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PRELIMINARY REPORT ON RETARDATION FACTORS AND
RADIONUCLIDE MIGRATION

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MASTER

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ABSTRACT

The available data on distribution coefficients for the biologically important radionuclides present in high level-waste were used to estimate retardation factors (K_f) for a mass transport hydrologic model. The radionuclides were divided into 3 groups: fission products with no sorption ($K_f = 1$), fission products with sorption ($K_f = 10^2$), and the actinides and their daughter products ($K_f = 10^4$). Minimum and maximum values were assigned the latter two groups. Uncertainties as a function of time were estimated at $\pm$ an order of magnitude.

INTRODUCTION

The primary geochemical control on radionuclide migration from a high-level waste repository is ion exchange. Other controls which constitute the geochemical barrier to radionuclide migration are ion filtration, hydrolysis and solubility.

The retention of radionuclides via ion exchange can be quantitatively expressed as a retardation factor; that is, the relative velocity of the groundwater to that expected for the individual radionuclide. The chief purpose of this report is documentation of the retardation factors used in Lawrence Livermore Laboratory's hydrologic modeling of radionuclide migration from a high-level waste repository. More detailed discussions on radionuclide chemistry, geochemical controls and how they influence waste/water/rock interactions are planned for future reports.

The mass transport hydrologic model referred to in this study was developed by The Analytic Sciences Corporation (TASC) for Lawrence Livermore Laboratory's high-level waste management program funded by NRC.

RETARDATION FACTORS

The retardation factor, $K_f = \frac{\text{water velocity}}{\text{nuclide velocity}} = 1 + rK_d$ where

K_d equals the ion exchange distribution coefficient, that is,

$$K_d = \frac{\left(\frac{\text{amount of nuclide}}{\text{in solid phase}} \right) / \left(\frac{\text{weight}}{\text{of solid}} \right)}{\left(\frac{\text{amount of nuclide}}{\text{in liquid phase}} \right) / \left(\frac{\text{volume of}}{\text{liquid}} \right)} = \left(\frac{\text{ml}}{\text{g}} \right)$$

and

$$r = \left(\frac{1 - \theta}{\theta} \right) \rho \quad \begin{array}{l} \rho = \text{rock density} \\ \theta = \text{effective porosity} \end{array}$$

When there is no sorption, $K_d = 0$ and the minimum value for the retardation factor is 1. Also note, as porosity increases, r decreases causing a decrease in K_f (Figure 1). For more complete discussions on sorption and the relative movement of radionuclides in groundwater, see Borg, et al, (1976) and Levy (1972).

Although the retardation factor is primarily defined as a function of the ion exchange distribution coefficient, retardation factors as used in the transport model represent the combined effects of ion exchange and adsorption of charged colloidal species on silica surfaces. Other processes which also affect radionuclide concentrations (e.g. low solubilities) are not accounted for in the retardation factor.

The retardation factors used in TASC's transport model originally came from Burkholder's estimated values for sorption in a western U.S. desert soil (Table 1). A recent evaluation of these retardation factors based on published K_d 's shows Burkholder's results are applicable to sedimentary rocks as well.

Figure 1: Retardation factor (K_f) as a function of the distribution coefficient (K_d).

TABLE I
RETARDATION FACTORS FOR WESTERN U.S. DESERT SOIL*

$K_f = 1$	Technetium		
	Iodine		
$K_f = 10^2$	Strontium	$K_f = 10^3$	Rubidium
	Antimony		Tin
	Polonium		Cesium
	Neptunium		Radium
$K_f = 10^4$	Yttrium	$K_f = 10^5$	Thorium
	Zirconium		
	Promethium		
	Niobium		
	Lead		
	Protactinium		
	Uranium		
	Plutonium		
	Americium		
	Curium		
	Europium		

*Burkholder, NR - CONF - 001, p. 456, values rounded to nearest order of magnitude.

As it was not practical to treat each radionuclide separately in the transport model, the 23 biologically significant nuclides in Table 2 were divided into three groups: fission products with no sorption properties, fission products with sorption, and the actinides and their daughter products. The fission product/actinide division separates the α emitters from the β and γ emitters. Maximum, minimum and preferred values were assigned for each group for both the shale and salt repositories. Values used in TASC's transport model are given in Table 3. Uncertainties with time for retardation factors are $\pm$ order of magnitude. Note that for the salt repository, sorption is modeled only in the aquifer (i.e., $K_f = 1$ in the salt layer and barrier bed). Salt has essentially no exchange capacity. In the barrier bed, it is assumed that the high salt content of the groundwater* will saturate the exchange sites to the extent that sorption will be severely limited. In the aquifer, it is assumed that dilution of the salty groundwater will decrease the salinity effect on ion exchange and sorption will take place as the radionuclides are transported through the aquifer.

