

**APPENDICES**

## **APPENDIX A**

### **Acronyms, Abbreviations, Symbols, and Notation**

## Appendix A

### Acronyms, Abbreviations, Symbols, and Notation

#### A.1.0 Acronyms And Abbreviations

AA	Atomic absorption
ASCII	American Standard Code for Information Interchange
ASTM	American Society for Testing and Materials
CCM	Constant capacitance (adsorption) model
CDTA	Trans-1,2-diaminocyclohexane tetra-acetic acid
CEAM	Center for Exposure Assessment Modeling at EPA's Environmental Research Laboratory in Athens, Georgia
CEC	Cation exchange capacity
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
DLM	Diffuse (double) layer (adsorption) model
DDL	Diffuse double layer (adsorption) model
DOE	U.S. Department of Energy
DTPA	Diethylenetriaminepentacetic acid
EDTA	Ethylenediaminetriacetic acid
EDX	Energy dispersive x-ray analysis
EPA	U.S. Environmental Protection Agency
EPRI	Electric Power Research Institute
HEDTA	N-(2-hydroxyethyl) ethylenedinitrilotriacetic acid
HLW	High level radioactive waste
IAEA	International Atomic Energy Agency
ICP	Inductively coupled plasma
ICP/MS	Inductively coupled plasma/mass spectroscopy
IEP (or iep)	Isoelectric point
LLNL	Lawrence Livermore National Laboratory, U.S. DOE
LLW	Low level radioactive waste
MCL	Maximum Contaminant Level
MEPAS	Multimedia Environmental Pollutant Assessment System
MS-DOS®	Microsoft® disk operating system (Microsoft and MS-DOS are register trademarks of Microsoft Corporation.)
NPL	Superfund National Priorities List
NRC	U.S. Nuclear Regulatory Commission
NWWA	National Water Well Association
OERR	Office of Remedial and Emergency Response, U.S. EPA
ORIA	Office of Radiation and Indoor Air, U.S. EPA
OSWER	Office of Solid Waste and Emergency Response, U.S. EPA

PC	Personal computers operating under the MS-DOS® and Microsoft® Windows operating systems (Microsoft® Windows is a trademark of Microsoft Corporation.)
PNL	Pacific Northwest Laboratory. In 1995, DOE formally changed the name of the Pacific Northwest Laboratory to the Pacific Northwest National Laboratory.
PNNL	Pacific Northwest National Laboratory, U.S. DOE
PZC	Point of zero charge
RCRA	Resource Conservation and Recovery Act
SCM	Surface complexation model
SDMP	NRC's Site Decommissioning Management Plan
TDS	Total dissolved solids
TLM	Triple-layer adsorption model
UK	United Kingdom (UK)
UK DoE	United Kingdom Department of the Environment
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation

## A.2.0 List of Symbols for the Elements and Corresponding Names

Symbol	Element	Symbol	Element	Symbol	Element
Ac	Actinium	Gd	Gadolinium	Po	Polonium
Ag	Silver	Ge	Germanium	Pr	Praseodymium
Al	Aluminum	H	Hydrogen	Pt	Platinum
Am	Americium	He	Helium	Pu	Plutonium
Ar	Argon	Hf	Hafnium	Ra	Radium
As	Arsenic	Hg	Mercury	Rb	Rubidium
At	Astatine	Ho	Holmium	Re	Rhenium
Au	Gold	I	Iodine	Rh	Rhodium
B	Boron	In	Indium	Rn	Radon
Ba	Barium	Ir	Iridium	Ru	Ruthenium
Be	Beryllium	K	Potassium	S	Sulfur
Bi	Bismuth	Kr	Krypton	Sb	Antimony
Bk	Berkelium	La	Lanthanum	Sc	Scandium
Br	Bromine	Li	Lithium	Se	Selenium
C	Carbon	Lu	Lutetium	Si	Silicon
Ca	Calcium	Lw	Lawrencium	Sm	Samarium
Cb	Columbium	Md	Mendelevium	Sn	Tin
Cd	Cadmium	Mg	Magnesium	Sr	Strontium
Ce	Cerium	Mn	Manganese	Ta	Tantalum
Cf	Californium	Mo	Molybdenum	Tb	Terbium
Cl	Chlorine	N	Nitrogen	Tc	Technetium
Cm	Curium	Na	Sodium	Te	Tellurium
Co	Cobalt	Nb	Niobium	Th	Thorium
Cr	Chromium	Nd	Neodymium	Ti	Titanium
Cs	Cesium	Ne	Neon	Tl	Thallium
Cu	Copper	Ni	Nickel	Tm	Thulium
Dy	Dysprosium	No	Nobelium	U	Uranium
Er	Erbium	Np	Neptunium	V	Vanadium
Es	Einsteinium	O	Oxygen	W	Tungsten
Eu	Europium	Os	Osmium	W	Wolfram
F	Fluorine	P	Phosphorus	Xe	Xenon
Fe	Iron	Pa	Protactinium	Y	Yttrium
Fm	Fermium	Pb	Lead	Yb	Ytterbium
Fr	Francium	Pd	Palladium	Zn	Zinc
Ga	Gallium	Pm	Promethium	Zr	Zirconium

### A.3.0 List of Symbols and Notation

$\rho_b$	Porous media bulk density (mass/length <sup>3</sup> )
Å	Angstrom, 10 <sup>-10</sup> meters
ads	Adsorption or adsorbed
$A_i$	Concentration of adsorbate (or species) I on the solid phase at equilibrium
am	Amorphous
aq	Aqueous
CEC	Cation exchange capacity
Ci	Curie
d	Day
dpm	Disintegrations per minute
e <sup>-</sup>	Free electron
Eh	Redox potential of an aqueous system relative to the standard hydrogen electrode
F	Faraday constant, 23,060.9 cal/V·mol
g	Gram
<sup>3</sup> H	Tritium
h	Hour
I	Ionic strength
IAP	Ion activity product
IEP	Isoelectric point
$K_d$	Concentration-based partition (or distribution) coefficient
$K_{r,298}$	Equilibrium constant at 298 K
$K_{r,T}$	Equilibrium constant at temperature T
l	Liter
M	Molar
m	Meter
mCi	Millicurie, 10 <sup>-3</sup> Curies
meq	Milliequivalent
mi	Mile
ml	Milliliter
mol	Mole
mV	Millivolt
N	Constant in the Freundlich isotherm model
n	Total porosity
$n_e$	Effective porosity
pCi	Picocurie, 10 <sup>-12</sup> Curies
pE	Negative common logarithm of the free-electron activity
pH	Negative logarithm of the hydrogen ion activity
pH <sub>zpc</sub>	pH for zero point of charge
ppm	Parts per million
R	Ideal gas constant, 1.9872 cal/mol·K

$R_f$	Retardation factor
s	Solid phase species
sec	Second
SI	Saturation index, as defined by $\log(IAP/K_{r,T})$
T	Absolute temperature, usually in Kelvin unless otherwise specified
t	Time
$t_{1/2}$	Half life
TDS	Total dissolved solids
TU	Tritium unit which is equivalent to 1 atom of $^3\text{H}$ (tritium) per $10^{18}$ atoms of $^1\text{H}$ (protium)
$v_c$	Velocity of contaminant through a control volume
$v_p$	Velocity of the water through a control volume
y	Year
Z	Valence state
z	Charge of ion
{ }	Activity
[ ]	Concentration

## **APPENDIX B**

### **Definitions**

## Appendix B

### Definitions

**Adsorption** - partitioning of a dissolved species onto a solid surface.

**Adsorption Edge** - the pH range where solute adsorption sharply changes from ~10% to ~90%.

**Actinon** - name occasionally used, especially in older documents, to refer to  $^{219}\text{Rn}$  which forms from the decay of actinium.

**Activity** - the effective concentration on an ion that determines its behavior to other ions with which it might react. An activity of ion is equal to its concentration only in infinitely dilute solutions. The activity of an ion is related to its analytical concentration by an activity coefficient,  $\gamma$ .

**Alkali Metals** - elements in the 1A Group in the periodic chart. These elements include lithium, sodium, potassium, rubidium, cesium, and francium.

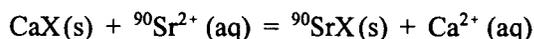
**Alpha Particle** - particle emitted from nucleus of atom during 1 type of radioactive decay. Particle is positively charged and has 2 protons and 2 neutrons. Particle is physically identical to the nucleus of the  $^4\text{He}$  atom (Bates and Jackson 1980).

**Alpha Recoil** - displacement of an atom from its structural position, as in a mineral, resulting from radioactive decay of the release an alpha particle from its parent isotope (*e.g.*, alpha decay of  $^{222}\text{Rn}$  from  $^{226}\text{Ra}$ ).

**Amphoteric Behavior** - the ability of the aqueous complex or solid material to have a negative, neutral, or positive charge.

**Basis Species** - see component species.

**Cation Exchange** - reversible adsorption reaction in which an aqueous species exchanges with an adsorbed species. Cation exchange reactions are



approximately stoichiometric and can be written, for example, as where X designates an exchange surface site.

**Cation Exchange Capacity (CEC)** - the sum total of exchangeable cations per unit mass of soil/sediment that a soil can adsorb.

**Clay Content** - particle size fraction of soil that is less than 2  $\mu\text{m}$  (unless specified otherwise).

**Code Verification** - test of the accuracy with which the subroutines of the computer code perform the numerical calculations.

**Colloid** - any fine-grained material, sometimes limited to the particle-size range of  $<0.00024$  mm (*i.e.*, smaller than clay size), that can be easily suspended. In its original sense, the definition of a colloid included any fine-grained material that does not occur in crystalline form.

**Complexation (Complex Formation)** - any combination of dissolved cations with molecules or anions containing free pairs of electrons.

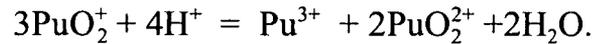
**Component Species** - "basis entities or building blocks from which all species in the system can be built" (Allison *et al.*, 1991). They are a set of linearly independent aqueous species in terms of which all aqueous speciation, redox, mineral, and gaseous solubility reactions in the MINTEQA2 thermodynamic database are written.

**Detrital Mineral** - "any mineral grain resulting from mechanical disintegration of parent rock" (Bates and Jackson 1980).

**Deuterium (D)** - stable isotopes  ${}^2\text{H}$  of hydrogen.

**Disproportionation** - is a chemical reaction in which a single compound serves as both oxidizing and reducing agent and is thereby converted into more oxidized and a more reduced derivatives (Sax and Lewis 1987). For the reaction to occur, conditions in the system must be temporarily changed to favor this

reaction (specifically, the primary energy barrier to the reaction must be lowered). This is accomplished by a number of ways, such as adding heat or microbes, or by radiolysis occurring. Examples of plutonium disproportionation reactions are:



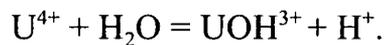
**Electron Activity** - unity for the standard hydrogen electrode.

**Far Field** - the portion of a contaminant plume that is far from the point source and whose chemical composition is not significantly different from that of the uncontaminated portion of the aquifer.

**Fulvic Acids** - breakdown products of cellulose from vascular plants (also see humic acids). Fulvic acids are the alkaline-soluble portion which remains in solution at low pH and is of lower molecular weight (Gascoyne 1982).

**Humic Acids** - breakdown products of cellulose from vascular plants (also see fulvic acids). Humic acids are defined as the alkaline-soluble portion of the organic material (humus) which precipitates from solution at low pH and are generally of high molecular weight (Gascoyne 1982).

**Hydrolysis** - a chemical reaction in which a substance reacts with water to form 2 or more new substances. For example, the first hydrolysis reaction of  $\text{U}^{4+}$  can be written as



**Hydrolytic Species** - an aqueous species formed from a hydrolysis reaction.

**Ionic Potential** - ratio ( $z/r$ ) of the formal charge ( $z$ ) to the ionic radius ( $r$ ) of an ion.

**Isoelectric Point (iep)** - pH at which a mineral's surface has a net surface charge of zero. More precisely, it is the pH at which the particle is electrokinetically uncharged.

**Lignite** - a coal that is intermediate in coalification between peat and

subbituminous coal.

**Marl** - an earthy substance containing 35-65% clay and 65-35% carbonate formed under marine or freshwater conditions

**Mass Transfer** - transfer of mass between 2 or more phases that includes an aqueous solution, such as the mass change resulting from the precipitation of a mineral or adsorption of a metal on a mineral surface.

**Mass Transport** - time-dependent movement of 1 or more solutes during fluid flow.

**Mire** - a small piece of marshy, swampy, or boggy ground.

**Model Validation** - integrated test of the accuracy with which a geochemical model and its thermodynamic database simulate actual chemical processes.

**Monomeric Species** - an aqueous species containing only 1 center cation (as compared to a polymeric species).

**Near Field** - the portion of a contaminant plume that is near the point source and whose chemical composition is significantly different from that of the uncontaminated portion of the aquifer.

**Peat** - an unconsolidated deposit of semicarbonized plant remains in a water saturated environment.

**Polynuclear Species** - an aqueous species containing more than 1 central cation moiety, *e.g.*,  $(\text{UO}_2)_2\text{CO}_3(\text{OH})_3^-$  and  $\text{Pb}_4(\text{OH})_4^{4+}$ .

**Protium (H)** - stable isotope  $^1\text{H}$  of hydrogen.

**Retrograde Solubility** - solubility that decreases with increasing temperature, such as those of calcite ( $\text{CaCO}_3$ ) and radon. The solubility of most compounds (*e.g.*, salt,  $\text{NaCl}$ ) increases with increasing temperature.

**Species** - actual form in which a dissolved molecule or ion is present in solution.

**Specific Adsorption** - surface complexation via a strong bond to a mineral surface.

For example, several transition metals and actinides are specifically adsorbed to aluminum- and iron-oxide minerals.

**Sol** - a homogeneous suspension or dispersion of colloidal matter in a fluid.

**Solid Solution** - a solid material in which a minor element is substituted for a major element in a mineral structure.

**Thoron** - name occasionally used, especially in older documents, to refer to  $^{220}\text{Rn}$  which forms from the decay of thorium.

**Tritium (T)** - radioactive isotope  $^3\text{H}$  of hydrogen.

**Tritium Units** - units sometimes used to report tritium concentrations. A tritium unit (TU) is equivalent to 1 atom of  $^3\text{H}$  (tritium) per  $10^{18}$  atoms of  $^1\text{H}$  (protium). In natural water that produces  $7.2 \times 10^{-3}$  disintegrations per minute per milliliter (dpm/ml) of tritium, 1 TU is approximately equal to 3.2 picocuries/milliliter (pCi/ml).

**APPENDIX C**

**Partition Coefficients For Cadmium**

## Appendix C

### Partition Coefficients For Cadmium

#### C.1.0 Background

Cadmium  $K_d$  values and some important ancillary parameters that have been shown to influence cadmium sorption were collected from the literature and tabulated. Data included in this data set were from studies that reported  $K_d$  values and were conducted in systems consisting of

- Natural soils (as opposed to pure mineral phases)
- Low ionic strength solutions (<0.1 M)
- pH values between 4 and 10
- Solution cadmium concentration less than 10<sup>-5</sup> M
- Low humic materials concentrations (<5 mg/l)
- No organic chelates (such as EDTA)

A total of 174 cadmium  $K_d$  values were found in the literature (see summary in Section C.3.0). At the start of the literature search, attempts were made to identify  $K_d$  studies that included ancillary data on aluminum/iron-oxide concentrations, calcium and magnesium solution concentrations, pH, cation exchange capacity (CEC), clay content, redox status, organic matter concentrations and sulfide concentrations. Upon reviewing the data and determining the availability of cadmium  $K_d$  measurements having ancillary information,  $K_d$  values were collected that included information on clay content, pH, CEC, total organic carbon (related to organic matter), and dissolved cadmium concentrations. The selection of these parameters was based on availability of data and the possibility that the parameter may impact cadmium  $K_d$  values. Of the 174 cadmium  $K_d$  values included in our tabulation, 62 values had associated clay content data, 174 values had associated pH data, 22 values had associated CEC data, 63 values had total organic carbon data, 172 values had associated cadmium concentration data, and 16 had associated aluminum/iron-oxide data. The descriptive statistics for this total set of cadmium  $K_d$  values are listed in Table C.1.

**Table C.1.** Descriptive statistics of the cadmium  $K_d$  data set for soils.

	Cadmium $K_d$ (ml/g)	Clay Content (wt.%)	pH	CEC (meq/100g)	TOC (mg/l)	Cd Conc. (mg/l)	Fe Oxides (wt.%)
Mean	226.7	14.2	5.88	21	5.5	3.67	1.32
Standard Error	44.5	1.7	0.09	3	0.85	0.48	0.53
Median	121.8	10.24	5.83	23	2.0	0.01	0.38
Mode	80.0	6	6.8	2	0.4	0.01	0.19
Std. Dev	586.6	13.5	1.16	15	6.8	6.27	2.12
Sample Variance	344086	182	1.34	245	45.9	39.4	4.51
Range	4359	86.2	6.20	58	32.4	34.9	8.28
Minimum	0.50	.9	3	2	0.2	0.01	0.01
Maximum	4360	87.1	9.2	60	32.6	35	8.29
No. Samples	174	62	174	22	63	172	16

## C.2.0 Approach and Regression Models

### C.2.1 Correlations with Cadmium $K_d$ Values

Linear regression analyses were conducted between the ancillary parameters and cadmium  $K_d$  values. The correlation coefficients from these analyses are presented in Table C.2. These results were used for guidance for selecting appropriate independent variables to use in the look-up table. The largest correlation coefficient was between pH and  $\log(K_d)$ . This value is significant at the 0.001 level of probability. Attempts at improving this correlation coefficient through the use of additional variables, *i.e.*, using multiple-regression analysis, were not successful. Multiple regression analyses were conducted with the following pairs of variables to predict cadmium  $K_d$  values: total organic carbon and pH, clay content and pH, total organic carbon and iron-oxides, and pH and CEC.

