

● Note

## DEFAULT SOIL SOLID/LIQUID PARTITION COEFFICIENTS, $K_d$ s, FOR FOUR MAJOR SOIL TYPES: A COMPENDIUM

Marsha I. Sheppard and D. H. Thibault

Environmental Research Branch, Whiteshell Nuclear Research Establishment,  
Atomic Energy of Canada Limited, Pinawa, Manitoba, Canada, R0E 1L0

### INTRODUCTION

MATHEMATICAL models are used in many countries to predict the dissolution of radioactive waste in underground vaults and the subsequent transport of radionuclides through the geosphere and biosphere. It is important to understand the interactions between radionuclides and various media along the path to the biosphere, whether disposal is in deep or shallow rock caverns or in shallow overburden facilities. The preliminary or screening assessments of the safety of these disposal concepts is usually carried out using generic data; in other words, a specific site has not been chosen. This precludes the use of parameter values measured *in situ* and relegates assessment modellers to using generic data that may not approximate real conditions (Till and Meyer 1983). Although such generic values are not necessarily useful in predicting absolute impact, standardization of generic data is useful.

Our research, using a soil transport model for the assessment of our disposal concept, shows that only four parameters are essential to accurately predict soil concentrations from either contaminated ground water or irrigation water (Sheppard and Bera 1984). These four parameters, in order of decreasing importance, are: soil retention, annual precipitation, soil texture, and depth to the water table.

The soil retention parameter in most assessment models is the soil solid/liquid partition coefficient,  $K_d$ , (Till and Meyer 1983; Isherwood 1981). This empirical parameter lumps all operating retention mechanisms into one value (Ames and Rai 1978). The  $K_d$  model assumes that the liquid and solid phases are at equilibrium and that there is a linear relationship between solute concentration in the solid ( $C_s$ ) and liquid ( $C_l$ ) phases (Sheppard 1985; Sheppard and Evenden 1988), as expressed by the equation:

$$C_s = K_d C_l \quad (1)$$

In the absence of detailed, site-specific information about the soil (e.g., its pH, porewater composition, organic matter content, etc.), and the specific reaction kinetics of each element of concern),  $K_d$ s are used as default retention parameters. However, the  $K_d$  model should be used with caution in simple systems and for elements with a well-known chemistry.

We have updated an earlier compilation of soil  $K_d$  values (Sheppard et al. 1984) prepared for assessing the concept of disposing Canadian spent fuel deep in Precambrian Shield plutonic rock. The objective of this note is to publish this large volume of data, which include data from previous compendiums (Ames and Rai 1978; Isherwood 1981), in a compressed and usable manner. This compendium includes data from the earlier report and from the recent literature and internal reports from many nuclear and government laboratories. In addition, we briefly discuss the factors that affect  $K_d$  values, such as soil texture, pH, competing cation effects, soil porewater concentration, and soil organic matter content. This note does not review the chemistry of the individual elements; for this, we recommend the original references documented by Thibault et al. (1990).

### METHODS

This compendium includes all important elements (Table 1) present in Canada's nuclear fuel waste vault inventory, with the exception of noble gases and H (Meht and Goodwin 1990). The data have been combined for all isotopes of each element; we have included both radiologically and chemically toxic elements. The data were extracted from the literature and compiled in a computerized spreadsheet.

The mineral soils were categorized by texture into sand, clay, and loam. The soils that contained  $\geq 70\%$  sand-sized particles were classified as sand soils, and those containing  $\geq 35\%$  clay-sized particles were classified as clay soils. Loam soils had an even distribution of sand, clay-, and silt-sized particles or consisted of up to 30% silt-sized particles. Organic soils contained  $> 30\%$  organic

Table 1. Summary of GM  $K_d$  values ( $L\ kg^{-1}$ ) for each element by soil type.<sup>a,b</sup>

Element	Sand	Loam	Clay	Organic
Ac	450*	1 500	2 400	5 400
Ag	90*	120	180	15 000
Al	1 900	9 600	8 400	112 000
<del>As</del>	250	800	1 300	3 000
Bi	100	450	400	1 500
Br	15	50	75	180
C	5	20	1	70
Ca	5	30	50	90
Cl	80	40	560	800
Cd	500	8 100	20 000	3 300
Ce	4 000	18 000	6 000	6 000
Co	60	1 300	550	1 000
Cr	70	30	1 500	270
Cs	280	4 600	1 900	270
Fe	230	800	165	600
Hf	450	1 500	2 400	5 400
Ho	250	800	1 300	3 000
I	1	5	1	25
K	15	55	75	200
Mn	50	750	180	150
Mo	10	125	90	25
Nb	160	550	900	2 000
Ni	400	300	650	1 100
Np	5	25	55	1 200
P	5	25	35	90
Pa	550	1 800	2 700	6 600
Pb	270	16 000	550	22 000
Pd	55	180	270	670
Po	150	400	3 000	7 300
Pu	550	1 200	5 100	1 900
Ra	500	36 000	9 100	2 400
Rb	55	180	270	670
Ra	10	40	60	150
Ru	55	1 000	800	66 000
Sb	45	150	250	350
Se	150	500	740	1 800
Si	35	110	180	400
Sm	245	800	1 300	3 000
Sn	130	450	670	1 600
Sr	15	20	110	150
Ta	220	900	1 200	3 300
Tc	0.1	0.1	1	1
Te	125	500	720	1 900
Th	3 200	3 300	5 800	89 000
U	35	15	1 600	410
V	170	750	1 000	2 600
Zn	200	1 300	2 400	1 600
Zr	600	2 200	3 300	7 300

\* Values with regular numbering are default values predicted using CRs.

<sup>b</sup> Values with *italic bold* numbering come from the literature.

latter and were either classic peat or muck soils, or the latter horizon of a mineral soil.

If a time series of  $K_d$  values was reported, we used only the  $K_d$  values for the longest time since these values would most closely approximate equilibrium conditions. Only one value was entered for each soil reported in the literature. For example, where  $K_d$  values were reported for the same soil for a range of soil:solution ratios, competing cations, contact solution concentrations, or pH values, the results were ln-transformed and averaged to provide a single geometric mean (GM) value. The transformation was justified because soil  $K_d$  values are lognormally distributed (Sheppard et al. 1984; Sheppard and Evenden 1989). The single values for each soil were also ln-transformed, and GMs and geometric standard deviations

(GSDs) were determined for each element by soil texture for the mineral soils and also for organic soils.

