilibrium adsorption of boron as affected by texture, salinity and alkalinity of soil. Ann. Arid Zone 19, 243-248.

Keren, R. and G.A. O'Connor. 1982. Effect of exchangeable ions and ionic strength on boron adsorption by montmorillonite and illite. Clays Clay Miner. 30, 341-346.

5.3 CADMIUM

Most of the work carried out with cadmium has been in response to environmental concerns about the application of sewage sludge to agricultural land. The sorption of cadmium on soils and sediments has been studied by Poelstra et al. (1979), Rendell et al. (1980), Hendrickson and Corey (1981), and Gerritse et al. (1982). The recommended K_d value means, standard deviations, ranges and distribution parameters for cadmium, based on Table [redacted] are given in Table 30.

TABLE 29

K_d VALUES FOR CADMIUM : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH[a] Saturated Paste	E_h (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	sandy soil			-	-	6.5	-	31.6	-	-	66.7[a]	Sandy soil (Braunschweig) 0-20 cm	Poelstra et al., 1979
	sandy soil			-	-	6.5	-	31.6	-	-	47.6[a]	Sandy soil (Braunschweig) 30-40 cm	Poelstra et al., 1979
	sandy soil		0	3.5	-	4.5 - 5.0	-	22	-	$[Ca^{2+}]$ = 0-0.015 mol/L	2.62×10^2	Soil C	Garritse et al., 1982
	sandy soil		20	2.5	-	7.5 - 8.0	-	16	-	$[Ca^{2+}]$ = 0-0.015 mol/L	5.0×10^2	Soil D	Garritse et al., 1982
	fine sand			1.4	-	8.2	-	11	-	-	72	Hollandale fine sand	Wong et al., 1983
Silt	<74 μm			0.72	-	8.4	-	60	1.07	-	76	Imperial (California)	Garcia-Miragaya, 1980
	<74 μm			1.8	-	6.0	-	25	1.07	-	9.8	Oliverhain (California)	Garcia-Miragaya, 1980
	<74 μm			1.5	-	5.8	-	24	0.29	-	16	Hoover (California)	Garcia-Miragaya, 1980
Clay	clay soil		23	6	trace	7.4	-	-	-	-	625[a]	(Valburg) 0-30 cm	Poelstra et al., 1979
Organic	organic			16.3	-	5.2	-	33.8	-	-	23	organic	Garcia-Miragaya, 1980
	peat soil			95	-	5.1 (H_2O)	-	-	-	-	370[a]	(Schoonebeek)	Poelstra et al., 1979
				90		4.5				$[Ca^{2+}]$ = 0-0.015 mol/L	1.44×10^3	Soil A	Garritse et al., 1982
				>90		4 to 5				$[Ca^{2+}]$ = 0-0.015 mol/L	9.0×10^3	Peat A	Garritse et al., 1982
				>90		6.2				$[Ca^{2+}]$ = 0-0.015 mol/L	5.76×10^3	Soil B	Garritse et al., 1982
	sphagnum peat			-	-	4 to 5	-	-	-	-	1.7×10^4	Peat	Wolf et al., 1977
	sphagnum peat			-	-	4 to 5				0.025 mm Ca^{2+}/kd. sol.	3×10^1	Peat	Wolf et al., 1977
	plantation muck			47	-	7.2	-	34	-	-	341	average of 3 layers	Wong et al., 1983

a 1 week equilibration

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TABLE 30

K_d FOR CADMIUM

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	189.7	194.1	5	47.6 to 500	2.095	0.4397
Silt	33.93	36.56	3	9.8 to 76	1.359	0.4645
Organic	4246	6110	8	23 to 1.7×10^4	2.880	1.090

Hendrickson and Corey (1981) reported K_d data from several authors, and their plot indicated that the K_d range is 0 to 6 mL/g and is significantly dependent on both the cadmium and calcium contents of the soil. Navrot et al. (1978) reported K_d values for cadmium for five Israeli soils ranging from 1×10^3 to 1×10^4 mL/g, and the K_d value was correlated to specific soil surface area.

Cadmium References

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Wolf, A., K. Bunzl, F. Dietl and W.F. Schmidt. 1977. Effect of calcium ions on the absorption of lead (2+), copper (2+), cadmium (2+) and zinc (2+) by humic substances. Chemosphere 6, 207-213.

Wong, K.V., S. Sengupta, D. Dasgupta, E.L. Daly, Jr., N. Nemerow and H.P. Gerrish. 1983. Heavy metal migration in soil-leachate systems. Biocycle 24, 30-33.

5.4 TELLURIUM

Allard et al. (1977) suggested that the K_d value for tellurium for clay/mud and granite is 1 mL/g. This work suggests that the K_d range may be 0 to 1 mL/g. We recommend a K_d value of 0 mL/g for assessment purposes; no distribution parameter values can be given.

Tellurium Reference

Allard, B., H. Kipatsi and J. Rydberg. 1977. Adsorption of long-lived radionuclides in clay and bedrock. Part 1. Determination of distribution coefficients. KBS Technical Report 55.