**Table C.2.** Correlation coefficients (r) of the cadmium  $K_d$  data set for soils.

	Cadmium $K_d$	log ( $K_d$ )	Clay Content	pH	CEC	TOC	Cd Conc.
Cadmium $K_d$	1						
log ( $K_d$ )	0.69	1					
Clay Conc.	-0.04	0.03	1				
pH	0.50	0.75	0.06	1			
CEC	0.40	0.41	0.62	0.35	1		
TOC	0.20	0.06	0.13	-0.39	0.27	1	
Cd Conc.	-0.02	-0.10	-0.39	0.22	-0.03	-0.09	1
Fe Oxide Conc.	0.18	0.11	-0.06	0.16	0.19	0.18	0.01

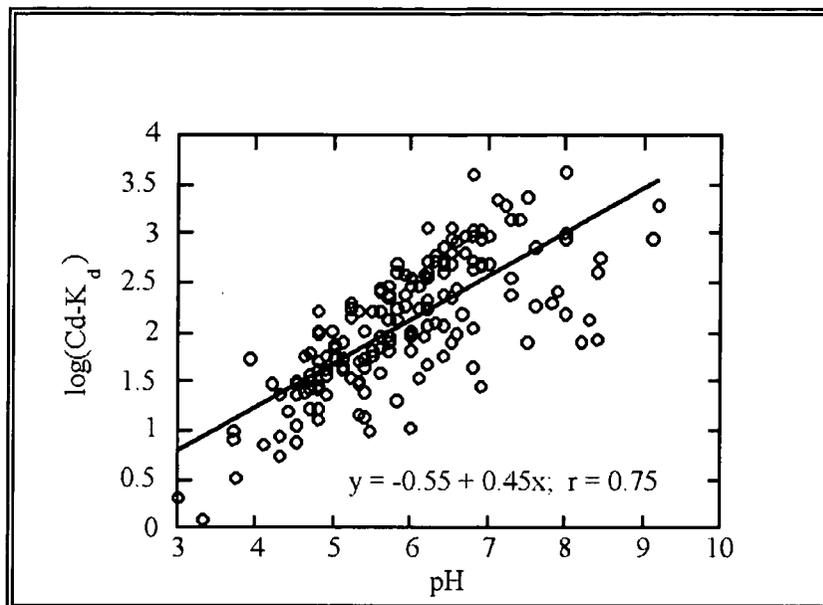
### *C.2.2 Cadmium $K_d$ Values as a Function of pH*

The cadmium  $K_d$  values plotted as a function of pH are presented in Figure C.1. A large amount of scatter exists in these data. At any given pH, the range of  $K_d$  values may vary by 2 orders of magnitude. This is not entirely satisfactory, but as explained above, using more than 1 variable to help categorize the cadmium  $K_d$  values was not fruitful.

The look-up table (Table C.3) for cadmium  $K_d$  values was categorized by pH. The regression equation for the line presented in Figure C.1 is:

$$\text{Cd } K_d = -0.54 + 0.45(\text{pH}). \quad (\text{C.1})$$

The minimum and maximum values were estimated based on the scatter of data points observed in Figure C.1.



**Figure C.1.** Relation between cadmium  $K_d$  values and pH in soils.

**Table C.3.** Look-up table for estimated range of  $K_d$  values for cadmium based on pH. [Tabulated values pertain to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength (< 0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.]

$K_d$ (ml/g)	pH		
	3 - 5	5 - 8	8 - 10
Minimum	1	8	50
Maximum	130	4,000	12,600

### C.3.0 Data Set for Soils

Table C.4 lists the available  $K_d$  values for cadmium identified for experiments conducted with only soils. The  $K_d$  values are listed with ancillary parameters that included clay content, pH, CEC, TOC, solution cadmium concentrations, and iron-oxide concentrations

**Table C.4.** Cadmium  $K_d$  data set for soils.

Cd $K_d$ (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
52.5	54.7	4.8	30.2	1.54	1	0.33	0.005 M CaNO <sub>3</sub>	Alligator Ap	Converted Freund. to $K_d$ Using 1ppm	1
288.4	8.3	5.7	2	0.61	1	0.1	0.005 M CaNO <sub>3</sub>	Cecil Ap	Converted Freund. to $K_d$ Using 1ppm	1
13.9	51.2	5.4	2.4	0.26	1	0.08	0.005 M CaNO <sub>3</sub>	Cecil B	Converted Freund. to $K_d$ Using 1ppm	1
186.6	0.9	5.9	22.54	6.62	1	1.68	0.005 M CaNO <sub>3</sub>	Kula Ap1	Converted Freund. to $K_d$ Using 1ppm	1
52.7	17.6	3.9	26.9	11.6	1	1.19	0.005 M CaNO <sub>3</sub>	Lafite Ap	Converted Freund. to $K_d$ Using 1ppm	1
91.2	28.2	6	11	1.67	1	0.19	0.005 M CaNO <sub>3</sub>	Molokai Ap	Converted Freund. to $K_d$ Using 1ppm	1
28.8	2.8	6.9	4.1	0.21	1	0.06	0.005 M CaNO <sub>3</sub>	Norwood Ap	Converted Freund. to $K_d$ Using 1ppm	1
97.9	6.2	6.6	8.6	0.83	1	0.3	0.005 M CaNO <sub>3</sub>	Olivier Ap	Converted Freund. to $K_d$ Using 1ppm	1
5.5	3.8	4.3	2.7	1.98	1	0	0.005 M CaNO <sub>3</sub>	Spodisol	Converted Freund. to $K_d$ Using 1ppm	1
755.1	23.9	7.6	48.1	4.39	1	0.19	0.005 M CaNO <sub>3</sub>	Webster Ap	Converted Freund. to $K_d$ Using 1ppm	1

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
14.4	2.8	5.3	2	2.03	1	0.42	0.005 M CaNO <sub>3</sub>	Windsor Ap	Converted Freund. to K <sub>d</sub> Using 1ppm	1
87.1		8.4	60	1.44	1	1.07	Water	Vertic Torrifluvent	Converted Freund. to K <sub>d</sub> Using 1ppm	2
33.88		5.2	33.8	32.6	1		Water	Organic	Converted Freund. to K <sub>d</sub> Using 1ppm	2
20.42		5.8	23.8	3	1	8.29	Water	Boomer, Ultic Haploxeralf	Converted Freund. to K <sub>d</sub> Using 1ppm	2
10.47		6	25	3.2	1	1.07	Water	UlticPalexeralf	Converted Freund. to K <sub>d</sub> Using 1ppm	2
80		8.2	8.2	0.21	35		0.01 M NaCl	Gevulot	Calc. Fig 1.	3
200		7.8	15.4	0.83	25		0.01 M NaCl	Bet Yizhaq	Calc. Fig 1.	3
133.3		8.3	18.9	0.23	30		0.01 M NaCl	Gilat	Calc. Fig 1.	3
181.8		7.6	31.8	0.79	25		0.01 M NaCl	Maaban Michael	Calc. Fig 1.	3
266.7		7.9	37	0.86	15		0.01 M NaCl	Hahoterim	Calc. Fig 1.	3
8	8	3.7		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4
17	8	4.8		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4
32	8	5.3		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4
64	8	6		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4
92	8	6.2		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4
110	8	6.8		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4
250	8	7.3		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
580	8	8.5		1.6	11.2		0.01 M NaNO <sub>3</sub>	Downer Loamy Sand		4
0.5	6	3.1		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
3.3	6	3.8		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
7.5	6	4.5		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
10	6	5.5		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
34	6	6.1		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
45	6	6.8		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
80	6	7.5		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
150	6	8		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
420	6	8.4		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
900	6	9.1		0.4	11.2		0.01 M NaNO <sub>3</sub>	Freehold Sandy Loam A Horizon		4
2.1	13	3		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
10	13	3.7		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
30	13	4.2		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
57	13	4.6		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.#
101	13	5		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
195	13	5.2		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
420	13	5.8		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
1,200	13	6.2		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
4,000	13	6.8		16.8	11.2		0.01 M NaNO <sub>3</sub>	Boonton Loam		4
1.2	16	3.3		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
7.1	16	4.1		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
27	16	4.8		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
53	16	5.1		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
170	16	5.6		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
300	16	6.1		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
390	16	6.2		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
910	16	6.5		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
1,070	16	6.8		9.8	11.2		0.01 M NaNO <sub>3</sub>	Rockaway Stony Loam		4
43	10	4.8		2.4	11.2		0.01 M NaNO <sub>3</sub>	Fill Material - Delaware River		4
67	10	5.7		2.4	11.2		0.01 M NaNO <sub>3</sub>	Fill Material - Delaware River		4
130	10	6.3		2.4	11.2		0.01 M NaNO <sub>3</sub>	Fill Material - Delaware River		4

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
150	10	6.7		2.4	11.2		0.01 M NaNO <sub>3</sub>	Fill Material - Delaware River		4
370	10	7.3		2.4	11.2		0.01 M NaNO <sub>3</sub>	Fill Material - Delaware River		4
880	10	8		2.4	11.2		0.01 M NaNO <sub>3</sub>	Fill Material - Delaware River		4
1,950	10	9.2		2.4	11.2		0.01 M NaNO <sub>3</sub>	Fill Material - Delaware River		4
1,000	12	8			1	3.7	Carbonate Groundwater r	Interbed	pH of Groundwater	5
4,360	12.4	8			1	2.5	Carbonate Groundwater r	Alluvium	pH of Groundwater	5
536.8	25.2	6.8	27.5				0.01 M NaCl	Soil A	Desorption	6
440	25.2	6.8	27.5				0.01 M NaCl	Soil A	Desorption	6
9		4.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
23.4		4.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
15.8		4.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
11.3		4.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
31.2		4.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
32.5		4.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
23		4.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
17.1		4.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
13.1		4.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
24.9		4.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
26.8		4.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
36.2		4.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
32.9		4.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
37.2		4.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
29.2		4.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
28.3		4.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
22.6		4.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
37.4		4.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
40.9		4.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
63.5		4.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
25.2		5.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
29.9		5.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
33.7		5.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
44.3		5.1			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
42.8		5.1			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
53.5		5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
56.2		4.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
68.7		5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
82.3		5.1			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
75.7		5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
95.2		4.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
103		4.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
160		4.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
43.3		5.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
55.2		5.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
52.2		5.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
40.3		5.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
56.1		5.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
67.5		5.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
102.9		5.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
164.4		5.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
163.8		5.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
202.1		5.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
172.4		5.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
149		5.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
72.8		5.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
81.6		5.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.†
90		5.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
94.3		5.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
48.1		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
56.5		6.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
81		6.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
122.3		6.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
121.4		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
101.5		6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
99.3		6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
107.8		6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
219.5		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
179.2		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
177		6.1			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
360.4		6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
305.2		6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
236.8		5.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
186.3		5.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
174.8		5.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
138.7		5.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
132.5		5.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
375.6		5.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
403.3		5.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
510.8		5.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
225.9		5.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
227.3		5.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
248		5.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
253.1		5.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
277.2		5.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
240.7		6.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
227.8		6.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
281.1		6.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
551.2		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
519.8		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
418.7		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
353.7		6.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
400.8		6.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
609.2		6.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
545.7		6.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.†
515.9		6.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
545.7		6.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
760.9		6.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
665.7		6.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
503.2		6.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
515.2		7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
488.9		6.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
481		6.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
461.6		6.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
1,151		6.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
868.7		6.6			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
637.2		6.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
970.9		6.7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
950.5		6.8			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
886.2		6.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
1,106		6.9			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
970.9		7			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
2,248		7.1			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
1,909		7.2			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7

Cd K <sub>d</sub> (ml/g)	Clay Cont. (wt%)	pH	CEC (meq/ 100 g)	TOC (wt%)	[Cd] (mg/l)	Fe Oxides (wt.%)	Solution	Soil Identification	Comments	Ref.*
1,411		7.3			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
1,383		7.4			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7
2,337		7.5			0.01		0.001M CaCl <sub>2</sub>	Agricultural Danish Soil	Co = 0.7 to 12.6 ppb	7

a 1 = Buchter *et al.*, 1989; 2 = Garcia-Miragaya, 1980; 3 = Navrot *et al.*, 1978; 4 = Allen *et al.*, 1995; 5 = Del Debbio, 1991; 6 = Madrid *et al.*, 1992; 7 = Anderson and Christensen, 1988

#### C.4.0 References

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## **APPENDIX D**

### **Partition Coefficients For Cesium**

## Appendix D

### Partition Coefficients For Cesium

#### D.1.0 Background

Three generalized, simplifying assumptions were established for the selection of cesium  $K_d$  values for the look-up table. These assumptions were based on the findings of the literature reviewed we conducted on the geochemical processes affecting cesium sorption. The assumptions are as follows:

- Cesium adsorption occurs entirely by cation exchange, except when mica-like minerals are present. Cation exchange capacity (CEC), a parameter that is frequently not measured, can be estimated by an empirical relationship with clay content and pH.
- Cesium adsorption onto mica-like minerals occurs much more readily than desorption. Thus,  $K_d$  values, which are essentially always derived from adsorption studies, will greatly overestimate the degree to which cesium will desorb from these surfaces.
- Cesium concentrations in groundwater plumes are low enough, less than approximately  $10^{-7}$  M, such that cesium adsorption follows a linear isotherm.

These assumptions appear to be reasonable for a wide range of environmental conditions. However, these simplifying assumptions are clearly compromised in systems with cesium concentrations greater than approximately  $10^{-7}$  M, ionic strengths greater than about 0.1 M, and pH values greater than about 10.5. These assumptions will be discussed in more detail in the following sections.

Based on the assumptions and limitation described above, cesium  $K_d$  values and some important ancillary parameters that influence cation exchange were collected from the literature and tabulated. Data included in this table were from studies that reported  $K_d$  values (not percent adsorbed or Freundlich or Langmuir constants) and were conducted in systems consisting of:

- Low ionic strength ( $< 0.1$  M)
- pH values between 4 and 10.5
- Dissolved cesium concentrations less than  $10^{-7}$  M
- Low humic material concentrations ( $< 5$  mg/l)
- No organic chelates (*e.g.*, EDTA)

The ancillary parameters included in these tables were clay content, mica content, pH, CEC, surface area, and solution cesium concentrations. This cesium data set included 176 cesium  $K_d$  values.

Two separate data sets were compiled. The first one (see Section D.3) included both soils and pure mineral phases. The lowest cesium  $K_d$  value was 0.6 ml/g for a measurement made on a system containing a soil consisting primarily of quartz, kaolinite, and dolomite and an aqueous phase consisting of groundwater with a relatively high ionic strength ( $I \approx 0.1$  M) (Lieser *et al.*, 1986) (Table D.1). The value is unexplainably much less than most other cesium  $K_d$  values present in the data set. The largest cesium  $K_d$  values was 52,000 ml/g for a measurement made on a pure vermiculite solid phase (Tamura, 1972). The average cesium  $K_d$  value was  $2635 \pm 530$  ml/g.

**Table D.1.** Descriptive statistics of cesium  $K_d$  data set including soil and pure mineral phases. [Data set is presented in Section D.3.]

	$K_d$ (ml/g)	Clay (%)	Mica (%)	pH	CEC (meq/100 g)	Surface Area (m <sup>2</sup> /g)
Mean	2,635	30	5.5	7.4	30.4	141.3
Standard Error	530	3.8	0.7	0.1	3.7	29.7
Median	247	42	4	8.2	4.8	31.2
Mode	40	42	4	8.2	1.8	17.7
Standard Deviation	7055	15	4.4	1.7	37.4	230.4
Sample Variance	49,781,885	226	20.0	2.8	1,396.9	53,106
Range	51,999	38	13	7.8	129.9	638
Minimum	0.6	4	2	2.4	0.00098	8
Maximum	52,000	42	15	10.2	130	646
No. Observations	177	15	41	139	103	60
Confidence Level (95.0%)	1,046.6	8.3	1.4	0.3	7.3	59.5

A second data set (see Section D.4) was created using only data generated from soil studies, that is, data from pure mineral phases, and rocks, were eliminated from the data set. Descriptive statistics of the soil-only data set are presented in Table D.2. Perhaps the most important finding of this data set is the range and median<sup>1</sup> of the 57  $K_d$  values. Both statistics decreased appreciably. In the soil-only data set, the median was 89 ml/g. The median is perhaps the single central estimate of a cesium  $K_d$  value for this data set. The range of  $K_d$  values was from 7.1 ml/g, for a measurement made on a sandy carbonate soil (Routson *et al.*, 1980), to 7610 ml/g for a measurement made on another carbonate soil containing greater than 50 percent clay and silt (Serne *et al.*, 1993). Interestingly, these 2 soils were both collected from the U.S. Department of Energy's Hanford Site in eastern Washington state.

**Table D.2.** Descriptive statistics of data set including soils only. [Data set is presented in Section D.4.]

	Cesium $K_d$ (ml/g)	Clay (%)	Mica (%)	pH	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)
Mean	651	5	5.6	6.9	34	57.5
Standard Error	188	0.6	0.6	0.3	8.9	13.4
Median	89	5.0	4	6.7	20	60
Mode	22	NA	4	4.0	60	70
Standard Deviation	1423	1.0	4.3	1.9	29.5	44.6
Sample Variance	2026182	1.0	18.4	3.6	870	1986
Range	7602	2.0	13	7.8	57.4	123.4
Minimum	7.1	7.1	2	2.4	2.6	6.6
Maximum	7610	6.0	15	10.2	70.0	130
No. Observations	57	3	45	55	11	11
Confidence Level (95%)	378	2.5	1.29	0.5	19.8	30

<sup>1</sup> The median is that value for which 50 percent of the observations, when arranged in order of magnitude, lie on each side.

The soil-only data set was frequently incomplete with regard to supporting data describing the experimental conditions under which the cesium  $K_d$  values were measured (Table D.2). Quite often the properties of the solid phase or the dissolved cesium concentration used in the  $K_d$  experiments were not reported. For instance, there were only 3 cesium  $K_d$  values that had accompanying clay content data, 11 cesium  $K_d$  values that had accompanying cation exchange data, and 11 cesium  $K_d$  values that had accompanying surface area data (Table D.2). Consequently, it was not possible to evaluate adequately the relationship between cesium  $K_d$  values and these important, independent soil parameters. This is discussed in greater detail below.

## **D.2.0 Approach and Regression Models**

### ***D.2.1 Correlations with Cesium $K_d$ Values***

A matrix of the correlation coefficients for the parameters included in the data set containing  $K_d$  values determined in experiments with both soils and pure mineral phases is presented in Table D.3. The correlation coefficients that are significant at or less than the 5 percent level of probability ( $P \leq 0.05$ ) are identified with a footnote. The parameter with the largest correlation coefficient with cesium  $K_d$  was CEC ( $r = 0.52$ ). Also significant was the correlation coefficient between cesium  $K_d$  values and surface area ( $r = 0.42$ ) and CEC and clay content ( $r = 0.64$ ). The poor correlation between cesium aqueous concentration ( $[Cs]_{aq}$ ) and cesium  $K_d$  values can be attributed to the fact that the former parameter included concentration of the solution prior and after contact with the soils. We report both under the same heading, because the authors frequently neglected to indicate which they were reporting. More frequently, the spike concentration (the cesium concentration prior to contact with the soil) was reported, and this parameter by definition is not correlated to  $K_d$  values as well as the concentrations after contact with soil (the denominator of the  $K_d$  term).

A matrix of the correlation coefficients for the parameters included in the data set containing  $K_d$  values determined in experiments with only soils is presented in Table D.4. As mentioned above (Table D.2), the reports in which soil was used for the  $K_d$  measurements tended to have little supporting data about the aqueous and solid phases. Consequently, there was little information for which to base correlations. This occasionally resulted in correlations that were not scientifically meaningful. For example, the correlation between CEC and cesium  $K_d$  was  $-0.83$ , for only 11 observations (10 degrees of freedom). The negative sign of this correlation contradicts commonly accepted principles of surface chemistry.