If no data existed in the literature for a given element, the soil-to-plant concentration ratio (CR) was used as an indicator of the element's mobility and to predict a default  $K_d$  value (Baes et al. 1984; Sheppard 1985). The CR values used were taken from Baes et al. (1984). This technique is successful because of the strong negative correlation between CR and  $K_d$  (Sheppard and Sheppard 1989). The model used had the following form:

$$\ln K_d = a + \text{STEX} + b (\ln \text{CR}). \quad (2)$$

where  $a$ ,  $b$ , and STEX are constants. The values for the coefficients were  $a = 4.62$ , and  $b = -0.5$ . The following

are STEX values for each soil classification: sand, STEX = -2.51; loam, STEX = -1.26; clay, STEX = -0.84; and organic, STEX = 0. The full regression analysis, all original tables, and the references are given in Thibault et al. (1990) and can be obtained on diskette from the authors.

## RESULTS AND DISCUSSION

### Factors affecting solute transport

The factors affecting element behavior in soil relate primarily to speciation and reactivity (Bond and Smiles 1988). Nonreactive elements do not interact with the solid phase, and the  $K_d$  value is zero; this is the case with H. If the element is reactive, it may behave either as an anion or a cation. The relationship between the porewater and adsorbed or exchangeable phases can be linear or nonlinear. If this relationship is nonlinear, the distribution between the phases is best described by the Freundlich isotherm (Sheppard et al. 1987; Sheppard and Thibault 1990). If the isotherm is or is assumed to be linear, then the Freundlich and  $K_d$  models are equivalent. Other adsorption models are discussed in Rai and Zachara (1984). At usual environmental concentrations, the differences between linear and nonlinear isotherms are often very small. The simplicity of the  $K_d$  model and the lack of data to accurately define the coefficient for the Freundlich isotherm favors the  $K_d$  model for preliminary or screening environmental assessments.

### Factors affecting $K_d$

Although much of the ancillary data needed to assess overall trends in  $K_d$  values are not reported, soil texture is generally available. The data for all elements were analyzed using analysis of variance (SAS 1985). This showed

that in all elements, the texture of mineral soils did not affect the  $K_d$  value. Certain elements, however, such as Ce, Pu, Sr, and Zn, showed an increase in  $K_d$  values across the whole texture range (Table 1). Other elements, such as Am, Cm, Co, Cs, Mn, Np, Ra, and Ru, had the lowest  $K_d$  values in sand soils, but showed no consistent trend for loam and clay soils. Grouping the elements by  $K_d$  and soil type emphasizes the variable dependence of the elements on texture (Table 2). We have highlighted Tc, I, and U in Table 2 to illustrate some of the trends in the data.

Including organic soils in this compilation adds another dimension of variation. Organic soils have an inherently complicated chemistry, including a broad range of redox conditions, and have the potential to form complexes with contaminant elements entering the system. We have not included  $K_d$  values determined under anoxic or anaerobic conditions because the greatest need is for information about predominantly unsaturated soils. It is these unsaturated soils that are used for upland crop production, urban development, etc., and form some of the direct exposure pathways to humans. The data for anoxic conditions can be found in the detailed report of Thibault et al. (1990). Higher  $K_d$  values in organic soils were found for Ag, Am, Cd, I, Ni, Np, Ru, Sr, and Th, compared to mineral soils.  $K_d$  values were similar between organic and mineral soils for Ce, Cm, Co, Cs, Fe, Mn, Mo, Pu, Tc, U, and Zn. Cerium and Pu are more affected by clay content and pH than by organic matter content (Coughtry et al. 1985). Cobalt is known to have complex reactions in low-organic matter sand aquifers (Killey et al. 1984). For Cs, Gillham et al. (1980) found that stable Cs concentrations affected the  $K_d$  value more than any other soil parameter. Enhanced Tc sorption with increasing organic matter content has been reported previously (Sheppard et al. 1990), and yet our data do not indicate that

Table 2. Grouping of elements by  $K_d$  values and soil type using GMs from this study. Default values are not shown here. Technetium, I, and U are highlighted to show the general trend with soil type.

$K_d$ values exp ( $\mu$ )	This Study			
	Sand	Loam	Clay	Organic
<1	8 (Tc)	(Tc)		
1-10	1 (I, Mo, Np)	1	1 (I, Tc)	(Tc)
10-100	Ag, Cd, Co, Cr, Mn, Ru, Sr (U)	Cd, Cr, Np, Sr (U)	Mo, Np	1 (I, Mo)
100-1 000	Ce, Cs, Fe, Ni, Pb, Po, Pu, Ra, Zn	Ag, Fe, Mn, Po	Ag, Cd, Co, Fe, Mn, Ni, Sr	Cd, Co, Cr, Cs, Sr (U)
1 000-10 000	Am, Cm, Th	Am, Ce, Co, Cs, Pu, Zn	Am, Cs, Pu, Ra, Th (U) Zn	Ni, Np, Pu, Zn
>10 000		Cm, Pb, Ra	Ce	Ag, Am, Pb, Ru, Th