The same retardation factors are used for both interstitial and fracture flow. This assumption is based on Fried and others (1977) Pu retardation experiments with artificial fissures in basalt.

Whether retardation factors are actually the same for both interstitial and fracture flow depends on the rate of ion exchange; that is, whether the passage time through the sorbing medium is sufficient for equilibrium to occur. Unfortunately, rates of reaction for radionuclide ion exchange are unknown. Because the flow rates for interstitial and fracture flow are low

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*Data from the Salado and Castille Formations of the Delaware Basin show local groundwater concentrations of $> 300,000$ ppm TDS.

TABLE 2
BIOLOGICALLY SIGNIFICANT NUCLIDES
From 10 yrs. to 10⁶ After Irradiation

Nuclide	T 1/2	Specific Activity ($\frac{\text{Ci}}{\text{g}}$)	MPC _w ($\frac{\text{Ci}}{\text{m}^3}$)	MPC _w ($\frac{\text{moles}}{\text{g}}$)	Time Significant (yr)
⁹⁰ Sr	28 yr	1.42 X 10 ²	3 X 10 ⁻⁷	2 X 10 ⁻¹⁴	10 - 100
⁹⁰ Y	64 hr	5.44 X 10 ⁵	2 X 10 ⁻⁵	4 X 10 ⁻¹⁶	10 - 100
⁹³ Zr	1.5 X 10 ⁶ yr	2.56 X 10 ⁻³	8 X 10 ⁻⁴	3 X 10 ⁻⁶	10 ³ - 10 ⁶
^{93m} Nb	14 yr	2.75 X 10 ⁻²	3 X 10 ⁻⁴	2 X 10 ⁻⁷	10 ³ - 10 ⁶
⁹⁹ Tc	2.1 X 10 ⁵ yr	1.72 X 10 ⁻²	3 X 10 ⁻⁴	2 X 10 ⁻⁷	10 ³ - 10 ⁵
¹²⁶ Sn	1 X 10 ⁵ yr	2.84 X 10 ⁻²	2 X 10 ⁻⁵	6 X 10 ⁻⁹	10 ³ - 10 ⁵
¹²⁹ I	1.7 X 10 ⁷ yr	1.63 X 10 ⁻⁴	6 X 10 ⁻⁸	3 X 10 ⁻⁹	10 ³ - 10 ⁶
¹³⁴ Cs	2.0 yr	1.33 X 10 ³	9 X 10 ⁻⁶	2 X 10 ⁻⁹	10
¹³⁷ Cs	30 yr	8.7 X 10	2 X 10 ⁻⁵	2 X 10 ⁻¹²	10 - 100
²¹⁰ Pb	20 yr	8.52 X 10 ¹	1 X 10 ⁻⁷	6 X 10 ⁻¹⁵	10 ⁴ - 10 ⁶
²¹⁰ Po	38 yr	4.49 X 10 ³	7 X 10 ⁻⁷	7 X 10 ⁻¹⁶	10 ⁴ - 10 ⁶
²²⁶ Ra	1.6 X 10 ³ yr	9.90 X 10 ⁻¹	3 X 10 ⁻⁸	1 X 10 ⁻¹³	10 ⁴ - 10 ⁶
²²⁹ Th	7.3 X 10 ³ yr	2.0 X 10 ⁻¹	7 X 10 ⁻⁶	2 X 10 ⁻¹⁰	10 ⁵ - 10 ⁶
²³⁰ Th	8.0 X 10 ⁴ yr	2.0 X 10 ⁻²	2 X 10 ⁻⁶	4 X 10 ⁻¹⁰	10 ⁵ - 10 ⁶
²³¹ Pa	3.3 X 10 ⁴ yr	4.7 X 10 ⁻²	9 X 10 ⁻⁷	8 X 10 ⁻¹¹	10 ⁶
²³³ U	1.6 X 10 ⁵ yr	9.6 X 10 ⁻³	3 X 10 ⁻⁵	1 X 10 ⁻⁸	10 ⁵ - 10 ⁶
²³⁷ Np	2.1 X 10 ⁶ yr	7.19 X 10 ⁻⁴	3 X 10 ⁻⁶	2 X 10 ⁻⁸	10 ³ - 10 ⁶