**Table D.3.** Correlation coefficients (r) of the cesium  $K_d$  value data set that included soils and pure mineral phases. [Data set is presented in Section D.3.]

	Cesium $K_d$	Clay Content	Mica	pH	CEC	Surface Area
Cesium $K_d$	1.00					
Clay Content	0.05	1.00				
Mica	0.29	0.00	1.00			
pH	0.10	-0.11	0.08	1.00		
CEC	0.52 <sup>a</sup>	0.64 <sup>a</sup>	NA	0.37	1.00	
Surface Area	0.42 <sup>a</sup>	0.35	NA	-0.11	0.47 <sup>a</sup>	1.00
$[Cs]_{aq}$	-0.07	0.85 <sup>a</sup>	0.29	0.13	-0.17	-0.15

a Correlation coefficient is significant at the 5% level of significance ( $P \leq 0.05$ ).

**Table D.4.** Correlation coefficients (r) of the soil-only data set. [Data set is presented in Section D.4.]

	Cesium $K_d$	Clay Content	Mica	pH	CEC	Surface Area
Cesium $K_d$	1.00					
Clay Content	-0.21	1.00				
Mica	0.27	0	1.00			
pH	0.11	0.4	0.07	1.00		
CEC	-0.83	NA	0.99 <sup>1</sup>	0.05	1.00	
Surface Area	-0.31	NA	0.99 <sup>1</sup>	-0.03	0.37	1.00
$[Cs]_{aq}$	0.18	NA	0.09	-0.04	0.00	0

<sup>1</sup> Correlation coefficient is significant at >5% level of significance ( $P \leq 0.05$ ).

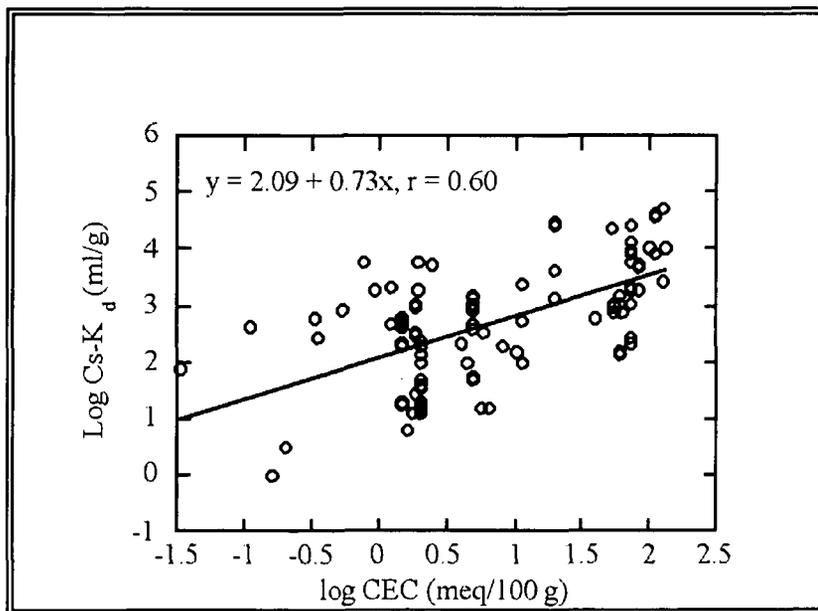
The high correlations between mica concentrations and CEC ( $r = 0.99$ ) and mica concentrations and surface area ( $r = 0.99$ ) are somewhat misleading in the fact that both correlations represent only 4 data points collected from 1 study site in Fontenay-aux-Roses in France (Legoux *et al.*, 1992).

### D.2.2 Cesium Adsorption as a Function of CEC and pH

Akiba and Hashimoto (1990) showed a strong correlation between cesium  $K_d$  values and the CEC of a large number of soils, minerals, and rock materials. The regression equation generated from their study was:

$$\log (Cs K_d) = 1.2 + 1.0 \log (CEC) \quad (D.1)$$

A similar regression analysis using the entire data set (mineral, rocks, and soils) is presented in Figure D.1.

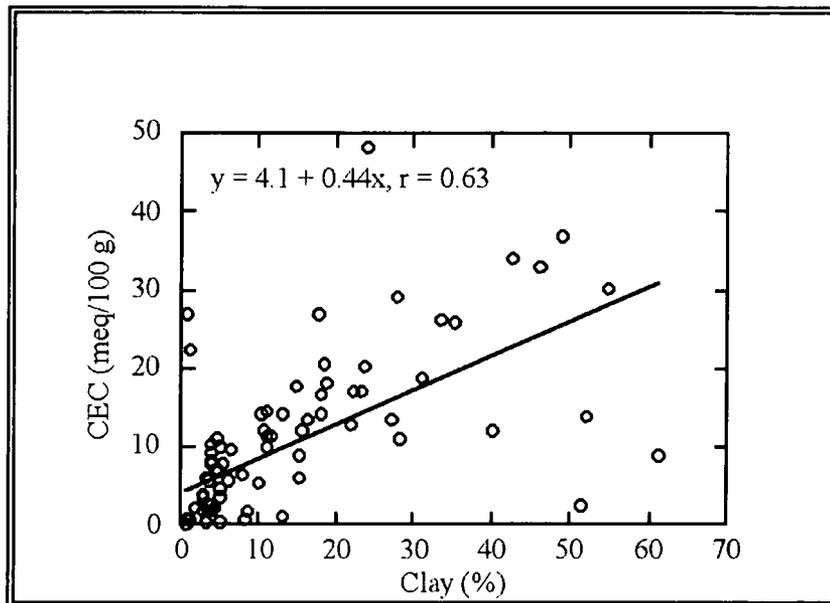


**Figure D.1.** Relation between cesium  $K_d$  values and CEC.

By transposing the CEC and cesium  $K_d$  data into logarithms, the regression correlation slightly increases from 0.52 (Table D.3) to 0.60 (Figure D.1). However, a great amount of scatter in the data can still be seen in the logarithmic transposed data. For instance, at  $\log(\text{CEC})$  of 0.25, the cesium  $K_d$  values range over 4 orders of magnitude. It is important to note that the entire cesium  $K_d$  data set only varies 5 orders of magnitude. Thus, the correlation with CEC, although the strongest of all the independent variables examined, did not reduce greatly the variability of possible cesium  $K_d$  values.

### D.2.3 CEC as a Function of Clay Content and pH

Because CEC values are not always available to contaminant transport modelers, an attempt was made to use independent variables more commonly available in the regression analysis. Multiple regression analysis was conducted using clay content and pH as independent variables to predict CEC values (Figure D.2). Clay content was highly correlated to CEC ( $r = 0.64$ ). Soil pH was not significantly correlated to either CEC or cesium  $K_d$  values.



**Figure D.2.** Relation between CEC and clay content.

#### D.2.4 Cesium Adsorption onto Mica-Like Minerals

Cesium adsorption onto mica-like minerals has long been recognized as a non-reversible reaction (Bruggenwert and Kamphorst, 1979; Comans *et al.*, 1989; Cremers *et al.*, 1988; Douglas, 1989; Evans *et al.*, 1983; Francis and Brinkley, 1976; Sawhney, 1972; Smith and Comans, 1996; Tamura, 1972). This is an important property in adsorption reactions because 1 of the assumptions in applying the  $K_d$  model to describe adsorption is that the rate at which adsorption occurs is equal to the rate at which desorption occurs. This phenomena is referred to as an adsorption hysteresis. Cesium adsorption onto mica-like minerals is appreciably faster than its desorption. The reason for this is that the cesium ion fits perfectly into the hexagonal ring formed on the tetrahedral sheet in the crystallographic structure of mica-like clays. This perfect fit does not permit other cations that exist at much greater concentrations in nature to exchange the cesium from these sites. This can be demonstrated using the data of Tamura (1972) (Table D.5). He measured cesium  $K_d$  values for mica, vermiculite, and kaolinite using a water and 0.1 M NaCl background solution. For mica, the  $K_d$  value remained about the same for both solutions. For the vermiculite and kaolinite, the cesium  $K_d$  values greatly decreased when the higher ionic strength solution was used. This indicates that the sodium, which existed at 11 orders of magnitude higher concentration than the cesium could out compete the adsorption of cesium on the vermiculite and kaolinite but not on the mica. Another point of interest regarding this data set is that the cesium  $K_d$  values do correlate with CEC of these different mineral phases when water is the background solution. However, when the higher ionic strength solution is used, the correlation with CEC no longer exists.

Comans *et al.* (1989) measured cesium  $K_d$  values of a mica (Fithian illite) by desorption and adsorption experiments. Portions of their data are presented in Table D.6. Cesium  $K_d$  values based on desorption experiments are appreciably greater than those measure in adsorption experiments.

**Table D.5.** Effect of mineralogy on cesium exchange. [Data are from Tamura (1972) who used an initial concentration of dissolved cesium of  $1.67 \times 10^{-12}$  M.]

Mineral Phases	CEC (meq/100 g)	$K_d$ in Water (ml/g)	$K_d$ in 0.1 M NaCl (ml/g)
Mica	20	26,000	28,600
Vermiculite	127	52,000	2,700
Kaolinite	11.2	2,500	94

**Table D.6.** Cesium  $K_d$  values measured on mica (Fithian illite) via adsorption and desorption experiments. [Data are from Comans *et al.* (1989).]

Experimental Conditions	Adsorption Cesium $K_d$	Desorption Cesium $K_d$
K-saturated Mica, $7 \times 10^{-6}$ M Cs	2,890	5,200
K-saturated Mica, $2 \times 10^{-7}$ M Cs	9,000	11,300
Ca-saturated Mica, $7 \times 10^{-6}$ M Cs	1,060	4,600
Ca-saturated Mica, $2 \times 10^{-7}$ M Cs	600,000	1,050,000

Essentially all  $K_d$  values reported in the literature are measured using adsorption experiments. Thus, in the case of soils containing mica-like soils, using adsorption  $K_d$  values will likely overestimate the degree to which desorption will occur. To account for this difference in adsorption and desorption, one could artificially increase the  $K_d$  values used in a transport code when cesium is desorbing from contaminated soil.

#### ***D.2.5 Cesium Adsorption as a Function of Dissolved Cesium Concentrations***

At very low concentrations, the adsorption isotherm for cesium is linear. The linear range varies dependent on the adsorbing phase and on the background aqueous phase (Akiba *et al.*, 1989; Sposito, 1989). Table D.7 provides the linear range of some Freundlich adsorption isotherm data reported in the literature. The upper limit of the linear range varies by several orders of magnitude depending on the solid phase and aqueous chemistry. The lowest upper limit reported in Table D.7 is  $1 \times 10^{-10}$  M cesium. This is in fact a rather high concentration when compared to those found in groundwater plumes. For instance, the highest reported  $^{137}\text{Cs}$  concentration in the groundwaters beneath the Hanford Site in 1994 was  $1.94 \times 10^{-13}$  M (or 2,310 pCi/l) for Well 299 E-28-23 (Hartman and Dresel, 1997). This is several orders of magnitude below the smallest upper limit reported in Table D.7, suggesting that most far-field radioactive cesium adsorption likely follows a linear isotherm. The simple  $K_d$  value describes a linear isotherm.

**Table D.7.** Approximate upper limits of linear range of adsorption isotherms on various solid phases.

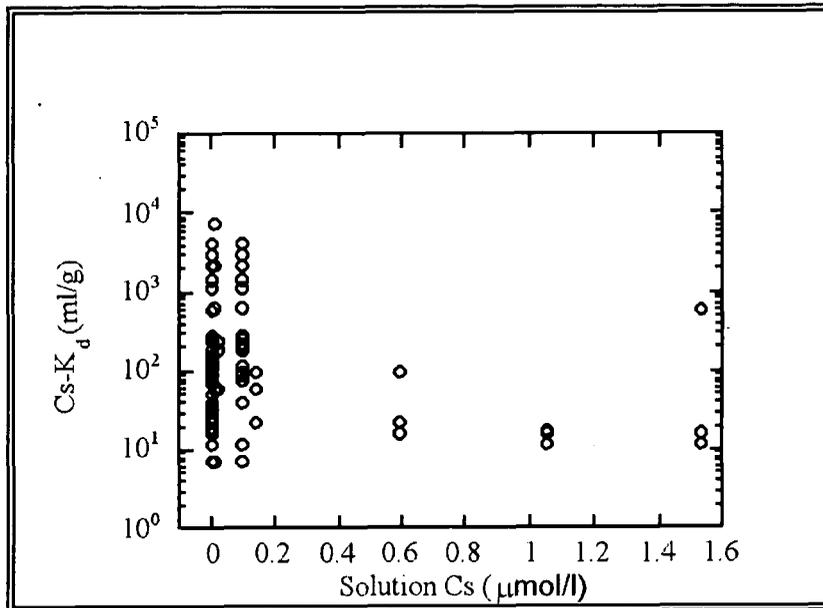
Upper Limit of Linear Range (M)	Solid Phase	Background Aqueous Phase	Reference
$1 \times 10^{-7}$	Itado Tuff	Deionized Water	Akida <i>et al.</i> , 1989
$1 \times 10^{-10}$	Sandstone	Deionized Water	Akida <i>et al.</i> , 1989
$5 \times 10^{-5}$	Limestone	Deionized Water	Akida <i>et al.</i> , 1989
$1 \times 10^{-10}$	Augite Andesite	Deionized Water	Akida <i>et al.</i> , 1989
$5 \times 10^{-9}$	Olivine Basalt	Deionized Water	Akida <i>et al.</i> , 1989
$1 \times 10^{-8}$	Rokko Granite	Deionized Water	Akida <i>et al.</i> , 1989
$5 \times 10^{-8}$	Biotite	Deionized Water	Akida <i>et al.</i> , 1989
$5 \times 10^{-7}$	Albite	Deionized Water	Akida <i>et al.</i> , 1989
$1 \times 10^{-6}$	K-Feldspar	Deionized Water	Akida <i>et al.</i> , 1989
$1 \times 10^{-1}$	Unwashed Kaolinite	Distilled Water/pH 10	Adeleye <i>et al.</i> , 1994
$<1 \times 10^{-5}$	Ca Montmorillonite	Distilled Water/pH 10	Adeleye <i>et al.</i> , 1994
$<1 \times 10^{-5}$	Na Montmorillonite	Distilled Water/pH 10	Adeleye <i>et al.</i> , 1994
$<1 \times 10^{-5}$	Na Kaolinite	Distilled Water/pH 10	Adeleye <i>et al.</i> , 1994
$1 \times 10^{-3}$	Na Montmorillonite	Distilled Water/pH 4	Adeleye <i>et al.</i> , 1994

When a wider range of cesium concentrations are considered, cesium adsorption onto soils and pure minerals has been reported to be almost without exception a non-linear relationship (Adeleye *et al.*, 1994; Akiba *et al.*, 1989; Ames *et al.*, 1982; Erten *et al.*, 1988; Konishi *et al.*, 1988; Lieser and Staunton, 1994; Steinkopff, 1989; Torstenfelt *et al.*, 1982). Most investigators have used a Freundlich equation to describe this relationship (Adeleye *et al.*, 1994; Konishi *et al.*, 1988; Shiao *et al.*, 1979; Staunton, 1994; Torstenfelt *et al.*, 1982). The Freundlich equation is

$$C_{s_{\text{absorbed}}} = a (C_{s_{\text{solution}}})^b \quad (\text{D.2})$$

where  $C_{s_{\text{absorbed}}}$  and  $C_{s_{\text{solution}}}$  are the cesium concentrations adsorbed and in solution, respectively, and  $a$  and  $b$  are fitting parameters. A short description of those Freundlich Equation reported in the literature are presented in Table D.8. The descriptive statistics of the Freundlich Equations

reported in Table D.8 are described in Table D.9. A plot of available cesium adsorption versus equilibrium cesium solution concentration is shown in Figure D.3.



**Figure D.3.**  $K_d$  values calculated from an overall literature Freundlich equation for cesium (Equation D.2).

**Table D.8.** Freundlich equations identified in literature for cesium.

<b>a<sup>1</sup></b>	<b>b<sup>1</sup></b>	<b>Range of Solution Cs Concentration (M)</b>	<b>Experimental</b>	<b>Ref.<sup>2</sup></b>
1.7	0.677		Water/Batcombe Sediment	1
3,300	0.909		Water/Denchworth Sediment	1
260	0.841		Water/Tedburn Sediment	1
16	0.749		Water/Teigngrace Sediment	1
12.2	0.745	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	Water/Batcombe Sediment	1
6,070	0.899	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	Water/Denchworth Sediment	1
1,290	0.849	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	Water/Tedburn Sediment	1
163	0.815	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	Water/Teigngrace Sediment	1
1.23	0.657	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	CaCl <sub>2</sub> /Batcombe Sediment	1
0.63	0.659		CaCl <sub>2</sub> /Batcombe Sediment	1
427	0.814	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	CaCl <sub>2</sub> /Denchworth Sediment	1
1.5	0.599		CaCl <sub>2</sub> /Denchworth Sediment	1
48.1	0.754	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	CaCl <sub>2</sub> /Tedburn Sediment	1
17	0.739		CaCl <sub>2</sub> /Tedburn Sediment	1
5.22	0.702	1x10 <sup>-8</sup> to 1x10 <sup>-12</sup>	CaCl <sub>2</sub> /Teigngrace Sediment	1
4.4	0.716		CaCl <sub>2</sub> /Teigngrace Sediment	1
0.22	1.1	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Bentonite/Water	2
0.017	0.53	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Bentonite/Water	2
0.13	1	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Bentonite/Groundwater	2
0.048	0.67	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Bentonite/Groundwater	2
5.10x10 <sup>-4</sup>	0.21	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Takadata Loam/Water	2
3.00x10 <sup>-3</sup>	0.48	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Takadata Loam/Groundwater	2
1.30x10 <sup>-5</sup>	0.013	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Hachinohe Loam/Water	2
2.30x10 <sup>-5</sup>	0.38	1x10 <sup>-9</sup> to 1.5x10 <sup>-2</sup>	Hachinohe Loam/Groundwater	2

a <sup>1</sup>	b <sup>1</sup>	Range of Solution Cs Concentration (M)	Experimental	Ref. <sup>2</sup>
2.70x10 <sup>-4</sup>	0.546	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Unwashed/Kaolinite/pH 2	3
5.20x10 <sup>-4</sup>	0.543	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Unwashed/Kaolinite/pH 4	3
2.04x10 <sup>-3</sup>	0.588	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Unwashed/Kaolinite/pH 10	3
2.27x10 <sup>-3</sup>	0.586	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Sodium/Kaolinite/pH 2	3
5.04x10 <sup>-2</sup>	0.723	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Sodium/Kaolinite/pH 4	3
3.49x10 <sup>-2</sup>	0.703	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Na/Kaolinite/pH 7	3
0.235	0.821	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Na/Kaolinite/pH 10	3
3.03x10 <sup>-2</sup>	0.804	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Kaolinite/pH 2	3
0.135	0.845	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Kaolinite/pH 4	3
0.247	0.881	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Kaolinite/pH 7	3
8.71x10 <sup>-3</sup>	0.694	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Kaolinite/pH 10	3
1.02x10 <sup>-4</sup>	0.503	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Na/Montmorillonite/pH 2	3
1.05x10 <sup>-2</sup>	0.709	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Na/Montmorillonite/pH 4	3
3.17x10 <sup>-2</sup>	0.755	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Na/Montmorillonite./pH 7	3
0.224	0.815	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Na/Montmorillonite/pH 10	3
0.241	0.839	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Montmorillonite/pH 2	3
0.481	0.897	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Montmorillonite/pH 4	3
1.84	0.938	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Montmorillonite/pH 7	3
0.274	0.82	1x10 <sup>-8</sup> to 1x10 <sup>-2</sup>	Ca/Montmorillonite/pH 10	3
3.40x10 <sup>-2</sup>	0.51	1x10 <sup>-7</sup> to 1x10 <sup>-3</sup>	Granite/pH 8.2	4
4.90x10 <sup>-2</sup>	0.5	1x10 <sup>-7</sup> to 1x10 <sup>-3</sup>	Granite/pH 8.2	4
4.00x10 <sup>-2</sup>	0.5			5

<sup>1</sup> Parameters "a" and "b" are fitting parameters in the Freundlich equation.