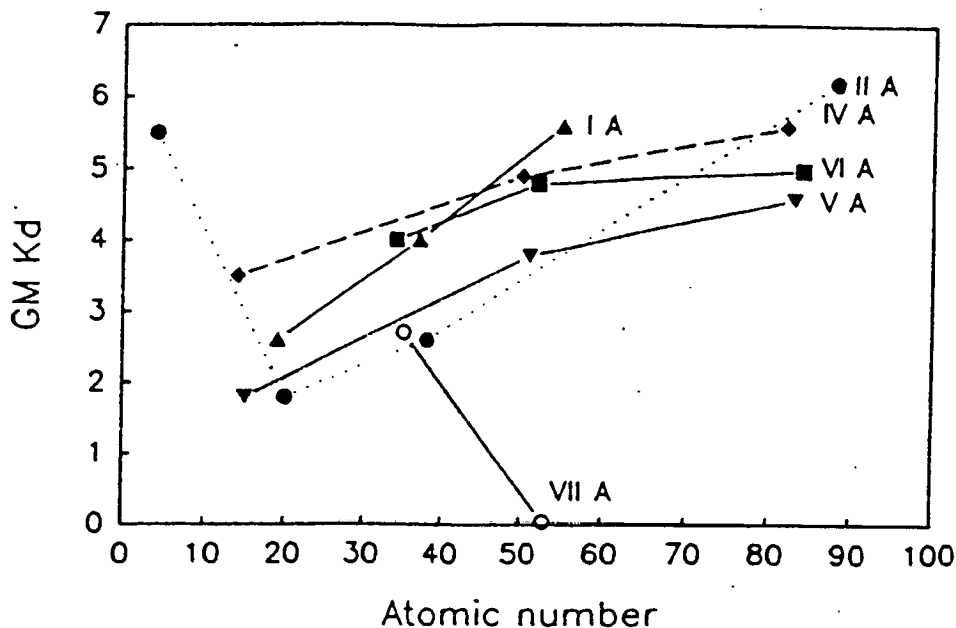


Fig. 1. Geometric mean  $\ln K_d$  values for sandy soils vs. atomic number for all the elements, by group.

organic matter content has any more effect on Tc than clay content (Table 1).

The soil:solution ratio can have a pronounced effect on the  $K_d$  value (Sheppard et al. 1983). The soil:solution ratio in batch sorption studies is usually 1 g of soil to 10 mL of solution. In the field, soils are unsaturated most of the time, with less than 50% of their pore space containing water. Determination of  $K_d$  values in batch experiments for elements such as Tc, dependent on the redox status

of the soil or the porewater pH, may alter the system dramatically and produce unrealistic results. The differences between  $K_d$  values determined in batch sorption experiments and those determined in *in-situ* core or field studies have been extensively documented (Sheppard et al. 1987; Hoffman et al. 1982; Matthies et al. 1989).

Solution or porewater concentration effects on  $K_d$  are usually the same for all elements. Data for Cs (Hoeffner 1985; Nikula 1982) and Sr (Nikula 1982) indicate

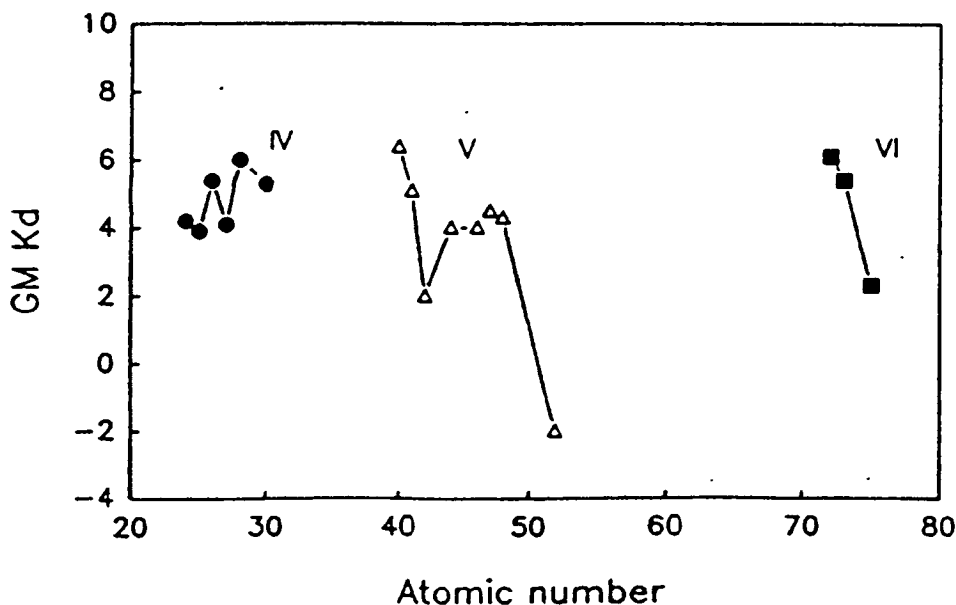
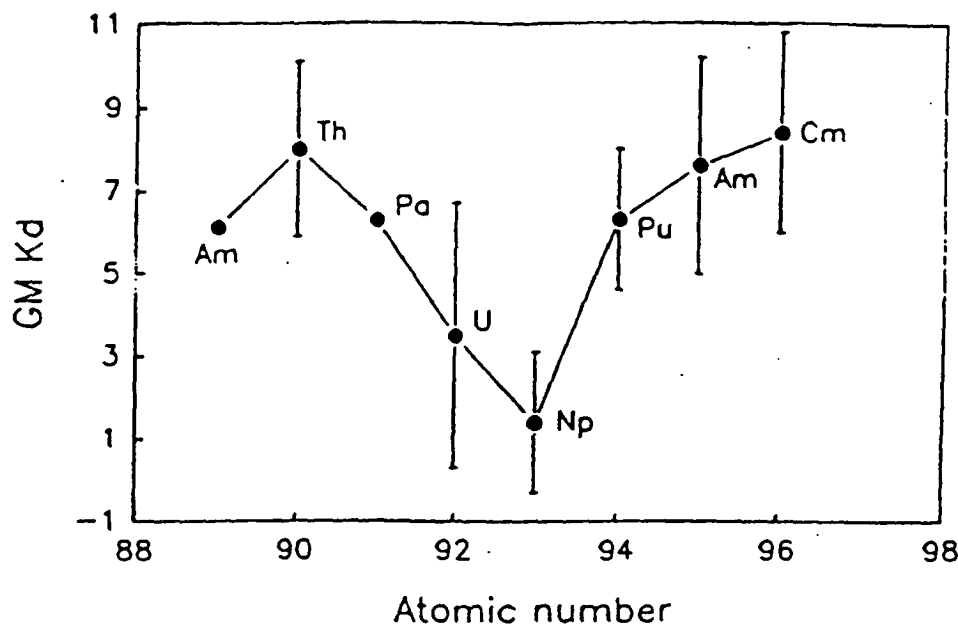


Fig. 2. Geometric mean  $\ln K_d$  values for sandy soils vs. atomic number for all the transition elements, for periods IV, V, and VI.