Continued

BIOLOGICALLY SIGNIFICANT NUCLIDES (continued)

From 10 yrs. to 10^6 After Irradiation

Nuclides	$T_{1/2}$	Specific Activity ($\frac{\text{Ci}}{\text{g}}$)	$\text{MPC}_w (\frac{\text{Ci}}{\text{m}^3})$	$\text{MPC}_w (\frac{\text{moles}}{\text{L}})$	Time Significant (yr)
^{238}Pu	86 yr	1.75×10^1	5×10^{-6}	1×10^{-12}	10 - 100
^{239}Pu	2.4×10^4 yr	6.2×10^{-2}	5×10^{-6}	3×10^{-10}	$10^3 - 10^5$
^{240}Pu	6.6×10^3 yr	2.26×10^{-1}	5×10^{-6}	9×10^{-11}	$10^3 - 10^4$
^{241}Am	4.6×10^2 yr	3.23	4×10^{-6}	5×10^{-12}	$10 - 10^4$
^{243}Am	8.0×10^3 yr	1.84×10^{-1}	4×10^{-6}	8×10^{-11}	$10 - 10^4$
^{244}Cm	18 yr	8.14×10^1	7×10^{-5}	4×10^{-13}	10 - 100

EQUATIONS:

$$\text{Specific Activity} = \frac{N \times 1.873 \times 10^{-11}}{T_{1/2} (\text{sec})}$$

$$N = \frac{\text{atoms}}{\text{mole}}$$

$$M = \text{g/mole}$$

$$\text{S.A.} = \frac{1.12755 \times 10^1}{M \times T_{1/2} (\text{sec})} = \frac{\text{Ci}}{\text{g}}$$

$$\text{MPC}_w (\frac{\text{moles}}{\text{L}}) = \frac{\text{MPC}_w (\frac{\text{Ci}}{\text{g}})}{\text{Spec. Act.} \times 10^3 \times M}$$

TABLE 3
RETARDATION FACTORS FOR THE TRANSPORT MODEL

Shale Repository

	<u>Minimum</u>	<u>Preferred</u>	<u>Maximum</u>
Iodine and Technetium	1	1	1
Other Fission Products	1	² 10	³ 10
Actinides	² 10	⁴ 10	⁵ 10

Salt Repository

Iodine and Technetium	1	1	1
Other Fission Products			
Salt repository and barrier bed:	1	1	1
Aquifer:		² 10	³ 10
Actinides			
Salt repository and barrier bed:	1	1	1
Aquifer:	² 10	⁴ 10	⁵ 10

(10^{-3} to 10 m/yr) it is assumed that equilibrium conditions exist. However, the uncertainty in retardation factors for fracture pathways is greater than the uncertainty in retardation for interstitial flow.

Group 1. IODINE AND TECHNETIUM

Assigning a zero sorption value for iodine and technetium is based on theoretical and experimental evidence. Both elements exist in natural waters as the negative ions, I^- and TcO_4^- . Since soils and rocks generally exhibit no anion exchange capacity except for that sometimes shown by soil organic matter, sorption should be zero.*

Soil experiments by Battelle at Hanford (BNWL 1950, Part 2) gave K_d 's for TcO_4^- of 2.8 and 0.007 (basically zero for our purposes). K_d 's for I^- were .08 - 52.6. The 52.6 value was attributed to Fe and Al oxide coatings on the soil particles, otherwise I^- showed no sorbtion. Gast and others at the University of Minnesota (C00 - 2447-5) report similar K_d values for TcO_4^- in soils. In an earlier study of radionuclide migration at Hanford, Tc migrated at about the same rate as groundwater (i.e., $K_f = 1$) (Brown, 1976). K_d values for Tc were also zero over a wide range of $NaHCO_3$ concentrations for a South Carolina topsoil (Routson, et al., 1975).

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*Natural anion exchange materials include kaolinite, apatite and hydrous aluminum oxide. All have low anion exchange capacities and are limited in their distribution in rocks and soils.