<sup>2</sup> References: 1 = Fukui, 1990; 2 = Konishi *et al.*, 1988; 3 = Adeleye *et al.*, 1994; 4 = Serne *et al.*, 1993; 5 = Shiao *et al.*, 1979.

**Table D.9.** Descriptive statistics of the cesium Freundlich equations (Table D.8) reported in the literature.

Statistic	a	b
Mean	252	0.696
Standard Error	150.2	0.029
Median	0.222	0.720
Mode	NA	0.815
Standard Deviation	1019	0.198
Sample Variance	1038711	0.039
Range	6070	1.087
Minimum	0.000013	0.013
Maximum	6070	1.1
95% Confidence Level	302	0.059

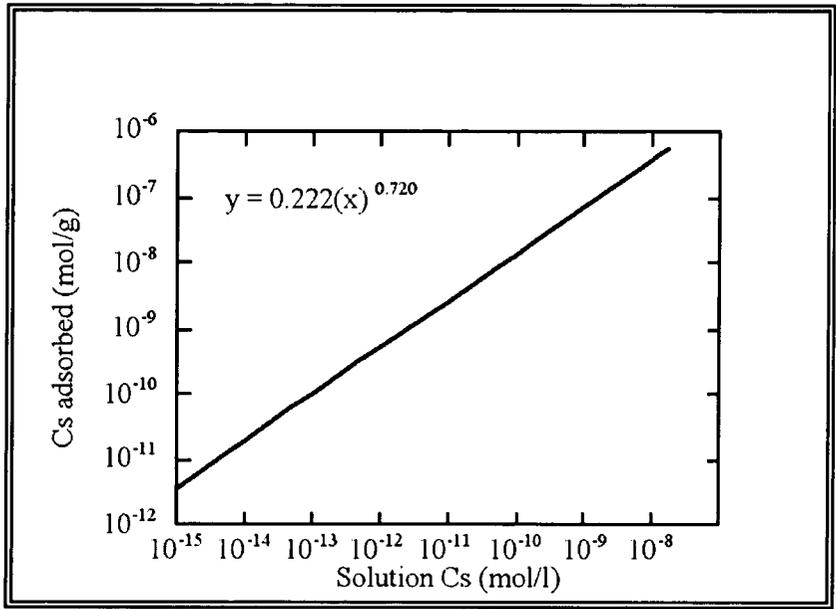
Using the medians of the a and b parameters from the literature, we come up with the overall equation:

$$C_{s_{\text{adsorbed}}} = 0.222(C_{s_{\text{solution}}})^{0.720} \quad (\text{D.3})$$

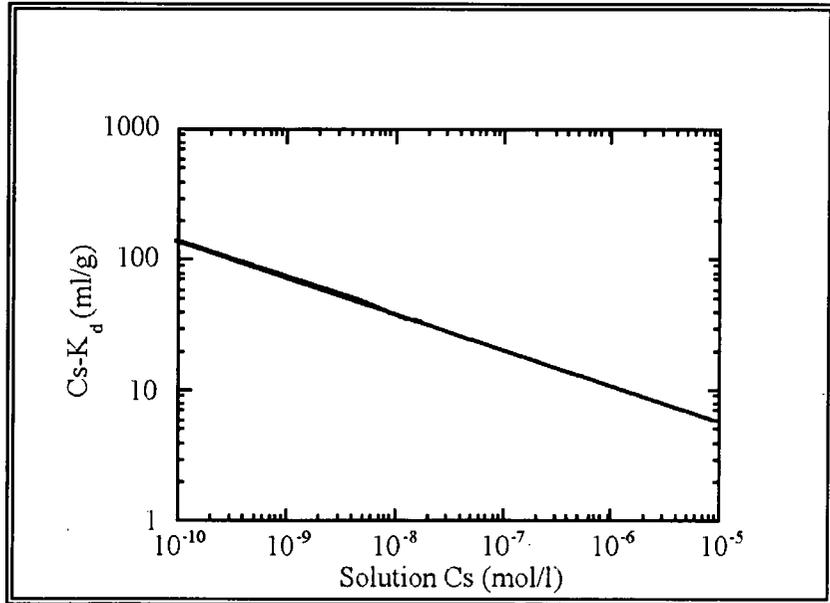
This equation is plotted in Figure D.4. Using  $C_{s_{\text{adsorbed}}}$  and  $C_{s_{\text{solution}}}$  from equation D.3, a  $K_d$  value can be calculated according to equations D.4,

$$K_d = C_{s_{\text{adsorbed}}}/C_{s_{\text{solution}}} \quad (\text{D.4})$$

Cesium  $K_d$  values calculated from Equations D.3 and D.4 are presented in Figure D.5.



**Figure D.4.** Generalized cesium Freundlich equation (Equation D.3) derived from the literature.



**Figure D.5.** Cesium  $K_d$  values calculated from generalized Freundlich equation (Equations D.3 and D.4) derived from the literature.

### ***D.2.6 Approach to Selecting $K_d$ Values for Look-up Table***

Linear regression analyses were conducted with data collected from the literature. These analyses were used as guidance for selecting appropriate  $K_d$  values for the look-up table. The  $K_d$  values used in the look-up tables could not be based entirely on statistical consideration because the statistical analysis results were occasionally nonsensible. For example, the data showed a negative correlation between pH and CEC, and pH and cesium  $K_d$  values. These trends contradict well established principles of surface chemistry. Instead, the statistical analysis was used to provide guidance as to the approximate range of values to use and to identify meaningful trends between the cesium  $K_d$  values and the solid phase parameters. Thus, the  $K_d$  values included in the look-up table were in part selected based on professional judgment. Again, only low-ionic strength solutions, such as groundwaters, were considered; thus no solution variables were included.

Two look-up tables containing cesium  $K_d$  values were created. The first table is for systems containing low concentrations (*i.e.*, less than about 5 percent of the clay-size fraction) of mica-like minerals (Table D.10). The second table is for systems containing high concentrations of mica-like minerals (Table D.11). For both tables, the user will be able to reduce the range of possible cesium  $K_d$  values with knowledge of either the CEC or the clay content.

The following steps were taken to assign values to each category in the look-up tables. A relation between CEC and clay content was established using data presented in this section. Three CEC and clay content categories were selected. The limits of these categories were arbitrarily assigned. The central estimates for the <5 percent mica look-up table (Table D.10) were assigned using the CEC/cesium  $K_d$  equation in Figure D.1. The central estimates for the >5 percent mica look-up table (Table D.11) were assigned by multiplying the central estimates from Table D.10 by a factor of 2.5. The 2.5 scaler was selected based on relationships existing in the values in the data set and in Table D.6. Finally, the lower and upper limits for these central estimates were estimated based on the assumption that there was 2.5 orders of magnitude variability associated with the central estimates. The variability was based on visual inspection of a number of figures containing the cesium  $K_d$  values, including Figure D.1.

The calculations and equations used to estimate the central, minimum, and maximum estimates used in the look-up tables are presented in Table D.12.

**Table D.10.** Estimated range of  $K_d$  values (ml/g) for cesium based on CEC or clay content for systems *containing <5% mica-like minerals* in clay-size fraction and  $<10^{-9}$  M aqueous cesium. [Table pertains to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength ( $< 0.1$  M), low humic material concentrations ( $<5$  mg/l), no organic chelates (such as EDTA), and oxidizing conditions]

$K_d$ (ml/g)	CEC (meq/100 g) / Clay Content (wt.%)		
	<3 / <4	3 - 10 / 4 - 20	10 - 50 / 20 - 60
Central	200	500	1,500
Minimum	10	30	80
Maximum	3,500	9,000	26,700

**Table D.11.** Estimated range of  $K_d$  values (ml/g) for cesium based on CEC or clay content for systems *containing >5% mica-like minerals* in Clay-size fraction and  $<10^{-9}$  M aqueous cesium. [Table pertains to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength ( $< 0.1$  M), low humic material concentrations ( $<5$  mg/l), no organic chelates (such as EDTA), and oxidizing conditions.]

$K_d$ (ml/g)	CEC (meq/100 g) / Clay Content (wt.%)		
	<3 / <4	3 - 10 / 4 - 20	10 - 50 / 20 - 60
Central	500	1250	3750
Minimum	30	70	210
Maximum	9,000	22,000	66,700

**Table D.12.** Calculations for values used in look-up table.

Mica Concentration in Clay Fraction (%)	Clay Content (wt.%)	CE <sup>1</sup> (ml/g)	Logarithm Scale		Base-10 Scale	
			Log CE	Lower Limit (Log CE)/2	Lower Limit $10^{(\log CE)/2}$ (ml/g)	Upper Limit $10^{\log CE + (\log CE)/2}$ (ml/g)
<5	<4	200	2.301	1.151	14	2,828
<5	4 -20	500	2.699	1.349	22	11,180
<5	20 - 60	1,500	3.176	1.588	39	58,095
>5	<4	500	2.699	1.349	22	11,180
>5	4 -20	1,250	3.097	1.548	35	44,194
>5	20 - 60	3,750	3.574	1.787	61	229,640

<sup>1</sup> CE = Central Estimate

### D.3.0 $K_d$ Data Set for Soils and Pure Mineral Phases

Table D.13 lists the available cesium  $K_d$  values identified for experiments conducted with soils and pure mineral phases.

**Table D.13.** Cesium  $K_d$  data base for soils and pure mineral phases

Cesium $K_d$ (ml/g)	Clay (wt.%)	Mica (%)	pH	CEC* (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Aqueous Cs ( $\mu$ M)	Background Aqueous	Soil and Mineral Phase ID and Information	Ref <sup>2</sup>
247			6.2			$1.90 \times 10^{-2}$	Gorleben Groundwater	Gorleben Sediment	1
62			6.2			$1.42 \times 10^{-1}$		Gorleben Sediment	1
22			6.2			$5.94 \times 10^{-1}$		Gorleben Sediment	1
16			6.2			1.05		Gorleben Sediment	1
12			6.2			1.53		Gorleben Sediment	1
167			8.1		189	$5.20 \times 10^{-3}$	Groundwater-1	S1: Quartz, Kaolinite, Plagioclase	2
1			7.8		113	$5.20 \times 10^{-3}$	Groundwater-2	S2: Quartz, Kaolinite, Dolomite	2
1500			9.3	60	70	$1.00 \times 10^{-1}$	Water pH 9.3	Bentonite	3
160			2.4	60	70	$1.00 \times 10^{-1}$	Groundwater pH 2.4	Bentonite	3
1100			9.3	60	70	$1.00 \times 10^{-1}$	Groundwater pH 9.3	Bentonite	3
4100			6.1	20	130	$1.00 \times 10^{-1}$	Water pH 6.1	Takadate loam	3
1400			7.7	20	130	$1.00 \times 10^{-1}$	Groundwater pH 7.7	Takadate loam	3
1100			6.6	70	60	$1.00 \times 10^{-1}$	Water pH 6.6	Hachinohe loam	3
280			8.3	70	60	$1.00 \times 10^{-1}$	Groundwater pH 8.3	Hachinohe loam	3
237			8.2	2	22	$1.00 \times 10^{-3}$		ym-22	4
8220			8.2	109	103	$1.00 \times 10^{-3}$		ym-38	4
325			8.2	6	43	$1.00 \times 10^{-3}$		ym-45	4
22100			8.2	51	19	$1.00 \times 10^{-3}$		ym-48	4
35800			8.2	107		$1.00 \times 10^{-3}$		ym-49	4
42600			8.2	107		$1.00 \times 10^{-3}$		ym-49	4
205			8.2	4		$1.00 \times 10^{-3}$		ym-54	4

Cesium Kd (ml/g)	Clay (wt.%)	Mica (%)	pH	CEC* (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Aqueous Cs (μM)	Background Aqueous	Soil and Mineral Phase ID and Information	Ref <sup>2</sup>
15200			8.4		31	1.00x10 <sup>-3</sup>	low salts	JA-18	4
8440			8.3		31	1.00x10 <sup>-3</sup>	hi salts	JA-18	4
143			8.2		8	1.00x10 <sup>-3</sup>	low salts	JA-32	4
73			8.5		8	1.00x10 <sup>-3</sup>	hi salts	JA-32	4
1390			8.4		100	1.00x10 <sup>-3</sup>	low salts	JA-37	4
757			8.5		100	1.00x10 <sup>-3</sup>	hi salts	JA-37	4
95		15	4			4.20x10 <sup>-4</sup>	0.005 M Na	Savannah River	5
120		15	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Savannah River	5
130		15	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Savannah River	5
130		15	7			4.20x10 <sup>-4</sup>	0.005 M Na	Savannah River	5
150		15	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	Savannah River	5
160		15	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Savannah River	5
72		3	4			4.20x10 <sup>-4</sup>	0.005 M Na	4-Mile Creek	5
79		3	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	4-Mile Creek	5
75		3	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	4-Mile Creek	5
98		3	7			4.20x10 <sup>-4</sup>	0.005 M Na	4-Mile Creek	5
83		3	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	4-Mile Creek	5
33		4	4			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	5
37		4	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	5
40		4	7			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	5
39		4	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	5
50		4	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	5
27		2	4			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	5
25		2	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	5
26		2	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	5
26		2	7			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	5
38		2	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	5
39		2	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	5
88		4	4			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Soil	5
92		4	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Soil	5
93		4	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Soil	5
85		4	7			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Soil	5

Cesium Kd (ml/g)	Clay (wt.%)	Mica (%)	pH	CEC* (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Aqueous Cs (μM)	Background Aqueous	Soil and Mineral Phase ID and Information	Ref <sup>2</sup>
94		4	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Soil	5
101		4	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Soil	5
88		5	4			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	5
89		5	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	5
90		5	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	5
84		5	7			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	5
101		5	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	5
22		2	4			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	5
31		2	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	5
37		2	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	5
40		2	7			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	5
78		2	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	5
27			8.25	1.83	17.7	2.72x10 <sup>2</sup>	0.002 M Groundwater	Umtanum Basalt	6
329			8.25	1.83	17.7	2.90x10 <sup>-1</sup>	0.002 M Groundwater	Umtanum Basalt	6
960			8.25	1.83	17.7	1.03x10 <sup>-3</sup>	0.002 M Groundwater	Umtanum Basalt	6
1088			8.25	1.83	17.7	9.11x10 <sup>-6</sup>	0.002 M Groundwater	Umtanum Basalt	6
1084			8.25	1.83	17.7	1.87x10 <sup>-6</sup>	0.002 M Groundwater	Umtanum Basalt	6
28			8.6	1.83	17.7	2.63x10 <sup>2</sup>	0.013 M Groundwater	Umtanum Basalt	6
289			8.6	1.83	17.7	3.31x10 <sup>-1</sup>	0.013 M Groundwater	Umtanum Basalt	6
951			8.6	1.83	17.7	1.05x10 <sup>-3</sup>	0.013 M Groundwater	Umtanum Basalt	6
1022			8.6	1.83	17.7	9.77x10 <sup>-6</sup>	0.013 M Groundwater	Umtanum Basalt	6
1025			8.6	1.83	17.7	1.95x10 <sup>-6</sup>	0.013 M Groundwater	Umtanum Basalt	6
18			8.2	1.5	10.3	3.61x10 <sup>2</sup>	0.002 M Groundwater	Flow E Basalt	6
189			8.2	1.5	10.3	5.00x10 <sup>-1</sup>	0.002 M Groundwater	Flow E Basalt	6
418			8.2	1.5	10.3	2.34x10 <sup>-3</sup>	0.002 M Groundwater	Flow E Basalt	6

Cesium Kd (ml/g)	Clay (wt.%)	Mica (%)	pH	CEC* (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Aqueous Cs (μM)	Background Aqueous	Soil and Mineral Phase ID and Information	Ref <sup>2</sup>
450			8.2	1.5	10.3	2.17x10 <sup>-5</sup>	0.002 M Groundwater	Flow E Basalt	6
487			8.2	1.5	10.3	3.98x10 <sup>-6</sup>	0.002 M Groundwater	Flow E Basalt	6
20			8.7	1.5	10.3	3.39x10 <sup>2</sup>	0.013 M Groundwater	Flow E Basalt	6
214			8.7	1.5	10.3	4.47x10 <sup>-1</sup>	0.013 M Groundwater	Flow E Basalt	6
488			8.7	1.5	10.3	2.00x10 <sup>-3</sup>	0.013 M Groundwater	Flow E Basalt	6
549			8.7	1.5	10.3	1.78x10 <sup>-5</sup>	0.013 M Groundwater	Flow E Basalt	6
617			8.7	1.5	10.3	3.24x10 <sup>-6</sup>	0.013 M Groundwater	Flow E Basalt	6
48			8.3	4.84	31.2	1.71x10 <sup>2</sup>	0.002 M Groundwater	Pomona Basalt	6
460			8.3	4.84	31.2	2.13x10 <sup>-1</sup>	0.002 M Groundwater	Pomona Basalt	6
1111			8.3	4.84	31.2	8.30x10 <sup>-4</sup>	0.002 M Groundwater	Pomona Basalt	6
1466			8.3	4.84	31.2	6.37x10 <sup>-6</sup>	0.002 M Groundwater	Pomona Basalt	6
1281			8.3	4.84	31.2	1.39x10 <sup>-6</sup>	0.002 M Groundwater	Pomona Basalt	6
56			8.55	4.84	31.2	1.51x10 <sup>2</sup>	0.013 M Groundwater	Pomona Basalt	6
389			8.55	4.84	31.2	2.57x10 <sup>-1</sup>	0.013 M Groundwater	Pomona Basalt	6
853			8.55	4.84	31.2	1.17x10 <sup>-3</sup>	0.013 M Groundwater	Pomona Basalt	6
952			8.55	4.84	31.2	1.05x10 <sup>-5</sup>	0.013 M Groundwater	Pomona Basalt	6
908			8.55	4.84	31.2	1.74x10 <sup>-6</sup>	0.013 M Groundwater	Pomona Basalt	6
212			8.3	71	646	4.50x10 <sup>1</sup>	0.002 M Groundwater	Smectite	6
1080			8.3	71	646	9.17x10 <sup>-1</sup>	0.002 M Groundwater	Smectite	6
13042			8.3	71	646	7.66x10 <sup>-5</sup>	0.002 M Groundwater	Smectite	6