 Fig. 3. Geometric mean  $\ln K_d$  values vs. atomic number for the actinide elements for sandy soils.

that a decrease in the molarity of the solution results in an increase in the  $K_d$  value. Data for I, Tc, Np, Cr, Mo, Cs, and U (Sheppard et al. 1987; Sheppard and Thibault 1990) showed a similar negative relationship between porewater concentration and  $K_d$  after 1 and 4 y in undisturbed sandy soil cores.

The presence of competing cations will have a most dramatic effect on the sorption of elements that behave predominantly as cations, but can also have an effect on anionic transport (Bond and Smiles 1988). For example, the sorption of Cd and Ra decreased when Ca was present in the soil porewater since Ca competes for the sorption

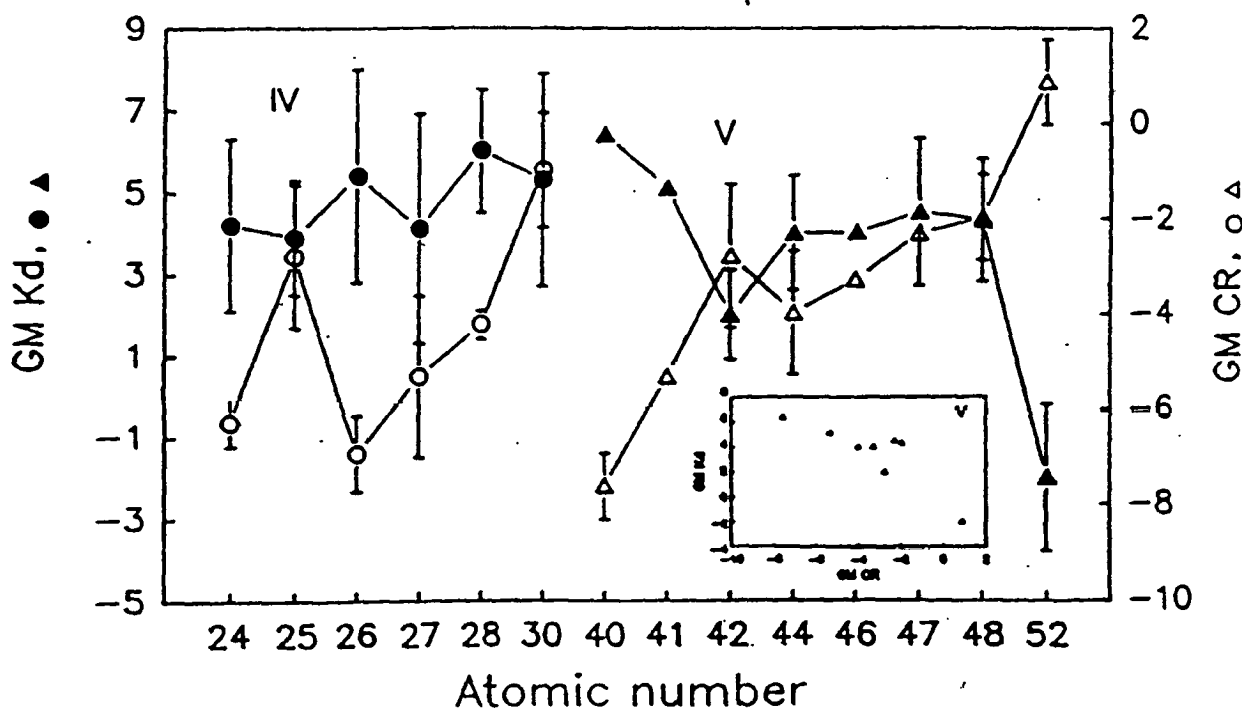

 Fig. 4. Geometric mean  $\ln K_d$  values for sandy soils and generic CR values vs. atomic number for transition elements in periods IV and V.

Table 3. Comparison of our compilation with those of Baes and Sharp (1983) and Coughtrey et al. (1985).

Element	This Study <sup>a</sup>								Baes & Sharp <sup>b</sup>		Coughtrey et al. <sup>c</sup>	
	Sand		Loam		Clay		Organic		Agricultural soils and clays of pH 4.5 to 7.0		Best Estimate	Range
	exp(m)	Range	exp(m)	Range	exp(m)	Range	exp(m)	Range	exp(m)	Range		
Ac	450	ND <sup>d</sup>	1500	ND	2400	ND	5400	ND	ND	ND	50	ND
Ag	90	2.7-1000	120	20-333	180	100-300	15000	4400-33000	110	10-1000	N 2000	1200-8700
Am	1900	0.2-300000	9400	400-40389	8400	25-400000	112000	6390-450000	810	1.0-47230	ND	ND
Br	15	ND	50	ND	75	ND	180	ND	ND	ND	<2	ND
Ca	5	ND	30	ND	50	ND	90	ND	4.1	1.2-9.8	ND	ND
Cd	80	2.7-625	40	7.0-962	560	112-2450	800	23-17000	6.7	1.26-26.0	32-50	ND
Co	500	40-3948	8100	1200-54000	20000	12000-11623	3300	0	1100	50-6000	ND	ND
Cu	4000	780-22970	18000	7644-44240	6000	ND	4000	0	2300	93.3-51900	ND	90-52000
Ce	60	.07-9000	1300	100-9700	550	20-14000	1000	120-4500	55	0.2-3800	ND	ND
Cr	70	1.7-1729	30	2.2-1000	1500	ND	270	6-2517	ND	ND	ND	ND
Cr(2)	ND	ND	ND	ND	ND	ND	ND	ND	2200	470-150000	ND	ND
Cr(6)	ND	ND	ND	ND	ND	ND	ND	ND	37	1.2-1800	ND	ND
Cs	280	0.2-10000	4400	540-61207	1900	37-31500	270	.4-145000	1100	10-52000	1800	1000-100000
Fe	220	5-4800	800	290-2240	145	15-2121	600	0	55	1.4-1000	9	4-9
I	1	.04-81	5	0.1-43	1	.2-29	25	1.4-160	ND	ND	N 6	ND
K	15	ND	55	ND	75	ND	200	ND	5.5	2.0-9.0	ND	ND
Mn	50	6.4-5000	750	40-77079	180	23.6-48945	150	0	150	0.2-10000	20	19-99
Mo	10	1.0-52	125	ND	90	13-480	25	14-50	20	0.37-400	9	ND
Ni	400	60-3600	300	ND	650	385-2467	1100	360-4700	ND	ND	N 20	ND
Np	5	0.5-390	25	1.3-79	55	.4-2575	1200	657-1900	11	0.16-929	N 50	0.14-929
Pb	270	19-1405	16000	3500-59000	550	ND	22800	9000-31590	99	4.5-7640	ND	ND
Po	150	9-7020	400	24-1830	1000	ND	7100	ND	540	196-1063	ND	ND
Pu	550	27-36000	1280	100-5933	5100	316-190000	1900	48-62000	1800	11-300000	5000	10-10000
Ra	10	ND	40	ND	60	ND	150	ND	ND	ND	ND	ND
Ru	55	5-490	1000	ND	800	ND	66000	39000-67000	220	40-1000	1-20	ND
Sa	55	16-70	150	ND	115	26-246	170	105-310	ND	ND	>9	ND
Sa(0)	ND	ND	ND	ND	ND	ND	ND	ND	2.7	1.2-0.6	ND	ND
Sr	15	.05-190	20	.01-300	110	3.6-32000	158	8-4800	27	0.15-3360	ND	ND
Tc	.1	.01-16	0.1	.01-.4	1	1.16-1.32	1	.02-340	.031	.0029-0.28	0.11	ND
Th	3200	207-150000	3180	ND	5800	244-140000	89000	1579-13000000	60000	2000-510000	ND	ND
U	35	.03-2200	15	.2-4500	1600	46-195100	410	33-7350	45	10.5-4400	ND	ND
Y	170	ND	720	ND	1000	ND	2600	ND	510*	160-1640*	ND	ND
Zn	200	.1-8080	1300	3.6-11080	2400	200-100000	1600	70-13000	16	0.1-0000	>20	ND