Group 2. OTHER FISSION PRODUCTS

Fission products with sorption characteristics include ^{90}Sr , ^{90}Y , ^{93}Zr , $^{93\text{m}}\text{Nb}$, ^{126}Sn , ^{134}Cs , and ^{137}Cs (see Table 2). The preferred retardation factor of 10^2 given this group is a conservative estimate based on the sorption characteristics of Sr and Cs, the only major contributors among the fission products to the waste inventory. ^{90}Y , in secular equilibrium with ^{90}Sr , has a half life of only 64 hours. The $^{93}\text{Zr} \rightarrow ^{93\text{m}}\text{Nb}$ pair's inventory is 2×10^{-5} that of the combined inventory of ^{90}Sr plus ^{137}Cs in 10 year old waste. The ^{126}Sn inventory is even less than ^{93}Zr .

A compilation of Sr and Cs distribution coefficients shows a wide range of values (Table 4). For Sr, K_d 's vary from 0.19 in salt to 4000 in tuff. K_d 's for Cs show an even greater distribution: 0.027 in salt, 17,800 in tuff. With the exception of Amchitka basalts, Cs K_d 's are larger, often by an order of magnitude, than those for Sr. Of the 25 K_d 's for Sr and Cs in Table 4, only the following are applicable to the geology used in TASC's model (i.e., sedimentary rocks, excluding carbonates).

	<u>K_d (Sr)</u>	<u>K_d (Cs)</u>
Shaley siltstone	8	309
Sandstone	1.4	102
Sands	1.7-43	22-314
Sediments	50	300
Soils	9-282	250-1053

Based on the above K_d 's, 10 was chosen to represent an average Sr distribution coefficient. This gives a retardation factor of approximately 10^2 in sandstone (see Figure 1). Cs should have greater retardation than

TABLE 4

DISTRIBUTION COEFFICIENTS: Sr AND Cs*

Reference	K_d		Conditions
	Sr	Cs	
Angelo, et al., 1962	16-135	792-9520	Basalt, 32-80 mesh, prep. groundwater
Baetsle & Dejonghe, 1972	1.7-3.8	22-314	Quartz sand, Ph 7.7
Bertem, et al., 1962	4-9	8-9	Granodiorite, 100-200 mesh, prep. groundwater.
	11-23	1030-1810	Granodiorite, 0.5-1 mm, prep. groundwater.
Brown, 1976	50	300	Hanford sediments.
Douhy, 1967	45-75	800-10 ³	Tuff
Essington & Nork, 1969	220	39	Basalt, 0.5-4 mm, 300 ppm TDS
	1220	280	Basalt, 0.5-4 mm, 300 ppm TDS
	1.1	6.5	Basalt, 0.5-4 mm, sea water
Godse, et al., 1962	143-282	617-1053	Soil, pH 6.8
Goldberg, et al., 1962	2070-3480	12,000-17,800	Tuff, 100-200 mesh, prep. groundwater
Hajek & Knoll, 1966	19-43	189-420	Soils
Hoffman, 1976	400	5000-8000	Tuff, chimney rubble, groundwater
Inoue, 1976	9.4-71	250-10 ³	Soils, Ca groundwater
Nork and Fenske, 1970	260	1020	Tuff, >0.4 mm, prep. water
	0.19	13.5	Carbonate, >4 mm, prep. water
	1.7	34.3	Granite, >4 mm, groundwater
	8.32	309	Shaley siltstone, >4 mm, well water
	1.37	102	Sandstone, >4 mm, well water
	0.19	0.027	Salt, >4 mm, saturated salt water
Nork, et al., 1971	48-2454	121-3165	Alluvium, 0.5-4 mm, groundwater
Parsons, 1961	13-43	100	Sands
Robertson, 1976	3	-	Basalt, in situ measurements
Stead, 1963	5-14	-	Dolomite, 4000 ppm TDS
	4000	-	Tuff

*Taken in part from Borg, et al., 1976, Table 23.

10^2 , but, to simplify calculations in the transport model, was included with Sr. The preferred value of 10^2 is used for both shale and sandstone units. The maximum and minimum values reflect the range of values commonly found for Sr in Table 4. The estimated uncertainty in the retardation factor as a function of time is $\pm$ an order of magnitude.

Group 3. ACTINIDES AND DAUGHTER PRODUCTS

The biologically important actinides in Table 2 are the products of 4 interrelated decay chains. Since the TASC transport model does not differentiate between parent nuclides and their daughters, all of the actinides are treated alike with regard to their retardation factors.

For example, $^{239}_{\text{Pu}}$ is the daughter of $^{243}_{\text{Am}}$. Although there is evidence that Am migrates faster than Pu (Fried, et al., 1977; Hajek, 1966; Hajek and Knoll, 1966), in TASC's transport model both Pu and Am are treated alike. This is not unreasonable considering that the difference in retardation between them is probably less than the uncertainty in the combined estimated retardation factor of 10^4 (see below).