Cesium Kd (ml/g)	Clay (wt.%)	Mica (%)	pH	CEC* (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Aqueous Cs (μM)	Background Aqueous	Soil and Mineral Phase ID and Information	Ref <sup>2</sup>
9794			8.3	71	646	1.00x10 <sup>-6</sup>	0.002 M Groundwater	Smectite	6
25000			8.3	71	646	7.00x10 <sup>-8</sup>	0.002 M Groundwater	Smectite	6
224			9.2	71	646	4.27x10 <sup>-1</sup>	0.013 M Groundwater	Smectite	6
2136			9.2	71	646	4.68x10 <sup>-2</sup>	0.013 M Groundwater	Smectite	6
5882			9.2	71	646	1.70x10 <sup>-4</sup>	0.013 M Groundwater	Smectite	6
8547			9.2	71	646	1.17x10 <sup>-6</sup>	0.013 M Groundwater	Smectite	6
8333			9.2	71	646	2.40x10 <sup>-7</sup>	0.013 M Groundwater	Smectite	6
5000	24		4.4	82		6.80x10 <sup>-2</sup>	1x10 <sup>-6</sup> M KCl	Batcombe	7
5000	24		4.4	82		6.80x10 <sup>-2</sup>	1x10 <sup>-5</sup> M KCl	Batcombe	7
4700	24		4.4	82		6.80x10 <sup>-2</sup>	1x10 <sup>-4</sup> M KCl	Batcombe	7
2000	24		4.4	82		6.80x10 <sup>-2</sup>	1x10 <sup>-3</sup> M KCl	Batcombe	7
9000	42		6.2	72		6.80x10 <sup>-2</sup>	1x10 <sup>-6</sup> M KCl	Tedburn	7
8000	42		6.2	72		6.80x10 <sup>-2</sup>	1x10 <sup>-5</sup> M KCl	Tedburn	7
9000	42		6.2	72		6.80x10 <sup>-2</sup>	1x10 <sup>-4</sup> M KCl	Tedburn	7
2000	42		6.2	72		6.80x10 <sup>-2</sup>	1x10 <sup>-3</sup> M KCl	Tedburn	7
1050	42		7.3	54		6.80x10 <sup>-2</sup>	1x10 <sup>-6</sup> M KCl	Teigngrace	7
1025	42		7.3	54		6.80x10 <sup>-2</sup>	1x10 <sup>-5</sup> M KCl	Teigngrace	7
1000	42		7.3	54		6.80x10 <sup>-2</sup>	1x10 <sup>-4</sup> M KCl	Teigngrace	7
800	42		7.3	54		6.80x10 <sup>-2</sup>	1x10 <sup>-3</sup> M KCl	Teigngrace	7
11000				130		1.00x10 <sup>-7</sup>	Water	Itago Tuff	8
10000				97		1.00x10 <sup>-7</sup>	Water	Ohya Tuff	8
5000				2.4		1.00x10 <sup>-7</sup>	Water	Sandstone	8
2000				1.9		1.00x10 <sup>-7</sup>	Water	Shale	8
6000				1.9		1.00x10 <sup>-7</sup>	Water	Augite Audesite	8
500				1.2		1.00x10 <sup>-7</sup>	Water	Plagio Rhyolite	8
5800				0.75		1.00x10 <sup>-7</sup>	Water	Olivine Basalt	8
900				0.54		1.00x10 <sup>-7</sup>	Water	Ionada Granite	8
260				0.35		1.00x10 <sup>-7</sup>	Water	Rokka Granite	8

Cesium Kd (ml/g)	Clay (wt.%)	Mica (%)	pH	CEC* (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Aqueous Cs (μM)	Background Aqueous	Soil and Mineral Phase ID and Information	Ref <sup>2</sup>
80				0.033		1.00x10 <sup>-7</sup>	Water	Limestone	8
2200				1.2		1.00x10 <sup>-7</sup>	Water	Biotite	8
1800				0.93		1.00x10 <sup>-7</sup>	Water	Chlorite	8
630				0.33		1.00x10 <sup>-7</sup>	Water	Hornblende	8
420				0.11		1.00x10 <sup>-7</sup>	Water	Grossular	8
460				0.0067		1.00x10 <sup>-7</sup>	Water	Forsterite	8
30				0.0034		1.00x10 <sup>-7</sup>	Water	K-feldspar	8
89				0.0032		1.00x10 <sup>-7</sup>	Water	Albite	8
31				0.00098		1.00x10 <sup>-7</sup>	Water	Quartz	8
1				0.15849		1.00x10 <sup>-1</sup>		Calcite	9
3				0.19953		1.00x10 <sup>-1</sup>		Apatite	9
6				1.58489		1.00x10 <sup>-1</sup>		Hematite	9
13				1.77828		1.00x10 <sup>-1</sup>		Orthoclase	9
16				5.62341		1.00x10 <sup>-1</sup>		Serpentine	9
200				7.94328		1.00x10 <sup>-1</sup>		Hornblende	9
631				39.8107		1.00x10 <sup>-1</sup>		Biotite	9
794				63.0957		1.00x10 <sup>-1</sup>		Muscovite	9
100				4.46684		1.00x10 <sup>-1</sup>		Gneiss	9
16				6.30957		1.00x10 <sup>-1</sup>		Diabase	9
158				10		1.00x10 <sup>-1</sup>		Stripa Granite	9
562				11.2202		1.00x10 <sup>-1</sup>		Finsjo Granite	9
900			5			1.00x10 <sup>-1</sup>		Biotite	9
790			7			1.00x10 <sup>-1</sup>		Biotite	9
700			9			1.00x10 <sup>-1</sup>		Biotite	9
2			5			1.00x10 <sup>-1</sup>		Hematite	9
4			7			1.00x10 <sup>-1</sup>		Hematite	9
8			9			1.00x10 <sup>-1</sup>		Hematite	9
40			5			1.00x10 <sup>-1</sup>		Hornblende	9
100			7			1.00x10 <sup>-1</sup>		Hornblende	9
240			9			1.00x10 <sup>-1</sup>		Hornblende	9
3			5			1.00x10 <sup>-1</sup>		Magnetite	9
5			7			1.00x10 <sup>-1</sup>		Magnetite	9

Cesium Kd (ml/g)	Clay (wt.%)	Mica (%)	pH	CEC* (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Aqueous Cs (μM)	Background Aqueous	Soil and Mineral Phase ID and Information	Ref <sup>2</sup>
9			9			1.00x10 <sup>-1</sup>		Magnetite	9
700			5			1.00x10 <sup>-1</sup>		Muscovite	9
810			7			1.00x10 <sup>-1</sup>		Muscovite	9
840			9			1.00x10 <sup>-1</sup>		Muscovite	9
7			5			1.00x10 <sup>-1</sup>		Orthoclase	9
14			7			1.00x10 <sup>-1</sup>		Orthoclase	9
7			9			1.00x10 <sup>-1</sup>		Orthoclase	9
52000				127		1.67x10 <sup>-6</sup>	Deionized Water	Vermiculite	10
26000				20		1.67x10 <sup>-6</sup>	Deionized Water	Illite	10
2500				11.2		1.67x10 <sup>-6</sup>	Deionized Water	Kaolinite	10
2700				127		1.67x10 <sup>-6</sup>	0.1 N NaCl	Vermiculite	10
28600				20		1.67x10 <sup>-6</sup>	0.1 N NaCl	Illite	10
94				11.2		1.67x10 <sup>-6</sup>	0.1 N NaCl	Kaolinite	10
7						1.00x10 <sup>-7</sup>	Groundwater	Hanford Vadose Sediment	11
12						1.00x10 <sup>-7</sup>	Groundwater	Hanford Vadose Sediment	11
2190	4	9	7.7			8.40x10 <sup>-3</sup>	Groundwater	Sediment CGS-1	12
7610	5	12	8.2			8.40x10 <sup>-3</sup>	Groundwater	Sediment TBS-1	12
620	6	9	7.9			8.40x10 <sup>-3</sup>	Groundwater	Sediment Trench-8	12

<sup>1</sup> CEC = cation exchange capacity; SA = surface area.  
<sup>2</sup> References: 1 = Lieser and Steinkopff, 1989; 2 = Lieser *et al.*, 1986; 3 = Konishi *et al.*, 1988; 4 = Vine *et al.*, 1980; 5 = Elprince *et al.*, 1977; 6 = Ames *et al.*, 1982; 7 = Staunton, 1994; 8 = Akiba *et al.*, 1989; 9 = Torstenfelt *et al.*, 1982; 10 = Tamura, 1972; 11 = Routson *et al.*, 1980; 12 = Serne *et al.*, 1993.

## D.4.0 Data Set for Soils

Table D.14 lists the available cesium  $K_d$  values identified for experiments conducted with only soils.

**Table D.14.** Cesium  $K_d$  data set for soils only.

Cesium $K_d$ (ml/g)	Clay (wt%)	Mica (%)	pH	CEC <sup>(a)</sup> (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Cs ( $\mu$ M)	Aqueous Phase	Soil ID and Information	Ref. <sup>2</sup>
247			6.2			$1.90 \times 10^{-2}$	Gorleben Groundwater	Gorleben Sediment	1
62			6.2			$1.42 \times 10^{-1}$		Gorleben Sediment	1
22			6.2			$5.94 \times 10^{-1}$		Gorleben Sediment	1
4100			6.1	20	130	$1.00 \times 10^{-1}$	Water pH 6.1	Takadate Loam	4
1400			7.7	20	130	$1.00 \times 10^{-1}$	Groundwater pH 7.7	Takadate Loam	4
1100			6.6	70	60	$1.00 \times 10^{-1}$	Water pH 6.6	Hachinohe Loam	4
280			8.3	70	60	$1.00 \times 10^{-1}$	Groundwater pH 8.3	Hachinohe loam	4
95		15	4			$4.20 \times 10^{-4}$	0.005 M Na	Sav. River Site Sediment	6
120		15	5.5			$4.20 \times 10^{-4}$	0.005 M Na	Sav. River Site Sediment	6
130		15	6.7			$4.20 \times 10^{-4}$	0.005 M Na	Sav. River Site Sediment	6
130		15	7			$4.20 \times 10^{-4}$	0.005 M Na	Sav. River Site Sediment	6
150		15	8.5			$4.20 \times 10^{-4}$	0.005 M Na	Sav. River Site Sediment	6
160		15	10.2			$4.20 \times 10^{-4}$	0.005 M Na	Sav. River Site Sediment	6
72		3	4			$4.20 \times 10^{-4}$	0.005 M Na	4-Mile Creek Sediment	6
79		3	5.5			$4.20 \times 10^{-4}$	0.005 M Na	4-Mile Creek Sediment	6
75		3	6.7			$4.20 \times 10^{-4}$	0.005 M Na	4-Mile Creek Sediment.	6
98		3	7			$4.20 \times 10^{-4}$	0.005 M Na	4-Mile Creek Sediment.	6
83		3	8.5			$4.20 \times 10^{-4}$	0.005 M Na	4-Mile Creek Sediment.	6
33		4	4			$4.20 \times 10^{-4}$	0.005 M Na	Par Pond Soil	6

Cesium K <sub>d</sub> (ml/g)	Clay (wt%)	Mica (% )	pH	CEC <sup>(a)</sup> (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Cs (μM)	Aqueous Phase	Soil ID and Information	Ref. <sup>2</sup>
37		4	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	6
40		4	7			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	6
39		4	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	6
50		4	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Par Pond Soil	6
27		2	4			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	6
25		2	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	6
26		2	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	6
26		2	7			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	6
38		2	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	6
39		2	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Steel Creek Soil	6
88		4	4			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Soil	6
92		4	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Sediment	6
93		4	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Sediment	6
85		4	7			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Sediment	6
94		4	8.5			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Sediment	6
101		4	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Lower 3 Runs Sediment	6
88		5	4			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	6
89		5	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	6
90		5	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	6
84		5	7			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	6
101		5	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Pen Branch Soil	6
22		2	4			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	6
31		2	5.5			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	6
37		2	6.7			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	6
40		2	7			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	6
78		2	10.2			4.20x10 <sup>-4</sup>	0.005 M Na	Upper 3 Runs Soil	6
7						1.00x10 <sup>-7</sup>	Groundwater	Hanford Vadose Sediment	8
12						1.00x10 <sup>-7</sup>	Groundwater	Hanford Vadose Sediment	8

Cesium K <sub>d</sub> (ml/g)	Clay (wt%)	Mica (%)	pH	CEC <sup>(a)</sup> (meq/100 g)	SA <sup>1</sup> (m <sup>2</sup> /g)	Cs (μM)	Aqueous Phase	Soil ID and Information	Ref. <sup>2</sup>
3,000		6	7.6	3	8.6	1.00x10 <sup>-1</sup>	Groundwater	Sediment A	10
4,800		7.5	5.9	4.3	12.2	1.00x10 <sup>-1</sup>	Groundwater	Sediment B	10
3,100		8	6.6	4.7	14.7	1.00x10 <sup>-1</sup>	Groundwater	Sediment C	10
3,000		5	8	2.6	6.6	1.00x10 <sup>-1</sup>	Groundwater	Sediment D	10
2,190	4	9	7.7			8.40x10 <sup>-3</sup>	Groundwater	Sediment CGS-1	11
7,610	5	12	8.2			8.40x10 <sup>-3</sup>	Groundwater	Sediment TBS-1	11
620	6	9	7.9			8.40x10 <sup>-3</sup>	Groundwater	Sediment Trench-8	11

<sup>1</sup> CEC = cation exchange capacity; SA = surface area.

<sup>2</sup> 1 = Lieser and Steinkopff, 1989; 4 = Konishi *et al.*, 1988; 6 = Elprince *et al.*, 1977; 8 = Routson *et al.*, 1980; 10 = Legoux *et al.*, 1992; 11 = Serne *et al.*, 1993.

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## **APPENDIX E**

### **Partition Coefficients For Chromium(VI)**

## Appendix E

### Partition Coefficients For Chromium(VI)

#### E.1.0 Background

The review of chromium  $K_d$  data obtained for a number of soils (summarized in Table E.1) indicated that a number of factors influence the adsorption behavior of chromium. These factors and their effects on chromium adsorption on soils and sediments were used as the basis for generating a look-up table. These factors are:

- Concentrations of Cr(III) in soil solutions are typically controlled by dissolution/precipitation reactions therefore, adsorption reactions are not significant in soil Cr(III) chemistry.
- Increasing pH decreases adsorption (decrease in  $K_d$ ) of Cr(VI) on minerals and soils. The data are quantified for only a limited number of soils.
- The redox state of the soil affects chromium adsorption. Ferrous iron associated with iron oxide/hydroxide minerals in soils can reduce Cr(VI) which results in precipitation (higher  $K_d$ ). Soils containing Mn oxides oxidize Cr(III) into Cr(VI) form thus resulting in lower  $K_d$  values. The relation between oxide/hydroxide contents of iron and manganese and their effects on  $K_d$  have not been adequately quantified except for a few soils.
- The presence of competing anions reduce Cr(VI) adsorption. The inhibiting effect varies in the order  $\text{HPO}_4^{2-}$ ,  $\text{H}_2\text{PO}_4^- \gg \text{SO}_4^{2-}$   $\text{CO}_3^{2-}/\text{HCO}_3^-$ ,  $\text{Cl}^-$ ,  $\text{NO}_3^-$ . These effects have been quantified as a function of pH for only 2 soils.

The factors which influence chromium adsorption were identified from the following sources of data. Experimental data for Cr(VI) adsorption onto iron oxyhydroxide and aluminum hydroxide minerals (Davis and Leckie, 1980; Griffin *et al.*, 1977; Leckie *et al.*, 1980; Rai *et al.*, 1986) indicate that adsorption increases with decreasing pH over the pH range 4 to 10. Such adsorption behavior is explained on the basis that these oxides show a decrease in the number of positively charged surface sites with increasing pH. Rai *et al.* (1986) investigated the adsorption behavior of Cr(VI) on amorphous iron oxide surfaces. The experiments were conducted with initial concentrations of  $5 \times 10^{-6}$  M Cr(VI). The results showed very high  $K_d$  values (478,630 ml/g) at lower pH values (5.65), and lower  $K_d$  values (6,607 ml/g) at higher pH values (7.80). In the presence of competing anions ( $\text{SO}_4^{2-}$ :  $2.5 \times 10^{-3}$  M, solution in equilibrium with  $3.5 \times 10^{-3}$  atm  $\text{CO}_2$ ), at the same pH values, the observed  $K_d$  values were 18,620 ml/g and 132 ml/g respectively leading to the conclusion that depending on concentration competing anions reduce Cr(VI) adsorption by at least an order of magnitude. Column experiments on 3 different soils conducted by Selim and Amacher (1988) confirmed the influence of soil pH on Cr(VI) adsorption. Cecil,

Windsor, and Olivier soils with pH values of 5.1, 5.4, and 6.4 exhibited chromium  $K_d$  values in the range ~9-100 ml/g, 2-10 ml/g, and ~1-3 ml/g respectively. Adsorption of Cr(VI) on 4 different subsoils was studied by Rai *et al.* (1988). The authors interpreted the results of these experiments using surface complexation models. Using their adsorption data, we calculated the  $K_d$  values for these soils. The data showed that 3 of the 4 soils studied exhibited decreasing  $K_d$  values with increasing pH. The  $K_d$  values for these soils were close to 1 ml/g at higher pH values (>8). At lower pH values (about 4.5) the  $K_d$  values were about 2 to 3 orders of magnitude greater than the values observed at higher pH values. One of the soils with a very high natural pH value (10.5) however did not show any adsorption affinity ( $K_d \leq 1$  ml/g) for Cr(VI).

The data regarding the effects of soil organic matter on Cr(VI) adsorption are rather sparse. In 1 study, Stollenwerk and Grove (1985) evaluated the effects of soil organic matter on adsorption of Cr(VI). Their results indicated that organic matter did not influence Cr(VI) adsorption properties. In another study, the Cr(VI) adsorption properties of an organic soil was examined by Wong *et al.* (1983). The chromium adsorption measurements on bottom, middle, and top layers of this soil produced  $K_d$  values of 346, 865, and 2,905 ml/g respectively. Also, another  $K_d$  measurement using an organic-rich fine sandy soil from the same area yielded a value of 1,729 ml/g.