<sup>a</sup> From our study, Tables A-1 to A-4.<sup>b</sup> Baes III and Sharp, 1983.<sup>c</sup> Coughtrey et al. 1985.<sup>d</sup> No data available.<sup>e</sup> From Baes III et al. 1984. (Table 2.13)

sites. Gee et al. (1983) reported a decrease in the  $K_d$  value for Cs from 1000 to about 10 when the molarity of NaCl in the solution increased from 0.002 M to 1.0 M. Nikula (1982) also reported this for sorption of Cs on sediments, and Hoeffner (1985) found this effect for Co and Sr for several competing salts. Uchida and Kamada (1987), Nikula (1982), and Routson et al. (1984) also confirmed the effect of Ca on the  $K_d$  values for Sr.

Wolfum and Bunzl (1986) reported that the  $K_d$  values measured for Tc decreased with an increase in molarity of  $\text{CaCl}_2$  in the porewater of a sphagnum peat. Wolf et al. (1977) found a decrease in  $K_d$  value for I when Ca was present. Since Tc and I usually behave as anions, this is further evidence that the presence and concentration of cations may have an effect on the migration of anions.

We examined the effect of pH on  $K_d$  for all elements in all soils. It was expected that the  $K_d$  value might increase toward higher pH values since acidic conditions often tend to keep contaminants in solution. No such effect was observed, although we know that the sorption of many elements, especially the actinides (Coughtrey et al. 1985) including Np (Nakayama 1988; NCRP 1988), is dependent on soil pH (Buchter et al. 1989).

The  $K_d$  data are most complete for sand soils. We plotted the GM  $K_d$  value in sand vs. atomic number of the elements by Mendeleev group (Fig. 1). Generally, within each group the  $K_d$  value increases with atomic number except for group VIIA, where I exhibited significantly lower  $K_d$  values than all other elements. Technetium, with atomic number 43 and a member of group VIIb (not shown in Fig. 1), has a GM  $\ln K_d$  value of -2.7 and also deviates from this general trend. A plot of these  $K_d$  values for all transition elements vs. atomic number shows similarity among periods and that the  $K_d$  value is relatively constant across the periods (Fig. 2). A plot of the GM  $K_d$  values for actinides vs. atomic number indicates that Np is the least sorptive (Fig. 3).

Comparisons of soil  $K_d$  values with CR values taken from Baes et al. (1984) for period IVA and VA elements showed the expected trends of high CR values corresponding to low  $K_d$  values (Fig. 4). This trend reflects the negative correlation between the two (Sheppard 1985; Sheppard and Sheppard 1989).

#### Comparison of data bases

The GM  $K_d$  values are summarized for each element by soil texture in Table 1, with the GSDs, ranges, and number of observations given in the Appendix. Our  $K_d$  values are compared to the well-accepted data of Baes and Sharp (1983) and a more recent compendium published by Coughtrey et al. (1985) in Table 3. Coughtrey

et al. (1985) do not include data for as many elements as Baes and Sharp, but there is general agreement. To compare the data of Baes and Sharp with our study, we used only our data for loam and clay because these represented the agricultural soils summarized by Baes and Sharp. The GM  $K_d$  values of Baes and Sharp are generally lower than our values for loam and clay, except for Cr, Po, Pu, Sr, U, and Th. Use of lower  $K_d$  values would lead to lower predicted soil concentrations in assessment models and could underpredict doses in some pathways.

## CONCLUSIONS

We have reviewed the literature and updated the soil  $K_d$  compendiums published by Baes and Sharp (1983), Sheppard et al. (1984), and Coughtrey et al. (1985). Our compilation also specifies  $K_d$  values by soil type. We conclude the  $K_d$  literature could be greatly improved if more attention was given to comprehensive descriptions of the soil used in the  $K_d$  determinations. Examination of the factors affecting solute transport, including solute reactivity and dependencies on concentration, pH, and redox status led us to conclude that the simplicity of the  $K_d$  model presently makes it appropriate for screening assessments. Modelers must be cautioned that the use of generic  $K_d$  values can lead to gross error if used to predict absolute impacts of a disposal facility; site-specific  $K_d$  values are essential.

This default  $K_d$  data base allows some general conclusions to be made about  $K_d$  values. Soil texture in mineral soils for all elements does not affect  $K_d$ ; however, for certain elements (Ce, Pu, Sr, and Zn) an increase in surface area, the predominant effect of texture, dramatically increased the  $K_d$  value. An increase in soil organic matter generally increased the soil  $K_d$  value. The presence and concentration of competing cations generally decreased the  $K_d$  value as expected, and pH had no significant effect in all soils and elements. Plots of soil  $K_d$  values vs. atomic number showed that  $K_d$  tended to increase with atomic number within the seven element groups. The  $K_d$  values for the actinides showed a distinct pattern, with Np as the least sorptive and most mobile actinide element. For poorly known elements, the use of the CR- $K_d$  correlation and position in the periodic table appears to be a reasonable method to predict a default  $K_d$  value.