In the pH range of 5 to 8 for natural waters, the reported K_d 's for the actinides vary greatly (Table 5):

Americium	$2 \times 10^2 - 5 \times 10^4$
Neptunium	3.2×10^2

Plutonium	$2.5 \times 10^2 - 3.8 \times 10^5$
Thorium	$40 - >10^5$

Using the numbers above, an average K_d of 200-300 is a conservative estimate. From Figure 1, this gives a retardation factor of approximately 10^4 . Minimum and maximum K_f values of 10^2 and 10^5 represent possible K_d 's of 10 and 10^4 . Uncertainties as a function of time are estimated at $\pm$ an order of magnitude.

The literature suggests that the relative retardation factors could be:

$$\text{Th} \geq \text{Pu} > \text{Am} > \text{Np} > \text{Ra}^*$$

The actual differences between them will be a function of the repository characteristics (e.g., rock type, flow rates, groundwater composition, etc.). For the generic repository, treating the actinides as a group is justified. However, it should be remembered that Pu in particular may have a retardation factor as high as 10^7 -- an important consideration in modeling a repository for high-level waste from a throw-away cycle.

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*As the daughter of ^{230}Th , the alkaline-earth element, ^{226}Ra , is included with the actinides. There is evidence from the deep sea cores that Ra migrates relative to Th by a factor of 10 to 100. Reconcentration effects (see Burkholder, 1976) caused by these differences are currently being evaluated.

TABLE 5

ACTINIDE DISTRIBUTION COEFFICIENTS

<u>Reference</u>	$K_d \left(\frac{\text{ml}}{\text{g}} \right)$	<u>Conditions</u>
<u>Americium</u>		
Hajek & Knoll, 1966	1	sand, 5 N NO_3^- , oil, org., pH 3
	2×10^2	sand, 5 N NO_3^- , oil, org., pH 7
	5×10^2	sand, sludge, pH 7
Van Dalen, et al., 1975	5×10^4	illite/kaolinite, 90% sat. NaCl, pH 7-8
	4×10^2	river sand, 90% sat. NaCl, pH 7-8
Routson, et al., 1975	$>1.2 \times 10^3$	desert sand, .002 M Ca, pH 2.5-3.1
	$>1.2 \times 10^3$	desert sand, 0.2 M Ca, pH 2.5-3.1
	67	sandy clay, .002 M Ca, pH 2.5-3.1
	1	sandy clay, .2 M Ca, pH 2.5-3.1
	1.6	sandy clay, 3 M Na, pH 2.5-3.1
	280	sandy clay, .015 M Na, pH 2.5-3.1
Fried, et al., 1977	$K_f = 10^4$	limestone, pH 6.7, pH 2.5-3.1
<u>Neptunium</u>		
Dahlman, et al., 1975	3.2×10^2	clay soil, pH 6.5, 5 mM $\text{Ca}(\text{NO}_3)_2$
Routson, et al., 1975	2.37	sand, .002M Ca, pH 2.5-3.1
	0.36	sand, .2M Ca, pH 2.5-3.1
	3.9	sand, .015M Na, pH 2.5-3.1
	3.2	sand, pH 2.5-3.1, 3.0M Na
	0.25	sandy clay, pH 2.5-3.1, .002 M Ca

TABLE 5 (Continued)