A series of column (lysimeter) measurements involving Cr(VI) adsorption on 4 different layers of a sandy soil yielded average  $K_d$  values that ranged from 6 to 263 ml/g (Sheppard *et al.*, 1987). These measurements showed that coarse-textured soils tend to have lower  $K_d$  values as compared to fine-textured soils such as loam ( $K_d \sim 1,000$  ml/g, Sheppard and Sheppard, 1987). Stollenwerk and Grove (1985) examined Cr(VI) adsorption on an alluvium from an aquifer in Telluride, Colorado. A  $K_d$  value of 5 ml/g was obtained for Cr(VI) adsorption on this alluvium. Removing organic matter from the soil did not significantly affect the  $K_d$  value. However, removing iron oxide and hydroxide coatings resulted in a  $K_d$  value of about 0.25 leading the authors to conclude that a major fraction of Cr(VI) adsorption capacity of this soil is due to its iron oxide and hydroxide content. Desorption experiments conducted on Cr adsorbed soil aged for 1.5 yrs indicated that over this time period, a fraction of Cr(VI) had been reduced to Cr(III) by ferrous iron and had probably coprecipitated with iron hydroxides.

Studies by Stollenwerk and Grove (1985) and Sheppard *et al.* (1987) using soils showed that  $K_d$  decreases as a function of increasing equilibrium concentration of Cr(VI). Another study conducted by Rai *et al.* (1988) on 4 different soils confirmed that  $K_d$  values decrease with increasing equilibrium Cr(VI) concentration.

Other studies also show that iron and manganese oxide contents of soils significantly affect the adsorption of Cr(VI) on soils (Korte *et al.*, 1976). However, these investigators did not publish either  $K_d$  values or any correlative relationships between  $K_d$  and the oxide contents. The adsorption data obtained by Rai *et al.* (1988) also showed that quantities of sodium dithionite-citrate-bicarbonate (DCB) extractable iron content of soils is a good indicator of a soil's ability to reduce Cr(VI) to Cr(III) oxidation state. The reduced Cr has been shown to coprecipitate with ferric hydroxide. Therefore, observed removal of Cr(VI) from solution when contacted with

chromium-reductive soils may stem from both adsorption and precipitation reaction. Similarly, Rai *et al.* (1988) also showed that certain soils containing manganese oxides may oxidize Cr(III) into Cr(VI). Depending on solution concentrations, the oxidized form (VI) of chromium may also precipitate in the form of  $\text{Ba}(\text{S,Cr})\text{O}_4$ . Such complex geochemical behavior chromium in soils implies that depending on the properties of a soil, the measured  $K_d$  values may reflect both adsorption and precipitation reactions.

An evaluation of competing anions indicated that Cr(VI) adsorption was inhibited to the greatest extent by  $\text{HPO}_4^{2-}$  and  $\text{H}_2\text{PO}_4^-$  ions and to a very small extent by  $\text{Cl}^-$  and  $\text{NO}_3^-$  ions. The data indicate that Cr(VI) adsorption was inhibited by anions in order of  $\text{HPO}_4^{2-}$ ,  $\text{H}_2\text{PO}_4^- \gg \text{SO}_4^{2-} \gg \text{Cl}^-$ ,  $\text{NO}_3^-$  (Leckie *et al.*, 1980; MacNaughton, 1977; Rai *et al.*, 1986; Rai *et al.*, 1988; Stollenwerk and Grove, 1985).

**Table E.1.** Summary of  $K_d$  values for Cr(VI) adsorption on soils.

Soil Description	Clay Content (wt.%)	Organic Carbon (wt.%)	Iron Oxide Content <sup>1</sup> (wt.%)	pH	CEC (meq/100g)	$K_d$ (ml/g)	Experimental Parameters	Reference
Organic Soil (Muck) Top Layer, Florida	NR	7.05	NR	7.1	0.453	2905	NR	Wong <i>et al.</i> (1983)
Organic Soil (Muck) Middle Layer, Florida	NR	6.71	NR	7.2	0.409	865	NR	
Organic Soil (Muck) Bottom Layer, Florida	NR	2.79	NR	7.3	0.158	346	NR	
Hallandale Fine sand, Florida	NR	1.45	NR	8.2	0.113	1729	NR	
Alluvium, Telluride, Colorado	1	0.1	1.2	6.45	NR	1.7 - 52	Batch experiment, deionized water, eq. Cr conc. 1.4 - 0.0004 mmol/l Batch experiment, groundwater (pH: 6.8) Batch experiment, groundwater, Soil with org matter removed Batch experiment, groundwater, Soil with iron oxides removed Column experiment, groundwater, initial Cr conc. (0.01 mmol/l)	Stollenwerk and Grove (1985)
Loam (Chernozem), Canada	NR	NR	NR	NR	60	1000	NR	Sheppard and Sheppard (1987)
Sand (Regosol), Canada	NR	NR	NR	NR	1.6	100	NR	
Sand (Brunisol) organic surface layer (LFH-Ah)	NR	NR	NR	5.2	8.1	263, 6	Column experiments (lysimeter). Solutions: leachate, groundwater	Sheppard <i>et al.</i> , 1987
Sand (Brunisol) upper layer (Ae)	NR	NR	NR	5.1	0.29	91, 35	Column experiments (lysimeter). Solutions: leachate, groundwater	
Sand (Brunisol) middle layer (Bfj)	NR	NR	NR	5.2	0.21	135, 160	Column experiments (lysimeter). Solutions: leachate, groundwater	
Sand (Brunisol) lower layer (Bfjgj)	NR	NR	NR	6.2	0.17	53, 9	Column experiments (lysimeter). Solutions: leachate, groundwater	
<sup>1</sup> Total iron oxide (Fe <sub>2</sub> O <sub>3</sub> ) content of soils. Values within parenthesis represent DCB extractable Fe content (mmol/g) of soils.								

Table E.1. Continued.

Soil Description	Clay Content (wt.%)	Organic Carbon (wt.%)	Iron <sup>1</sup> Oxide Content (wt.%)	pH	CEC (meq/100g)	K <sub>d</sub> (ml/g)	Experimental Parameters	Reference
Bayamon Series Soil	NR	NR	NR	NR	NR	8	NR	Ramirez <i>et al.</i> (1985)
Puerto RicoToa Series Soil	NR	NR	NR	NR	NR	110	NR	
Holton/Cloudland Series Soil (Btx horizon), Tenn.	34	0.05	5.90 (0.435) <sup>a</sup>	4.28	7.3	1585 to 1 <sup>2</sup>	O.1 NaNO <sub>3</sub> , pH: 4.45-9.84, Cr: 10 <sup>-6</sup> M	Rai <i>et al.</i> (1988)
Kenoma Series Soil (Be+C horizon), Kansas	46	0.32	5.40 (0.162) <sup>a</sup>	6.94	28.4	28 to 1 <sup>2</sup>	O.1 NaNO <sub>3</sub> , pH: 5.62-8.42, Cr: 10 <sup>-6</sup> M	
Kenoma Series Soil (Be+C horizon), Kansas	46	0.32	5.40 (0.162) <sup>a</sup>	6.94	28.4	12 to 1 <sup>2</sup>	O.1 NaNO <sub>3</sub> , pH: 5.02-7.49, Cr: 10 <sup>-6</sup> M, SO <sub>4</sub> : 10 <sup>-2.7</sup> M, CO <sub>2</sub> : 10 <sup>-1.46</sup> atm.	
Ocala Series Soil (C4 horizon), Nevada	31	0.14	4.38 (0.009) <sup>a</sup>	10.7	35.7	1 to 0 <sup>2</sup>	O.1 NaNO <sub>3</sub> , pH: 5.14-9.37, Cr: 10 <sup>-6</sup> M	
Cecil/Pacolet Series Soil (Be horizon), N. Carolina	28	0.07	7.70 (0.266) <sup>a</sup>	5.33	4.4	646 to 1 <sup>2</sup>	O.1 NaNO <sub>3</sub> , pH: 4.49-9.29, Cr: 10 <sup>-6</sup> M	
Cecil/Pacolet Series Soil (Be horizon), N. Carolina	28	0.07	7.70 (0.266) <sup>a</sup>	5.33	4.4	59 to 1 <sup>2</sup>	O.1 NaNO <sub>3</sub> , pH: 4.69-8.92, Cr: 10 <sup>-6</sup> M, SO <sub>4</sub> : 10 <sup>-2.7</sup> M	
Cecil/Pacolet Series Soil (Be horizon), N. Carolina	28	0.07	7.70 (0.266) <sup>a</sup>	5.33	4.4	427 to 0 <sup>2</sup>	O.1 NaNO <sub>3</sub> , pH: 4.49-9.29, Cr: 10 <sup>-6</sup> M, CO <sub>2</sub> : 10 <sup>-1.46</sup> atm.	
Cecil Series soil	NR	0.24	10.2	5.1	3.72	~9 - 100	0.005 M Ca(NO <sub>3</sub> ) <sub>2</sub> background initial Cr. 1 - 100 mg/l	Selim and Amacher (1988)
Olivier Series soil	NR	0.99	1.14	6.4	8.31	~1 - 3	0.005 M Ca(NO <sub>3</sub> ) <sub>2</sub> background initial Cr 1 - 100 mg/l	
Windsor Series soil	NR	0.94	2.20	5.4	1.20	~2 - 10	0.005 M Ca(NO <sub>3</sub> ) <sub>2</sub> background initial Cr 1 - 100 mg/l	

NR = Not Reported.  
<sup>1</sup> Total iron oxide (Fe<sub>2</sub>O<sub>3</sub>) content of soils. Values within parenthesis represent DCB extractable Fe content (mmol/g) of soils.  
<sup>2</sup>K<sub>d</sub> values listed from low to high pH conditions used for experiments.

## E.2.0 Approach

The approach used to develop the look-up table was to identify the key parameters that control Cr(VI) adsorption reactions. From the data of Rai *et al.* (1988) and other studies of Cr(VI) adsorption on soils pH was identified as a key parameter. The data show (Table E.2) that the  $K_d$  values are significantly higher at lower pH values and decline with increasing pH. Also,  $K_d$  values for soils show a wider range at lower pH, but values for all soils converge as pH value approaches about 8. Another parameter which seems to influence soil adsorption of Cr(VI) is the capacity of soils to reduce Cr(VI) to Cr(III). Leckie *et al.* (1980) and Rai *et al.* (1988) showed that iron oxides in the soil reduce Cr(VI) to Cr(III) and precipitate Cr(III) as a  $(Fe,Cr)(OH)_3$  mineral. Also, studies conducted by Rai *et al.* (1988) show that DCB extractable iron content is a good indicator as to whether a soil can reduce significant quantities of Cr(VI) which results in higher  $K_d$  values. It is important to note the total iron oxide content is a poor indicator of a soil's Cr(VI) reducing capacity and that DCB extractable iron better represents the fraction of iron content that would reduce Cr(VI) to Cr(III). The data indicated that Holton/Cloudland soil with the highest concentrations of DCB extractable iron (0.435 mmol/g) exhibited higher  $K_d$  values than other soils which did not show an observable Cr(VI) reduction tendency.

Based on this information, 4 ranges of pH, which encompass the pH range of most natural soils, were selected for the look-up table (Table E.3). Within each pH range, 3 ranges of DCB extractable iron content were selected to represent the categories of soils that definitely reduce ( $\geq 0.3$  mmol/g), probably reduce (0.26 to 0.29 mmol/g), and do not reduce ( $\leq 2.5$  mmol/g) Cr(VI) to Cr(III) form. The range of  $K_d$  values to be expected within each of the 12 categories was estimated from the data listed in Table E.2. The variations of  $K_d$  values as a function of pH and DCB extractable iron as independent variables based on experimental data (Table E.2) is also shown as a 3-dimensional graph (Figure E.1). The graph indicates that soils with lower pH values and higher DCB extractable iron contents exhibit greater adsorption (higher  $K_d$ ) of Cr(VI). At higher pH values ( $>7$ ), Cr(VI) adsorption tends to be very low (very low  $K_d$  values) irrespective of DCB extractable iron content. Similarly, soils which contain very low DCB extractable iron, adsorb very little Cr(VI) (very low  $K_d$  values) irrespective of soil pH values.

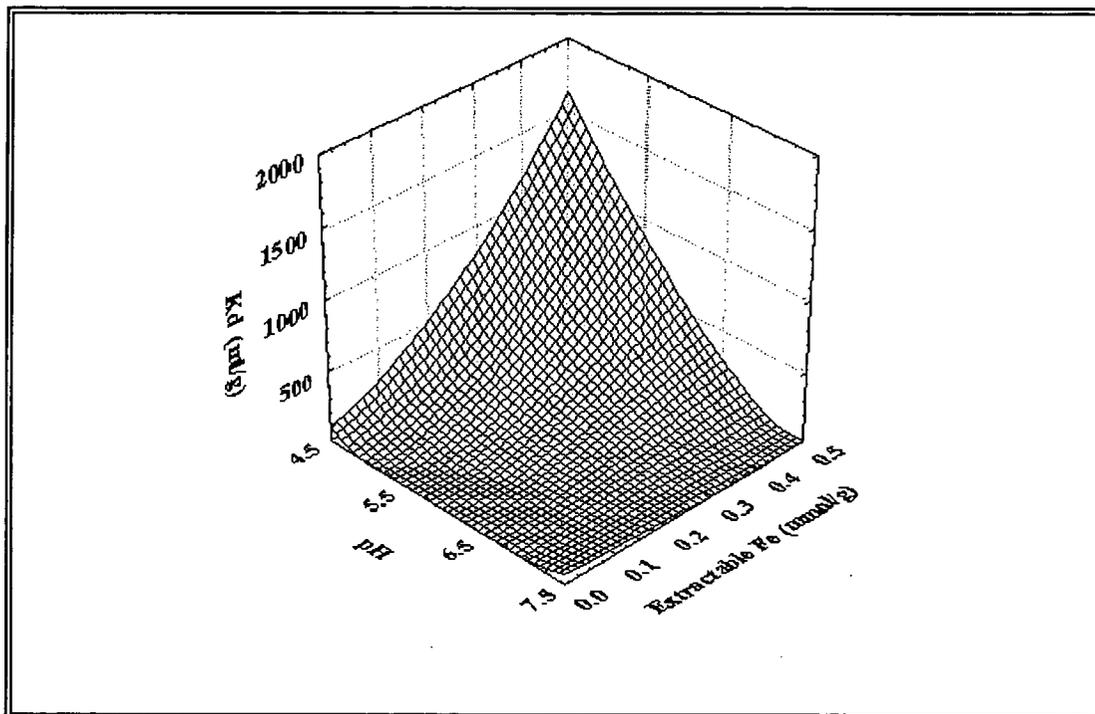
Additionally, Cr(VI) adsorption studies show that the presence of competing anions such as  $HPO_4^{2-}$ ,  $H_2PO_4^-$ ,  $SO_4^{2-}$ ,  $CO_3^{2-}$ , and  $HCO_3^-$  will reduce the  $K_d$  values as compared to a noncompetitive adsorption process. The only available data set that can be used to assess the competing anion effect was developed by Rai *et al.* (1988). However, they used fixed concentrations of competing anions namely  $SO_4^{2-}$ ,  $CO_3^{2-}$ , and  $HCO_3^-$  (fixed through a single selected partial pressure of  $CO_2$ ) concentrations (Tables E.4 and E.5). Among these competing anions,  $SO_4^{2-}$  at about 3 orders of magnitude higher concentrations ( $2 \times 10^{-3}$  M or 191.5 mg/l) than Cr(VI) concentration depressed Cr(VI)  $K_d$  values roughly by an order of magnitude as compared to noncompetitive adsorption. Therefore, the look-up table was developed on the assumption that  $K_d$  values of Cr(VI) would be reduced as soluble  $SO_4^{2-}$  concentrations increase from 0 to  $2 \times 10^{-3}$  M (or 191.5 mg/l).

Table E.2. Data from Rai *et al.* (1988) for the adsorption of Cr(VI) as a function of pH.

Kenoma Soil				Cecil/Pacolet Soil				Holton/Cloudland Soil				Ocala Soil			
pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)	pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)	pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)	pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)
8.42	3.03	6.25	1	9.26	3.05	5.66	2	9.84	3.03	6.33	1	9.37	3.02	6.56	0
7.71	3.05	5.84	2	9.29	3.05	5.88	1	8.67	3.04	6.11	1	9.40	3.03	6.05	1
7.70	3.04	5.97	1	8.57	3.11	5.34	6	8.60	3.08	5.60	3	8.94	3.02	7.71	0
7.35	3.09	5.54	4	7.80	3.30	5.00	20	8.29	3.09	5.53	4	8.94	3.02	6.67	0
7.40	3.08	5.59	3	7.41	3.44	4.89	35	8.27	3.07	5.70	2	8.67	3.04	6.00	1
7.20	3.03	5.36	5	7.38	3.46	4.88	38	8.08	3.11	5.45	5	8.61	3.02	6.36	0
7.16	3.13	5.37	6	6.99	3.66	4.81	71	7.55	3.30	5.04	18	8.33	3.04	6.00	1
6.89	3.16	5.27	8	6.94	3.65	4.81	69	7.15	3.44	4.92	33	8.30	3.03	6.07	1
6.92	3.15	5.29	7	6.67	3.79	4.78	102	7.05	3.51	4.89	42	7.56	3.03	6.14	1
6.70	3.23	5.13	13	6.49	3.79	4.78	102	6.96	3.60	4.85	56	7.53	3.02	6.48	0
6.47	3.26	5.09	15	6.19	3.99	4.75	174	6.88	3.61	4.85	58	7.53	3.02	6.86	0
6.02	3.36	4.98	24	6.16	3.94	4.75	155	6.26	4.26	4.74	331	7.07	3.03	6.25	1
6.02	3.35	4.99	23	5.89	4.08	4.74	219	6.20	4.25	4.74	324	7.18	3.03	6.19	1
5.61	3.39	4.95	28	5.84	4.06	4.74	209	5.40	4.55	4.73	661	6.92	3.03	6.21	1
5.62	3.40	4.95	28	5.46	4.19	4.73	288	5.39	4.63	4.73	794	6.88	3.02	6.48	0
				5.49	4.21	4.73	302	4.90	4.75	4.73	1047	6.61	3.03	6.12	1
				4.98	4.33	4.72	407	4.87	4.74	4.73	1023	5.71	3.02	6.68	0
				4.98	4.32	4.72	398	4.63	4.79	4.72	1175	5.14	3.04	6.01	1
				4.49	4.52	4.71	646	4.63	4.80	4.72	1202				
				4.49	4.39	4.72	468	4.51	4.85	4.72	1349				
								4.51	4.82	4.72	1259				
								4.50	4.88	4.72	1445				
								4.45	4.92	4.72	1585				

**Table E.3.** Estimated range of  $K_d$  values for Cr(VI) as a function of soil pH, extractable iron content, and soluble sulfate.