*Acknowledgments*—The authors wish to thank P. A. Smith for data manipulation and C. L. Gagawchuk and D. Burgoyne for manuscript editing. We also thank S. C. Sheppard, M. E. Stephens, and R. Zach for helpful comments on a draft of the manuscript.

## REFERENCES

- Ames, L. L.; Rai, D. Radionuclide interactions with soil and rock media. Vol. 1 and 2. Las Vegas, NV: U.S. EPA, Office of Radiation Programs; EPA 520/6-78-007-b; 1978.
- Baes, C. F., III; Sharp, R. D. A proposal for estimation of soil leaching and leaching constants for use in assessment models. *J. Environ. Qual.* 12(1):17-28; 1983.
- Baes, C. F., III; Sharp, R. D.; Sjoreen, A. L.; Shor, R. W. A review and analysis of parameters for assessing transport of

- environmentally released radionuclides through agriculture. Oak Ridge, TN: Health and Safety Division, Oak Ridge National Laboratory; ORNL-5786; 1984.
- Bond, W. J.; Smiles, D. E. Predicting the average movement of reactive solutes in soils. *Soil Use Mgmt.* 4(4):115-120; 1988.
- Buchter, B.; Davidoff, B.; Amacher, M. C.; Hinz, C.; Iskandar, I. K.; Selim, H. M. Correlation of Freundlich  $K_d$  and  $n$  retention parameters with soils and elements. *Soil Sci.* 148: 370-380; 1989.
- Coughtrey, P. J.; Jackson, D.; Thorne, M. C. Radionuclide distribution and transport in terrestrial and aquatic ecosystems. A compendium of data. Netherlands: A. A. Balkema; 1985.
- Gee, G. W.; Rai, D.; Serne, R. J. Mobility of radionuclides in soil. In: *Chemical mobility and reactivity in soil systems*. SSSA Spec. Publ. 11:203-227; 1983.
- Gillham, R. W.; Cherry, J. A.; Lindsay, L. E. Cesium distribution coefficients in unconsolidated geological materials. *Health Phys.* 39:637-649; 1980.
- Hoefner, S. L. Radionuclide sorption on Savannah River Plant burial ground soil—a summary and interpretation of laboratory data. Aiken, SC: Savannah River Lab. E. I. du Pont de Nemours and Co.; Report DP-1702; 1985.
- Hoffman, F. O.; Garten, C. T., Jr.; Lucas, D. M.; Huckabee, J. W. Environmental behavior of technetium in soil and vegetation. Implications for radiological assessments. *Environ. Sci. Technol.* 16:214-217; 1982.
- Isherwood, D. Geoscience data base handbook for modeling a nuclear waste repository, 1. Washington, D.C.: U.S. Nuclear Regulatory Commission: NUREG/CR-0912, Vol. 1; UCRL-52719, Vol. 1; 1981.
- Killey, R. W. D.; McHugh, J. O.; Champ, D. R.; Cooper, E. L.; Young, J. L. Subsurface cobalt-60 migration from a low-level waste disposal site. *Environ. Sci. Technol.* 18:148-157; 1984.
- Matthies, M.; Bruggemann, R.; Munzer, B.; Schemewski, G.; Trapp, S. Exposure and ecotoxicity estimation for environmental chemicals (E4CHEM): application of fate models for surface water and soil. *Ecol. Modelling* 47:115-130; 1989.
- Mehta, K.; Goodwin, B. W. Identification of contaminants of concern for the postclosure assessment of the Canadian concept for nuclear fuel waste disposal. Chalk River, Ontario: Chalk River Nuclear Laboratories; Atomic Energy of Canada Limited Report AECL-XXXXX; 1990.
- Nakayama, S.; Arimoto, H.; Yamada, N.; Moriyama, H.; Higashi, K. Column experiments on migration of neptunium (V). *Radiochimica Acta* 44/45:179-182; 1988.
- National Council on Radiological Protection. Neptunium: Radiation protection guidelines. Bethesda, MD: NCRP; NCRP Report No. 90; 1988.
- Nikula, A. Sorption in typical Finnish soils and some fracture filling of bedrock. Helsinki, Finland: Imatra Power Company; 1982 (in Finnish).
- Rai, D.; Zachara, J. M. Chemical attenuation rates, coefficients, and constants in leachate migration, Vol. 1. Palo Alto, CA: Electric Power Research Institute; EPRI EA-3356; 1984.
- Routson, R. C.; Barney, G. S.; Smith, R. M. Radionuclide sorption on low-exchange capacity Hanford Site soils. *Commun. Soil Sci. Plant Anal.* 15(4):375-400; 1984.
- SAS. SAS user's guide, statistics. Version 5 edition. Cary, NC: SAS Institute Inc.; 1985.
- Sheppard, M. I. Radionuclide partitioning coefficients in soils and plants and their correlation. *Health Phys.* 49:106-111; 1985.
- Sheppard, M. I.; Beals, D. I.; Thibault, D. H.; O'Connor, P. Soil nuclide distribution coefficients and their statistical distribution. Chalk River, Ontario: Chalk River Nuclear Labs; Atomic Energy of Canada Limited Report AECL-8364; 1984.
- Sheppard, M. I.; Bera, K. E. Sensitivity analysis of the SCEMR1 model. Chalk River, Ontario: Chalk River Nuclear Labs; Atomic Energy of Canada Limited Technical Record TR-318; 1984.
- Sheppard, M. I.; Thibault, D. H. A four-year mobility study of selected trace elements and heavy metals. *J. Env. Qual.* 1990 (in press).
- Sheppard, M. I.; Thibault, D. H.; Mitchell, J. H. Element leaching and capillary rise in sandy soil cores: Experimental results. *J. Env. Qual.* 16:273-284; 1987.
- Sheppard, M. I.; Vandergraaf, T. T.; Thibault, D. H.; Reid, J. A. K. Technetium and uranium: Sorption by and plant uptake from peat and sand. *Health Phys.* 44:635-643; 1983.
- Sheppard, S. C.; Evenden, W. G. The assumption of linearity in soil and plant concentration ratios: An experimental evaluation. *J. Environ. Radioactivity* 7:221-247; 1988.
- Sheppard, S. C.; Evenden, W. G. Comparison of partition coefficients for  $^{54}\text{Mn}$  and soil-extractable Mn, including relationship to plant uptake. *Can. J. Soil Sci.* 69:351-365; 1989.
- Sheppard, S. C.; Sheppard, M. I. Impact of correlations on stochastic estimates of soil contamination and plant uptake. *Health Phys.* 57:653-657; 1989.
- Sheppard, S. C.; Sheppard, M. I.; Evenden, W. G. A novel method used to examine variation in Tc sorption among 34 soils, aerated and anoxic. *J. Environ. Radioactivity* 11:215-233; 1990.
- Thibault, D. H.; Sheppard, M. I.; Smith, P. A. A critical compilation and review of default soil solid/liquid partition coefficients,  $K_d$ , for use in environmental assessments. Chalk River, Ontario: Chalk River Nuclear Labs; Atomic Energy of Canada Limited Report AECL-10125; 1990.
- Till, J. E.; Meyer, H. R., eds. Radiological assessment: A textbook on environmental dose analysis. Washington, D.C.: U.S. Nuclear Regulatory Commission: NUREG/CR-3332; ORNL-5968; 1983.
- Uchida, S.; Kamada, H. Sorption of strontium on soils in layered and aerated zones. *Hoken Butsuri* 22(2):179-187; 1987.
- Wolf, A. I.; Bunzl, K.; Dietl, F.; Schmidt, W. F. Effect of calcium ions on the absorption of lead (2+), copper (2+), cadmium (2+), and zinc (2+) by humic substances. *Chemosphere* 6: 207-213; 1977.
- Wolfrum, C.; Bunzl, K. Sorption and desorption of technetium by humic substances under oxic and anoxic conditions. *J. Radioanal. Nucl. Chem.* 99(2):315-323; 1986.