<u>Reference</u>	$K_d \left(\frac{\text{ml}}{\text{g}} \right)$	<u>Conditions</u>
	0.16	sandy clay, pH 2.5-3.1, .2M Ca
	0.7	sandy clay, pH 2.5-3.1, .015 M Na
	0.4	sandy clay, pH 2.5-3.1, 3.0 M Na
<u>Plutonium</u>		
Bondietti, et al., 1975	$1-1.7 \times 10^5$	soil clays, pH 6.5, Pu(IV), 5 mM Ca
	7.5×10^4	soil clays, pH 6.5, Pu(VI), 5 mM Ca
	2.1×10^4	montmorillonite, pH 6.5, Pu(IV), 5 mM Ca
	2.5×10^2	montmorillonite, pH 6.5, Pu(VI), 5 mM Ca
Dahlman, et al., 1975	3×10^5	clay soil, pH 6.5, Pu(IV), 5 mM Ca
Dursuma and Parsi, 1974	$1-9 \times 10^4$	Mediterranean sediment
Glover, et al., 1976	35 to 1.4×10^4	17 soil samples, Pu(10^{-6} , 10^{-7} , 10^{-8} M)
Hajek, 1966	$K_f = 10^4$	surface soil, groundwater
Hetherington, et al., 1975	$2-4.8 \times 10^4$	5% clay, 50% silt, 45% sand
Miner, et al., 1974	$>4.3 \times 10^2$	soils
Mo and Lowman, 1975	$.16-3.8 \times 10^5$	clacareous sediment
Noshkin, et al., 1976	2.5×10^5	Ave. value for Enewitak groundwater particulates
Noshkin, 1977 unpub. data	1.1×10^5	coral soil and sediment
	8×10^4	clay silt particulates
Pillai & Mathew, 1976	9×10^4	Ave. value for suspended silt
Wahlgren, 1976	3×10^5	Lake Michigan particulates
Woodhead, 1976	5×10^4	sediments
Fried & Friedman, 1976	$K_f = 10^4$	tuff, Pu(IV), pH 6.7
	$K_f = 3 \times 10^2$	tuff, Pu(VI), pH 6.7

TABLE 5 (Continued)

<u>Reference</u>	<u>$K_d \left(\frac{\text{ml}}{\text{g}} \right)$</u>	<u>Conditions</u>
<u>Plutonium (Cont'd.)</u>		
Fried, et al., 1977	$K_f = 3 \times 10^4$	limestone, pH 6.7, Pu(IV)
	$K_f = 2 \times 10^4$	basalt "fissure", pH 6.7, Pu(IV)
<u>Thorium</u>		
Nishiwaki, et al., 1972	40-130	med. sand, pH 8.15
	310-470	v. f. sand, pH 8.15
	$2700-10^4$	silt/clay, pH 8.15
Rancon, 1973	8	schist soil, pH 3.2, 1g/1 Th.
	60	schist soil, pH 3.2, 1g/1 Th.
	120	illite, pH 3.2, 1 g/1 Th.
	10^3	illite, pH 3.2, 1 g/1 Th.
	$>10^5$	illite, pH >6, . /g/1 Th.
Dahlman, et al., 1975	1.6×10^5	clay soil, pH 6.5, 5 mM $\text{Ca}(\text{NO}_3)_2$
Bondietti, et al., 1976	1.6×10^5	silt loam, pH 6.5, Ca sat. clay
	4×10^5	montmorillonite, pH 6.5, Ca sat. clay
<u>Uranium</u>		
Rancon, 1975	3×10^2	clay soil, pH 5.5, 1ppm UO_2^{+2}
	2×10^3	clay soil, pH 10, 1 ppm UO_2^{+2}
	2.7×10^2	clay soil, pH 12, 1 ppm UO_2^{+2}
Dahlman, et al., 1975	4.4×10^3	clay soil, pH 6.5, U(VI), 5 mM $\text{Ca}(\text{NO}_3)_2$
Bondietti, et al., 1975	6.2×10^4	silt loam, pH 6.5, U(VI), Ca sat.

Future Work

For the purposes of examining the geochemical barrier to radionuclide migration, the estimated retardation factors described above are adequate for TASC's model. However, future improvements to the transport model will require more selective grouping of the radionuclides based on their retardation behavior. Instead of 3 groups we may want 6, or more, depending on how the actinide decay chains are handled. These refinements will be severely limited by the available data.

Our basic knowledge of waste/water/rock interactions is inadequate. The affect of the geochemical environment on retardation can be estimated -- but with much uncertainty. Assumptions need testing. For example, the assumption that ion exchange is zero in shale beds above a salt repository should be questioned. The decision on whether a salt repository can offer acceptable protection may depend on the answer.

Recommendations for research currently needed to answer questions related to ion exchange and retardation include:

1. An evaluation of K_d measurement techniques; i.e., static vs dynamic.
2. The effect on radionuclide retardation by changes in Eh, pH, groundwater composition and rock type.
3. A study of the rates of ion exchange and the question of flow rate vs equilibrium time.

The available data on distribution coefficients for the biologically important radionuclides present in high level-waste were used to estimate retardation factors (K_f) for a mass transport hydrologic model. The radionuclides were divided into 3 groups: fission products with no sorption ($K_f = 1$), fission products with sorption ($K_f = 10^2$), and the actinides and their daughter products ($K_f = 10^4$). Minimum and maximum values were assigned the latter two groups. Uncertainties as a function of time were estimated at $\pm$ an order of magnitude.

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