Soluble Sulfate Conc (mg/l)	$K_d$ (ml/g)	pH											
		4.1 - 5.0			5.1 - 6.0			6.1 - 7.0			≥7.1		
		DCB Extractable Fe (mmol/g)			DCB Extractable Fe (mmol/g)			DCB Extractable Fe (mmol/g)			DCB Extractable Fe (mmol/g)		
		≤0.25	0.26 - 0.29	≥0.30	≤0.25	0.26 - 0.29	≥0.30	≤0.25	0.26 - 0.29	≥0.30	≤0.25	0.26 - 0.29	≥0.30
0 - 1.9	Min	25	400	990	20	190	390	8	70	80	0	0	1
	Max	35	700	1770	34	380	920	22	180	350	7	30	60
2 - 18.9	Min	12	190	460	10	90	180	4	30	40	0	0	1
	Max	15	330	820	15	180	430	10	80	160	3	14	30
19 - 189	Min	5	90	210	4	40	80	2	15	20	0	0	0
	Max	8	150	380	7	80	200	5	40	75	2	7	13
≥190	Min	3	40	100	2	20	40	1	7	8	0	0	0
	Max	4	70	180	3	40	90	2	20	35	1	3	6



**Figure E.1.** Variation of  $K_d$  for Cr(VI) as a function of pH and DCB extractable iron content without the presence of competing anions.

### E.3.0 Data Set for Soils

The data set used to develop the look-up table is from the adsorption data collected by Rai *et al.* (1988). The adsorption data for Cr(VI) as a function of pH developed for 4 well-characterized soils were used to calculate the  $K_d$  values (Table E.2). All 4 soil samples were obtained from subsurface horizons and characterized as to their pH, texture, CEC, organic and inorganic carbon contents, surface areas, extractable (hydroxylamine hydrochloride, and DCB) iron, manganese, aluminum, and silica, KOH extractable aluminum and silica, and clay mineralogy. Additionally, Cr oxidizing and reducing properties of these soils were also determined (Rai *et al.*, 1988). Effects of competing anions such as sulfate and carbonate on Cr(VI) adsorption were determined for 2 of the soils (Cecil/Pacolet, and Kehoma). The  $K_d$  values from competitive anion experiments were calculated (Tables E.4 and E.5) and used in developing the look-up table (Table E.3).

**Table E.4.** Data from Rai *et al.* (1988) on effects of competing anions on Cr(VI) adsorption on Cecil/Pacolet soil.

Cr(VI) <sup>1</sup>				Cr(VI) + Sulfate <sup>1</sup>				Cr(VI) + Carbonate <sup>1</sup>			
pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)	pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)	pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)
9.26	3.05	5.66	2	8.92	3.05	6.27	1	9.62	3.05	6.88	0
9.29	3.05	5.88	1	8.38	3.07	5.71	2	9.15	3.05	6.79	0
8.57	3.11	5.34	6	8.38	3.04	5.70	2	9.01	3.06	6.35	1
7.80	3.30	5.00	20	7.70	3.12	5.28	7	7.92	3.06	6.12	1
7.41	3.44	4.89	35	7.67	3.12	5.28	7	7.95	3.06	6.10	1
7.38	3.46	4.88	38	7.37	3.19	5.11	12	7.53	3.08	5.85	2
6.99	3.66	4.81	71	7.24	3.23	5.09	14	7.52	3.07	6.06	1
6.94	3.65	4.81	69	6.85	3.34	4.95	24	7.19	3.12	5.55	4
6.67	3.79	4.78	102	6.76	3.37	4.96	26	7.31	3.10	5.67	3
6.49	3.79	4.78	102	6.58	3.43	4.92	32	7.22	3.12	5.55	4
6.19	3.99	4.75	174	6.56	3.34	4.95	25	6.99	3.13	5.48	4
6.16	3.94	4.75	155	6.15	3.55	4.85	50	6.70	3.22	5.21	10
5.89	4.08	4.74	219	6.15	3.51	4.88	43	6.68	3.21	5.24	9
5.84	4.06	4.74	209	5.75	3.58	4.82	58	5.84	3.65	4.87	60
5.46	4.19	4.73	288	5.79	3.56	4.86	51	6.08	3.54	4.91	43
5.49	4.21	4.73	302	5.35	3.60	4.83	59	5.12	4.11	4.78	214
4.98	4.33	4.72	407	5.33	3.59	4.84	57	5.12	4.14	4.78	229
4.98	4.32	4.72	398	4.68	3.55	4.86	49	4.76	4.20	4.78	263
4.49	4.52	4.71	646	4.69	3.47	4.86	41	4.75	4.11	4.78	214
4.49	4.39	4.72	468					4.33	4.39	4.76	427
								4.34	4.37	4.77	398

<sup>1</sup> Cr(VI) concentration: 10<sup>-6</sup> M, Sulfate Concentration: 10<sup>-2.7</sup> M, CO<sub>2</sub>: 10<sup>-1.6</sup> atm.

**Table E.5.** Data from Rai *et al.* (1988) on effects of competing anions on Cr(VI) adsorption on Kenoma soil.

Cr(VI) <sup>1</sup>				Cr(VI) + Sulfate + Carbonate <sup>1</sup>			
pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)	pH	-log C (mol/m <sup>3</sup> )	-log S (mol/kg)	K <sub>d</sub> (ml/g)
8.42	3.03	6.25	1	7.49	3.06	6.22	1
7.71	3.05	5.84	2	7.42	3.06	6.35	1
7.70	3.04	5.97	1	7.3	3.07	5.98	1
7.35	3.09	5.54	4	7.38	3.08	5.9	2
7.40	3.08	5.59	3	7.08	3.08	5.83	2
7.20	3.03	5.36	5	6.93	3.1	5.64	3
7.16	3.13	5.37	6	6.49	3.15	5.43	5
6.89	3.16	5.27	8	6.52	3.16	5.39	6
6.92	3.15	5.29	7	6.32	3.17	5.33	7
6.70	3.23	5.13	13	6.32	3.18	5.31	7
6.47	3.26	5.09	15	5.97	3.23	5.21	10
6.02	3.36	4.98	24	5.97	3.21	5.25	9
6.02	3.35	4.99	23	5.7	3.23	5.2	11
5.61	3.39	4.95	28	5.69	3.24	5.18	11
5.62	3.40	4.95	28	5.54	3.24	5.19	11
				5.52	3.25	5.18	12
				5.03	3.18	5.32	7
				5.02	3.21	5.26	9

Cr(VI) concentration: 10<sup>-6</sup> M, Sulfate Concentration: 10<sup>-2.7</sup> M, CO<sub>2</sub> : 10<sup>-1.6</sup> atm.

#### E.4.0 References

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**APPENDIX F**

**Partition Coefficients For Lead**

## Appendix F

### Partition Coefficients For Lead

#### F.1.0 Background

The review of lead  $K_d$  data reported in the literature for a number of soils led to the following important conclusions regarding the factors which influence lead adsorption on minerals, soils, and sediments. These principles were used to evaluate available quantitative data and generate a look-up table. These conclusions are:

- Lead may precipitate in soils if soluble concentrations exceed about 4 mg/l at pH 4 and about 0.2 mg/l at pH 8. In the presence of phosphate and chloride, these solubility limits may be as low as 0.3 mg/l at pH 4 and 0.001 mg/l at pH 8. Therefore, in experiments in which concentrations of lead exceed these values, the calculated  $K_d$  values may reflect precipitation reactions rather than adsorption reactions.
- Anionic constituents such as phosphate, chloride, and carbonate are known to influence lead reactions in soils either by precipitation of minerals of limited solubility or by reducing adsorption through complex formation.
- A number of adsorption studies indicate that within the pH range of soils (4 to 11), lead adsorption increases with increasing pH.
- Adsorption of lead increases with increasing organic matter content of soils.
- Increasing equilibrium solution concentrations correlates with decreasing lead adsorption (decrease in  $K_d$ ).

Lead adsorption behavior on soils and soil constituents (clays, oxides, hydroxides, oxyhydroxides, and organic matter) has been studied extensively. However, calculations by Rickard and Nriagu (1978) show that the solution lead concentrations used in a number of adsorption studies may be high enough to induce precipitation. For instance, their calculations show that lead may precipitate in soils if soluble concentrations exceed about 4 mg/l at pH 4 and about 0.2 mg/l at pH 8. In the presence of phosphate and chloride, these solubility limits may be as low as 0.3 mg/l at pH 4 and 0.001 mg/l at pH 8. Therefore, in experiments in which concentrations of lead exceed these values, the calculated  $K_d$  values may reflect precipitation reactions rather than adsorption reactions.

Based on lead adsorption behavior of 12 soils from Italy, Soldatini *et al.* (1976) concluded that soil organic matter and clay content were 2 major factors which influence lead adsorption. In these experiments, the maximum adsorption appeared to exceed the cation exchange capacity

(CEC) of the soils. Such an anomaly may have resulted from precipitation reactions brought about by high initial lead concentrations used in these experiments (20 to 830 mg/l).

Lead adsorption characteristics of 7 alkaline soils from India were determined by Singh and Sekhon (1977). The authors concluded that soil clay, organic matter, and the calcium carbonate influenced lead adsorption by these soils. However, the initial lead concentrations used in these experiments ranged from 5 to 100 mg/l, indicating that in these alkaline soils the dominant lead removal mechanism was quite possibly precipitation.

In another adsorption study, Abd-Elfattah and Wada (1981) measured the lead adsorption behavior of 7 Japanese soils. They concluded that soil mineral components which influenced lead adsorption ranged in the order: iron oxides>halloysite>imogolite, allophane>humus, kaolinite>montmorillonite. These data may not be reliable because high lead concentrations (up to 2,900 mg/l) used in these experiments may have resulted in precipitation reactions dominating the experimental system.

Anionic constituents, such as phosphate, chloride, and carbonate, are known to influence lead reactions in soils either by precipitation of minerals of limited solubility or by reducing adsorption through complex formation (Rickard and Nriagu, 1978). A recent study by Bargar *et al.* (1998) showed that chloride solutions could induce precipitation of lead as solid  $PbOHCl$ . Presence of synthetic chelating ligands such as ethylenediaminetetraacetic acid (EDTA) has been shown to reduce lead adsorption on soils (Peters and Shem, 1992). These investigators showed that the presence of strongly chelating EDTA in concentrations as low as 0.01 M reduced  $K_d$  for lead by about 3 orders of magnitude. By comparison quantitative data is lacking on the effects of more common inorganic ligands (phosphate, chloride, and carbonate) on lead adsorption on soils.

A number of adsorption studies indicate that within the pH range of soils (4 to 11), lead adsorption increases with increasing pH (Bittel and Miller, 1974; Braids *et al.*, 1972; Griffin and Shimp, 1976; Haji-Djafari *et al.*, 1981; Hildebrand and Blum, 1974; Overstreet and Krishnamurthy, 1950; Scudato and Estes, 1975; Zimdahl and Hassett, 1977). Griffin and Shimp (1976) also noted that clay minerals adsorbing increasing amounts of lead with increasing pH may also be attributed to the formation of lead carbonate precipitates which was observed when the solution pH values exceeded 5 or 6.

Solid organic matter such as humic material in soils and sediments are known to adsorb lead (Rickard and Nriagu, 1978; Zimdahl and Hassett, 1977). Additionally, soluble organic matter such as fulvates and amino acids are known to chelate soluble lead and affect its adsorption on soils (Rickard and Nriagu, 1978). Gerritse *et al.* (1982) examined the lead adsorption properties of soils as a function of organic matter content of soils. Initial lead concentrations used in these experiments ranged from 0.001 to 0.1 mg/l. Based on adsorption data, the investigators expressed  $K_d$  value for a soil as a function of organic matter content (as wt.%) and the distribution coefficient of the organic matter. The data also indicated that irrespective of soil organic matter content, lead adsorption increased with increasing soil pH (from 4 to 8). In

certain soils, lead is also known to form methyl-lead complexes (Rickard and Nriagu, 1978). However, quantitative relationship between the redox status of soils and its effect on overall lead adsorption due to methylation of lead species is not known.

Tso (1970), and Sheppard *et al.* (1989) studied the retention of  $^{210}\text{Pb}$  in soils and its uptake by plants. These investigators found that lead in trace concentrations was strongly retained on soils (high  $K_d$  values). Lead adsorption by a subsurface soil sample from Hanford, Washington was investigated by Rhoads *et al.* (1992). Adsorption data from these experiments showed that  $K_d$  values increased with decreasing lead concentrations in solution (from 0.2 mg/l to 0.0062 mg/l). At a fixed pH of 8.35, the authors found that  $K_d$  values were log-linearly correlated with equilibrium concentrations of lead in solution. Calculations showed that if lead concentrations exceeded about 0.207 mg/l, lead-hydroxycarbonate (hydrocerussite) would probably precipitate in this soil.

The  $K_d$  data described above are listed in Table F.1.

## F.2.0 Approach

The initial step in developing a look-up table consisted of identifying the key parameters which were correlated with lead adsorption ( $K_d$  values) on soils and sediments. Data sets developed by Gerritse *et al.* (1982) and Rhoads *et al.* (1992) containing both soil pH and equilibrium lead concentrations as independent variables were selected to develop regression relationships with  $K_d$  as the dependent variable. From these data it was found that a polynomial relationship existed between  $K_d$  values and soil pH measurements. This relationship (Figure F.1) with a correlation coefficient of 0.971 ( $r^2$ ) could be expressed as:

$$K_d \text{ (ml/g)} = 1639 - 902.4(\text{pH}) + 150.4(\text{pH})^2 \quad (\text{F.1})$$

The relationship between equilibrium concentrations of lead and  $K_d$  values for a Hanford soil at a fixed pH was expressed by Rhoads *et al.* (1992) as:

$$K_d \text{ (ml/g)} = 9,550 C^{-0.335} \quad (\text{F.2})$$

where  $C$  is the equilibrium concentration of lead in  $\mu\text{g/l}$ . The look-up table (Table F.2) was developed from using the relationships F.1 and F.2. Four equilibrium concentration and 3 pH categories were used to estimate the maximum and minimum  $K_d$  values in each category. The relationship between the  $K_d$  values and the 2 independent variables (pH and the equilibrium concentration) is shown as a 3-dimensional surface (Figure F.2). This graph illustrates that the highest  $K_d$  values are encountered under conditions of high pH values and very low equilibrium lead concentrations and in contrast, the lowest  $K_d$  values are encountered under lower pH and higher lead concentrations. The  $K_d$  values listed in the look-up table encompasses the ranges of pH and lead concentrations normally encountered in surface and subsurface soils and sediments.

**Table F.1.** Summary of  $K_d$  values for lead adsorption on soils.

Soil Description	Clay Content (wt.%)	Organic Carbon (wt.%)	Iron Oxide content (wt.%)	pH	CEC (meq/100g)	$K_d$ (ml/g)	Experimental Parameters	Reference
Sediment, Split Rock Formation, Wyoming	--	--	--	2.0	--	20	--	Haji-Djafari <i>et al.</i> , 1981
	--	--	--	4.5	--	100	--	
	--	--	--	5.75	--	1,500	--	
	--	--	--	7.0	--	4,000	--	
Sand (Soil C)	0	--	--	4.5	22	280	Batch Experiment	Gerritse <i>et al.</i> (1982)
Sand (Soil C)	0	--	--	5.0	22	1295	Batch Experiment	
Sandy Loam (Soil D)	2	--	--	7.5	16	3,000	Batch Experiment	
Sandy Loam (Soil D)	2	--	--	8.0	16	4,000	Batch Experiment	
Loam (Soil 2)	15	--	--	7.3	17	21,000	Batch Experiment	Sheppard <i>et al.</i> (1989)
Medium Sand (Soil 3)	2	--	--	4.9	5.8	19	Batch Experiment	
Organic soil (Soil 4)	<1	--	--	5.5	120	30,000	Batch Experiment	
Fine Sandy Loam (Soil 6)	11	--	--	7.4	8.7	59,000	Batch Experiment	
Sand (Hanford)	0.06	<0.01	0.41	8.35	5.27	13,000 - 79,000	Batch tracer studies (Initial activities 2.38 - 23.4 $\mu\text{Ci/l}$ )	Rhoads <i>et al.</i> (1992)

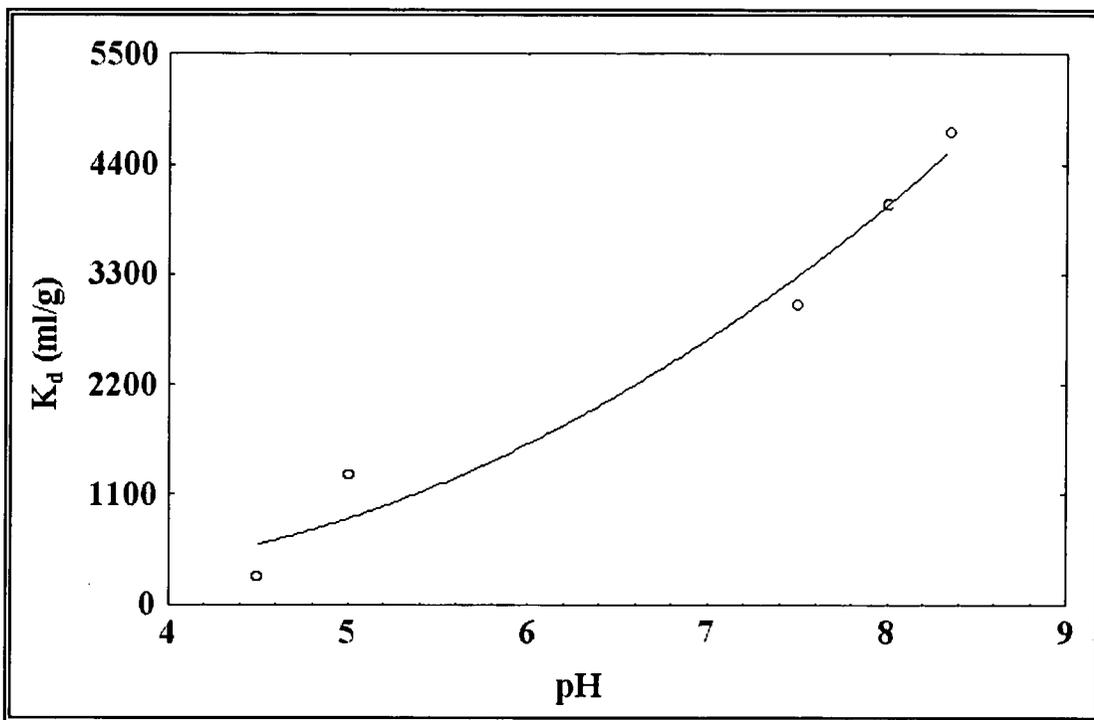


Figure F.1. Correlative relationship between  $K_d$  and pH.