5.5 ZINC

Gerritse et al. (1982) suggested that the K_d values for both zinc and cadmium range from 1×10^3 to 1×10^4 mL/g, and their data for sandy and organic soils show that the two elements exhibit very similar sorption behaviour. The recommended K_d value means, standard deviations, ranges and distribution parameters for zinc, based on Table 31, are given in Table 32.

TABLE 32

K_d FOR ZINC

Soil Type	$\bar{K}_d$ (mL/g)	S.D.	n	K_d Range (mL/g)	Lognormal Distribution μ	σ
Sand	622.0	911.6	5	0.1 to 2120	1.762	1.694
Silt	51.8	68.17	2	3.6 to 100	1.278	1.021
Organic	4092	4909	6	70 to 1.3×10^4	3.185	0.83

TABLE 31

K_d VALUES FOR ZINC : LITERATURE SURVEY SUMMARY

Soil Type	% Sand	% Silt	% Clay	% Organic	% $CaCO_3$	pH Saturated Paste	E (V)	CEC (meq/100 g)	% Free Iron Oxides	Competing Cation	K_d (mL/g)	Soil Location or Description	Reference
Sand	sandy soil			3.5	-	4.5 - 5.0	-	22	-	$[Ca^{+2}]$ = 0-0.015 mol/L	7.0×10^1	Soil C	Gerritse et al., 1982
	sandy soil			2.5	-	7.5 - 8.0	-	16	-	$[Ca^{2+}]$ = 0-0.015 mol/L	2.12×10^3	Soil D	Gerritse et al., 1982
	-			-	-	4.8	-	-	-	0.1 mol/L $CaCl_2$	0.1	Florida 1	Graham, 1973
	-			-	-	6.2	-	-	-	0.1 mol/L $CaCl_2$	50	Florida 2	Graham, 1973
	fine sand			1.4	-	8.2	-	11	-	-	870	Hallandale fine sand	Wong et al., 1973
Silt	silt loam			-	-	5.0	-	-	-	0.1 mol/L $CaCl_2$	3.6	Missouri 23	Graham, 1973
	silt loam			-	-	7.4	-	-	-	0.1 mol/L $CaCl_2$	100	Missouri 24	Graham, 1973
Organic	organic			90	-	4.5	-	-	-	$[Ca^{2+}]$ = 0-0.015 mol/L	1.89×10^3	Soil A	Gerritse et al., 1982
	organic			>90	-	4 - 5	-	-	-	$[Ca^{2+}]$ = 0-0.015 mol/L	6.3×10^3	Peat A	Gerritse et al., 1982
	organic			>90	-	6.2	-	-	-	$[Ca^{2+}]$ = 0-0.015 mol/L	2.88×10^3	Soil B	Gerritse et al., 1982
	sphagnum peat			-	-	4 - 5	-	-	-	-	1.3×10^4	Peat	Wolf et al., 1977
	sphagnum peat			-	-	4 - 5	-	-	-	0.025 meq Ca^{2+}/mL sol.	7.0×10^1	Peat	Wolf et al., 1977
	muck			47	-	7.2	-	34	-	-	412	Plantation muck (average of 3 layers)	Wong et al., 1983

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Zinc References

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5. CONCLUSIONS

The paucity of K_d values for organic soil is the most striking observation from our review of the literature. Plutonium, lead, technetium, cadmium and zinc were the only nuclides for which more than two K_d values have been determined for an organic soil (see Table 33). The next most important observation is that very little work has been done with mineral soils for some of the more mobile nuclides with K_d values up to 100 mL/g, such as uranium, technetium, molybdenum, iodine, selenium, carbon, boron, and tellurium. There may be good reasons why more K_d work is not warranted for these nuclides in the Canadian waste management program, such as the formation of precipitates or reduction to an immobile species in the vault or geosphere. Our major recommendation is that effort be directed towards the chemistry (including parameter determination, i.e., K_d determinations) of organic soils, and in particular the reactions of uranium, technetium, iodine, selenium and carbon with organic soils. In spite of the limited data base, it is possible to select reasonable K_d distribution parameter values for most nuclides for long-term waste management assessment purposes.

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TABLE 33

AVAILABILITY OF K_d DATA FOR EACH NUCLIDE BY SOIL TYPE

Nuclide	Soil Type			
	Sand	Silt	Clay	Organic
Actinium	X	X	X	X
Americium				X
Antimony	X	X	X	X
Bismuth			X	X
Boron	X	X	X	X
Cadmium			X	
Calcium	X	X	X	X
Carbon	X	X	X	X
Cesium				X
Iodine				X
Lead		X	X	
Molybdenum	X	X	X	X
Neptunium				X
Nickel	X	X	X	X
Palladium	X	X	X	X
Plutonium				
Polonium			X	X
Protactinium	X	X	X	X
Radium				X
Rare Earths	X	X	X	X
Selenium	X	X	X	X
Silver	X	X	X	X
Strontium				X
Technetium			X	
Tellurium	X	X	X	X
Thorium	X	X	X	X
Tin	X	X	X	X
Uranium	X	X		X
Zinc		X	X	
Zirconium & Niobium	X	X	X	X

X denotes 2 or fewer K_d values.

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