## Appendix

Table A-1. Sand soil  $K_d$  values ( $L\ kg^{-1}$ ).

Element	# Observations	$\mu^a$	$\sigma^b$	exp ( $\mu$ ) <sup>c</sup>	Range
Ac		6.1		450	
Ag	12	4.5	1.8	90	2.7 to 1 000
Am	29	7.6	2.6	1 900	8.2 to 300 000
Be		5.5		250	
Bi		4.6		100	
Br		2.7		15	
C	3	1.1	0.8	5	1.7 to 7.1
Ca		1.8		5	
Cd	14	4.3	1.5	80	2.7 to 625
Ce	12	6.2	1.6	500	40 to 3 968
Cm	2	8.4	2.4	4 000	780 to 22 970
Co	33	4.1	2.8	60	.07 to 9 000
Cr	15	4.2	2.1	70	1.7 to 1 729
Cs	81	5.6	2.5	280	0.2 to 10 000
Fe	16	5.4	2.6	220	5 to 6 000
H	3	-2.7	0.4	0.06	0.04 to 0.1
Hf		6.1		450	
Ho		5.5		250	
I	22	0.04	2.2	1.0	0.04 to 81
K		2.6		15	
Mn	54	3.9	1.4	50	6.4 to 5 000
Mo	15	2.0	1.1	10	1.0 to 52
Nb		5.1		160	
Ni	11	6.0	1.5	400	60 to 3 600
Np	16	1.4	1.7	5	0.5 to 390
P		1.8		5	
Pa		6.3		550	
Pb	3	5.6	2.3	270	19 to 1 405
Pd		4.0		55	
Po	36	5.0	1.6	150	9 to 7 020
Pu	39	6.3	1.7	550	27 to 36 000
Ra	3	6.2	3.2	500	57 to 21 000
Rb		4.0		55	
Re		2.3		10	
Ru	7	4.0	1.4	55	5 to 490
Sb	1	3.8		45	
Se	3	4.0	0.4	55	36 to 70
Si		3.5		35	
Sm		5.5		245	
Sn		4.9		130	
Sr	81	2.6	1.6	15	0.05 to 190
Ta		5.4		220	
Tc	19	-2.0	1.8	0.1	0.01 to 16
Te		4.8		125	
Th	10	8.0	2.1	3 200	207 to 150 000
U	24	3.5	3.2	35	0.03 to 2 200
Y		5.1		170	
Zn	22	5.3	2.6	200	0.1 to 8 000
Zr		6.4		600	

a Mean of the natural logarithms of the observed values.

b Standard deviation of the natural logarithms of the observed values.

c Geometric mean rounded to two significant digits. Default values for  $\mu$  and exp ( $\mu$ ) have been predicted using CRs for nuclides with no "# observations."

Table A-2. Loam soil  $K_d$  values ( $L\ kg^{-1}$ ).

Element	# Observations	$\mu^a$	$\sigma^b$	exp ( $\mu$ ) <sup>c</sup>	Range
Ac		7.3		1 500	
Ag	4	4.8	1.1	120	28 to 333
Am	20	9.2	1.4	9 600	400 to 48 309
Be		6.7		800	
Bi		6.1		450	
Br		3.9		50	
C		2.9		20	
Ca		3.4		30	
Cd	8	3.7	1.6	40	7.0 to 962
Ce	5	9.0	1.5	8 100	1 200 to 56 000
Cm	4	9.8	0.7	18 000	7 666 to 44 260
Co	23	7.2	1.3	1 300	100 to 9 700
Cr	4	3.4	2.9	30	2.2 to 1 000
Cs	54	8.4	1.3	4 600	560 to 61 287
Fe	18	6.7	0.7	800	290 to 2 240
H		3.0		20	
Hf		7.3		1 500	
Ho		6.7		800	
I	33	1.5	2.0	5	0.1 to 43
K		4.0		55	
Mn	38	6.6	2.6	750	40 to 77 079
Mo		4.8		125	
Nb		6.3		550	
Ni		5.7		300	
Np	11	3.2	1.2	25	1.3 to 79
P		3.2		25	
Pa		7.5		1 800	
Pb	3	9.7	1.4	16 000	3 500 to 59 000
Pd		5.2		180	
Po	13	6.0	1.3	400	24 to 1 830
Pu	21	7.1	1.2	1 200	100 to 5 933
Ra	3	10.5	3.1	36 000	1 262 to 530 000
Rb		5.2		180	
Re		3.7		40	
Ru	2	6.9		1 000	
Sb		5.0		150	
Se	1	5.0		150	
Si		4.7		110	
Sm		6.7		800	
Sn		6.1		450	
Sr	43	3.0	1.7	20	0.01 to 300
Ta		6.8		900	
Tc	10	-2.3	1.1	0.1	0.01 to 0.4
Te		6.3		500	
Th		8.1		3 300	
U	8	2.5	3.3	15	0.2 to 4 500
Y		6.6		720	
Zn	12	7.2	2.4	1 300	3.6 to 11 000
Zr		7.7		2 200	

a Mean of the natural logarithms of the observed values.

b Standard deviation of the natural logarithms of the observed values.

c Geometric mean rounded to two significant digits. Default values for  $\mu$  and exp ( $\mu$ ) have been predicted using CRs for nuclides with no "# Observations."

Table A-3. Clay soil  $K_d$  values ( $L\ kg^{-1}$ ).

Element	# Observations	$\mu^a$	$\sigma^b$	exp ( $\mu$ ) <sup>c</sup>	Range
Ac		7.8		2 400	
Ag	5	5.2	0.4	180	100 to 300
Am	11	9.0	2.6	8 400	25 to 400 000
Ba		7.2		1 300	
Bi		6.4		600	
Br		4.3		75	
C		0.8		1	
Ca		3.9		50	
Cd	10	6.3	0.9	560	112 to 2 450
Ce	4	9.9	0.5	20 000	12 000 to 31 6
Cm		8.7		6 000	
Co	15	6.3	1.8	550	20 to 14 000
Cr		7.3		1 500	
Cs	28	7.5	1.6	1 900	37 to 31 500
Fe	7	5.1	1.6	165	15 to 2 121
H		3.3		30	
Hf		7.8		2 400	
Ho		7.2		1 300	
I	8	0.5	1.5	1	0.2 to 29
K		4.3		75	
Mn	23	5.2	2.0	180	23.6 to 48 941
Mo	7	4.5	1.2	90	13 to 400
Nb		6.8		900	
Ni	10	6.5	0.7	650	305 to 2 467
Np	4	4.0	3.8	55	0.4 to 2 575
P		3.5		35	
Pa		7.9		2 700	
Pb		6.3		550	
Pd		5.6		270	
Po		8.0		3 000	
Pu	18	8.5	2.1	5 100	316 to 190 00
Ra	8	9.1	1.3	9 100	696 to 56 000
Rb		5.6		270	
Re		4.1		60	
Ru	1	6.7		800	
Sb		5.5		250	
Se	14	4.7	0.5	115	36 to 246
Si		5.2		180	
Sm		7.2		1 300	
Sn		6.5		670	
Sr	24	4.7	2.0	110	3.6 to 32 000
Ta		7.1		1 200	
Tc	4	0.2	0.06	1	1.16 to 1.32
Te		6.6		720	
Th	5	8.6	2.6	5 800	244 to 160 0
U	7	7.3	2.9	1 600	46 to 395 10
Y		6.9		1 000	
Zn	23	7.8	1.4	2 400	200 to 100 0
Zr		8.1		3 300	

a Mean of the natural logarithms of the observed values.

b Standard deviation of the natural logarithms of the observed values.

c Geometric mean rounded to two significant digits. Default values for  $\mu$  and exp ( $\mu$ ) have been predicted using CRs for nuclides with no "# Observations."

Note that the last part of these ranges may be cut off

Table A-4. Organic soil  $K_d$  values ( $L\ kg^{-1}$ ).

Element	# Observations	$\mu^a$	$\sigma^b$	$\exp(\mu)^c$	Range
Ac		8.6		5 400	
Ag	4	9.6	0.9	15 000	4 400 to 33 000
Am	5	11.6	1.7	112 000	6 398 to 450 000
Be		8.0		3 000	
Bi		7.3		1 500	
Br		5.2		180	
C		4.2		70	
Ca		4.5		90	
Cd	9	6.7	2.3	800	23 to 17 000
Ce	1	8.1		3 300	
Cm	1	8.7		6 000	
Co	6	6.9	1.5	1 000	120 to 4 500
Cr	4	5.6	2.7	270	6.0 to 2 517
Cs	9	5.6	3.6	270	0.4 to 145 000
Fe	1	6.4		600	
H		4.3		75	
Hf		8.6		5 400	
Ho		8.0		3 000	
I	9	3.3	2.0	25	1.4 to 368
K		5.3		200	
Mn	1	5.0		150	
Mo	3	3.3	0.5	25	18 to 50
Nb		7.6		2 000	
Ni	6	7.0	0.9	1 100	360 to 4 700
Np	3	7.1	0.4	1 200	857 to 1 900
P		4.5		90	
Pa		8.8		6 600	
Pb	6	10.0	0.5	22 000	9 000 to 31 590
Pd		6.5		670	
Po		8.9		7 300	
Pu	7	7.5	2.6	1 900	60 to 62 000
Ra		7.8		2 400	
Rb		6.5		670	
Re		5.0		150	
Ru	5	11.1	0.3	66 000	39 000 to 87 000
Sb		6.3		550	
Se	4	5.1	0.5	170	105 to 310
Si		6.0		400	
Sm		8.0		3 000	
Sn		7.4		1 600	
Sr	12	5.0	1.8	150	8 to 4 800
Ta		8.1		3 300	
Tc	24	0.4	1.4	1	0.02 to 340
Te		7.5		1 900	
Th	3	11.4	4.6	89 000	1 579 to 13 000 000
U	6	6.0	2.5	410	33 to 7 350
Y		7.9		2 600	
Zn	8	7.4	1.6	1 600	70 to 13 000
Zr		8.9		7 300	

a Mean of the natural logarithms of the observed values.

b Standard deviation of the natural logarithms of the observed values.

c Geometric mean rounded to two significant digits. Default values for  $\mu$  and  $\exp(\mu)$  have been predicted using CRs for nuclides with no "# Observations."

**AR-080**