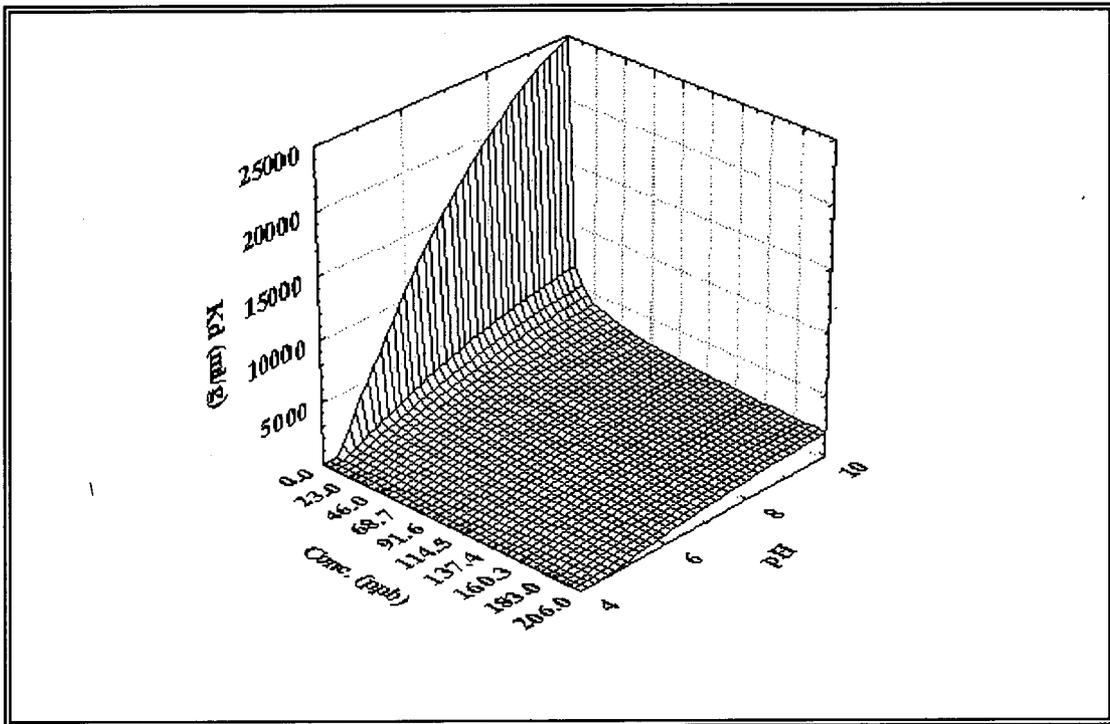


Figure F.2. Variation of  $K_d$  as a function of pH and the equilibrium lead concentrations.

### F.3.0 Data Set for Soils

The data sets developed by Gerritse *et al.* (1982) and Rhoads *et al.* (1992) were used to develop the look-up table (Table F.2). Gerritse *et al.* (1982) developed adsorption data for 2 well-characterized soils using a range of lead concentrations (0.001 to 0.1 mg/l) which precluded the possibility of precipitation reactions. Similarly, adsorption data developed by Rhoads *et al.* (1992) encompassed a range of lead concentrations from 0.0001 to 0.2 mg/l at a fixed pH value. Both these data sets were used for estimating the range of  $K_d$  values for the range of pH and lead concentration values found in soils.

**Table F.2.** Estimated range of  $K_d$  values for lead as a function of soil pH, and equilibrium lead concentrations.

Equilibrium Lead Concentration ( $\mu\text{g/l}$ )	$K_d$ (ml/g)	Soil pH		
		4.0 - 6.3	6.4 - 8.7	8.8 - 11.0
0.1 - 0.9	Minimum	940	4,360	11,520
	Maximum	8,650	23,270	44,580
1.0 - 9.9	Minimum	420	1,950	5,160
	Maximum	4,000	10,760	20,620
10 - 99.9	Minimum	190	900	2,380
	Maximum	1,850	4,970	9,530
100 - 200	Minimum	150	710	1,880
	Maximum	860	2,300	4,410

#### F.4.0 References

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## **APPENDIX G**

### **Partition Coefficients For Plutonium**

## Appendix G

### Partition Coefficients For Plutonium

#### G.1.0 Background

A number of studies have focussed on the adsorption behavior of plutonium on minerals, soils, and other geological materials. A review data from diverse literature sources indicated that  $K_d$  values for plutonium typically range over 4 orders of magnitude (Thibault *et al.*, 1990). Also, from these data a number of factors which influence the adsorption behavior of plutonium have been identified. These factors and their effects on plutonium adsorption on soils and sediments were used as the basis for generating a look-up table. These factors are:

- Typically, in many experiments, the oxidation state of plutonium in solution was not determined or controlled therefore it would be inappropriate to compare the  $K_d$  data obtained from different investigations.
- In natural systems with organic carbon concentrations exceeding  $\sim 10$  mg/kg, plutonium exists mainly in trivalent and tetravalent redox states. If initial plutonium concentrations exceed  $\sim 10^{-7}$  M, the measured  $K_d$  values would reflect mainly precipitation reactions and not adsorption reactions.
- Adsorption data show that the presence of ligands influence plutonium adsorption onto soils. Increasing concentrations of ligands decrease plutonium adsorption.
- If no complexing ligands are present plutonium adsorption increases with increasing pH (between 5.5 and 9.0).
- Plutonium is known to adsorb onto soil components such as aluminum and iron oxides, hydroxides, oxyhydroxides, and clay minerals. However, the relationship between the amounts of these components in soils and the measured adsorption of plutonium has not been quantified.

Because plutonium in nature can exist in multiple oxidation states (III, IV, V, and VI), soil redox potential would influence the plutonium redox state and its adsorption on soils. However, our literature review found no plutonium adsorption studies which included soil redox potential as a variable. Studies conducted by Nelson *et al.* (1987) and Choppin and Morse (1987) indicated that the oxidation state of dissolved plutonium under natural conditions depended on the colloidal organic carbon content in the system. Additionally, Nelson *et al.* (1987) also showed that plutonium precipitation occurred if the solution concentration exceeded  $10^{-7}$  M.

A number of investigators have examined potential adsorption of plutonium on minerals, soils, and other geological substrates. Earlier experiments conducted by Evans (1956), Tamura (1972), Van Dalen *et al.* (1975) showed that plutonium adsorption onto mineral surfaces was

influenced significantly by the type of mineral, the pH and mineral particle size. The reported values ranged from zero for quartz (Tamura, 1972) to 4,990 ml/g for montmorillonite (Evans, 1956). [The  $K_d$  for glauconite tabulated by Evans (1956) was listed as "infinite" (certainly greater than 5,000 ml/g), because the concentration of dissolved plutonium measured in the  $K_d$  defemination was below detection.] These  $K_d$  values are only qualitative because, the initial concentrations of plutonium used in these experiments were apparently high enough to induce precipitation of plutonium solid phases therefore, the observed phenomena was likely due to mainly precipitation and not adsorption. Second, the redox status of plutonium was unknown in these experiments thus these reported  $K_d$  values cannot be  $K_d$  readily compared to values derived from other experiments.

The importance of the plutonium redox status on adsorption was demonstrated by Bondiotti *et al.* (1975) who reported about 2 orders of magnitude difference in  $K_d$  values between hexavalent (250 ml/g) and tetravalent (21,000 ml/g) plutonium species adsorbing on to montmorillonite. Bondiotti *et al.* (1975) also demonstrated that natural dissolved organic matter (fulvic acid) reduces plutonium from hexavalent to tetravalent state thus potentially affecting plutonium adsorption in natural systems. Some of the earlier adsorption experiments also demonstrated that complexation of plutonium by various ligands significantly influences its adsorption behavior. Increasing concentrations of acetate (Rhodes, 1957) and oxalate (Bensen, 1960) ligands resulted in decreasing adsorption of plutonium. Adsorption experiments conducted more recently (Sanchez *et al.*, 1985) indicate that increasing concentrations of carbonate ligand also depresses the plutonium adsorption on various mineral surfaces.

Even though the adsorption behavior of plutonium on soil minerals such as glauconite (Evans, 1956), montmorillonite (Billon, 1982; Bondiotti *et al.*, 1975), attapulgite (Billon, 1982), and oxides, hydroxides, and oxyhydroxides (Evans, 1956; Charyulu *et al.*, 1991; Sanchez *et al.*, 1985; Tamura, 1972; Ticknor, 1993; Van Dalen *et al.*, 1975) has been studied, correlative relationships between the type and quantities of soil minerals in soils and the overall plutonium adsorption behavior of the soils have not been established.

Adsorption experiments conducted by Billon (1982) indicated  $K_d$  values for Pu(IV) ranging from about 32,000 to 320,000 ml/g (depending on pH) for bentonite or attapulgite as adsorbents. Because of relatively high initial concentrations of plutonium [ $1.7 \times 10^{-6}$  to  $4 \times 10^{-6}$  M of Pu(IV)] used in these experiments, it is likely that precipitation and not adsorption resulted in very high  $K_d$  values. Additional experiments conducted with Pu(VI) species on bentonite substrate resulted in  $K_d$  values ranging from about 100 to 63,100 ml/g when pH was varied from 3.1 to 7.52. The validity of these data are questionable because of high initial concentrations of plutonium used in these experiments may have induced precipitation of plutonium.

Experiments conducted by Ticknor (1993) showed that plutonium sorbed on goethite and hematite from slightly basic solutions [(pH: 7.5) containing high dissolved salts, but extremely low bicarbonate concentrations ( $8.2 \times 10^{-6}$  to  $2.9 \times 10^{-4}$  M)] resulted in distribution coefficients,  $K_d$ , ranging from 170 to 1,400 ml/g. According to Pius *et al.* (1995), significant removal of Pu(IV) from solutions containing 0.1 to 1 M concentrations of sodium carbonate was observed with alumina, silica gel, and hydrous titanium oxide as substrates. These investigators also noted that the presence of carbonate lowered the sorption distribution coefficient for these adsorbents.

However, even at 0.5 M carbonate, the coefficients were 60 ml/g, 1,300 ml/g, and 15,000 ml/g, respectively, for alumina, silica gel, and hydrous titanium oxide. In another study using bicarbonate solutions, the distribution coefficient for Pu(IV) sorption on alumina was lowered to about 30 ml/g at 0.5 M bicarbonate (Charyulu *et al.*, 1991). However, one should note that the initial concentrations of Pu(IV) used by these investigators ranged from  $8.4 \times 10^{-6}$  to  $4.2 \times 10^{-5}$  M, which means that the solutions were probably supersaturated with respect to  $\text{PuO}_2 \cdot x\text{H}_2\text{O}$  solid phase. Because of the experimental conditions used by Pius *et al.* (1995) and Charyulu *et al.* (1991), the principal mechanism of plutonium removal from solution could have been precipitation as easily as adsorption.

Barney *et al.* (1992) measured adsorption of plutonium from carbonate-free wastewater solutions onto commercial alumina adsorbents over a pH range of 5.5 to 9.0. Plutonium adsorption  $K_d$  values increased from about 10 ml/g at a pH of 5.5 to about 50,000 ml/g at a pH of 9.0. The slopes of the  $K_d$  compared to the pH curves were close to 1, which indicated that 1 hydrogen ion is released to the solution for each plutonium ion that is adsorbed on the alumina surface. This behavior is typical of adsorption reactions of multivalent hydrolyzable metal ions with oxide surfaces. Changing the initial concentration of plutonium from about  $10^{-9}$  to  $10^{-10}$  M did not affect the  $K_d$  values, which showed that plutonium precipitation was not significant in these tests. Also, the initial plutonium concentrations were below the measured solubility limits of plutonium hydroxide. This experiment demonstrated that in carbonate-free systems, plutonium would be adsorbed on alumina substrates.

Another study of adsorption of Pu(IV) and Pu(V) on goethite was conducted by Sanchez *et al.* (1985). The experimental conditions used by these investigators were evaluated for assessing whether the reaction being studied was indeed adsorption. The initial plutonium concentrations used in their experiments were  $10^{-10}$  and  $10^{-11}$  moles per liter. These concentrations are well below the equilibrium saturation levels for  $\text{PuO}_2 \cdot x\text{H}_2\text{O}$ . The equilibrating solutions used in these experiments contained salts such as  $\text{NaNO}_3$ ,  $\text{NaCl}$ ,  $\text{Na}_2\text{SO}_4$ , and  $\text{NaHCO}_3$  and did not contain any ionic constituents that may have potentially formed solid solution precipitates. Therefore, it is reasonably certain that the dominant reaction being studied was adsorption and not precipitation of pure or solid solution phases.

The Pu(IV) and (V) adsorption data obtained in 0.1 M  $\text{NaNO}_3$  electrolyte medium by Sanchez *et al.* (1985) indicated isotherms typical of metal and/or metal-like complex specie adsorption on substrate (Benjamin and Leckie, 1981). This indicated that Pu(IV) and Pu(V) adsorbed onto the ionized hydroxyl sites in the form of free ions and their hydrolytic species with metal ion and the metal-ion part of the complexes adsorbing onto the surface. The adsorption isotherms obtained at the higher initial concentration ( $10^{-10}$  M) of total soluble Pu(IV) and Pu(V) showed that the adsorption edges (pH value at which 50 percent adsorption occurs) increased towards a higher pH value, which is typical of the metal-like adsorption behavior of adsorbing species (Benjamin and Leckie, 1981). These data also showed that the adsorption edges for Pu(V) was shifted about 2 pH units higher as compared to the adsorption edges observed for Pu(IV), indicating that plutonium in the higher oxidation state (pentavalent) had lower adsorbing affinity as compared with tetravalent plutonium. This difference in adsorption was attributed to the fact that Pu(V) hydrolyzes less strongly than Pu(IV),

The Pu(IV) and Pu(V) adsorption data obtained in 0.1 M NaNO<sub>3</sub> media represents conditions where only free cations and the respective hydrolytic species are the adsorbing species. Extensive experimental observations have shown that, when present, strong complexing agents have a significant effect on the metal ion adsorption (Benjamin and Leckie, 1981). This modified adsorption behavior in the presence of complex-forming ligands is characterized by Benjamin and Leckie as ligand-like adsorption. Sanchez *et al.* (1985) also conducted experiments to examine the effect of dissolved carbonate (from 10 to 1,000 meq/l) on the adsorption of Pu(IV) and Pu(V) on goethite. Their adsorption data showed that at a fixed pH value of 8.6, increasing carbonate concentration beyond 100 meq/l greatly decreased the adsorption of plutonium in both oxidation states. These data demonstrated that practically no Pu(IV) or Pu(V) adsorption occurred on goethite when the total carbonate concentration approached 1,000 meq/l (0.5 M CO<sub>3</sub>). However, data collected by Glover *et al.* (1976) showed that, at very low concentrations of dissolved carbonate (*i.e.*, 0.1-6 meq/l) typically encountered in soils, adsorption of Pu(IV) increased with increasing dissolved carbonate concentration. These results indicate that Pu(IV) in these soils may adsorb in the form of PuHCO<sub>3</sub><sup>3+</sup> species.

Such complete suppression of Pu(IV) and Pu(V) adsorption was attributed to the presence of anionic plutonium-hydroxy carbonate species in solution and to the fact that goethite at this pH contains mainly negatively charged sites that have negligible affinity to adsorb anionic species. This adsorption behavior of Pu(IV) and Pu(V) in the presence of carbonate ions that form strong hydroxy carbonate complexes is typical of ligand-like adsorption of metal ions described by Benjamin and Leckie (1981). Ligand-like adsorption is described as adsorption of a metal-ligand complex that is analogous to adsorption of the free ligand species. Also, the metal-ligand complexes may not adsorb at all if these complexes are highly stable. These data clearly demonstrate that increasing total carbonate and hydroxyl solution concentrations significantly decrease Pu(IV) and Pu(V) on iron oxyhydroxide surfaces.

Similar suppression of adsorption of higher valence state actinides in the presence of carbonate and hydroxyl ions has been observed by a number of investigators. Some of these studies include adsorption of U(VI) on goethite (Hsi and Langmuir, 1985; Koehler *et al.*, 1992; Tripathi, 1984), ferrihydrite (Payne *et al.*, 1992), and clinoptilolite (Pabalan and Turner, 1992), and Np(V) adsorption on ferrihydrite, hematite, and kaolinite (Koehler *et al.*, 1992).

Some of the early plutonium adsorption experiments on soils were conducted by Rhodes (1957) and Prout (1958). Rhodes (1957) conducted plutonium adsorption experiments using a calcareous subsurface soil from Hanford as the adsorbent. The data indicated that adsorption varied as a function of pH ranging from 18 ml/g under highly acidic conditions to >1980 ml/g at highly alkaline conditions. These data are unreliable because initial plutonium concentration of  $6.8 \times 10^{-7}$  M used in these experiments may have resulted in precipitation of plutonium solid phases. Prout (1958) studied adsorption of plutonium in +3, +4, and +6 redox states on a Savannah River Plant soil as a function of pH. The calculated  $K_d$  ranged from <10 to >10,000 ml/g, ~100 to ~10,000 ml/g, and <10 to ~3,000 ml/g for Pu(III), Pu(IV), and Pu(VI) respectively. Maximum  $K_d$  values were observed between pH values of about 6.5 and 8.5. Because the initial concentrations of plutonium used in these experiments were about  $1 \times 10^{-6}$  M, precipitation reaction may have accounted for the observed removal of plutonium from solution phase.

Bondietti *et al.* (1975) conducted Pu(IV) adsorption studies with the clay fraction isolated from a silt loam soil as the adsorbent. The  $K_d$  values from these experiments were reported to be as high as  $1.04 \times 10^6$  and  $1.68 \times 10^5$  ml/g. Experiments conducted by Dahlman *et al.* (1976) also showed exceedingly high  $K_d$  value ( $3 \times 10^5$  ml/g) for Pu(IV) adsorption on clay fraction from a silt loam soil. In view of this anomalously high  $K_d$  value, the authors concluded that actual mechanism of plutonium removal from solution phase may have been the precipitation reaction.

Nishita *et al.* (1976) extracted plutonium from a contaminated clay loam soil with solutions ranging in pH from 1.21 to 13.25. The solution pH in these experiments were adjusted with nitric acid and sodium hydroxide. The calculated  $K_d$  from these experiments varied from 3.02 to 3,086 ml/g, with highest  $K_d$  values noted within the pH range of 4.7 to 7.1. In another set of experiments Nishita (1978) extracted plutonium from the same clay loam soil with acetate (a ligand which forms complexes with plutonium) containing extraction solutions. The pH values for these set of extractions ranged from 2.81 to 11.19. The calculated  $K_d$  values in this experiment ranged from 37 to 2,857 ml/g with highest  $K_d$  values being observed between pH values 8.6 to 9.7.

Plutonium adsorption on 14 soil samples obtained from 7 different U.S. Department of Energy (DOE) sites were studied by Glover *et al.* (1976). Initial concentrations of plutonium in these experiments were  $10^{-8}$ ,  $10^{-7}$ , and  $10^{-6}$  M, respectively. The observed  $K_d$  values ranged from 30 to 14,000 ml/g. It is likely that removal of plutonium observed under higher initial concentrations ( $10^{-7}$ , and  $10^{-6}$  M) may have been due to precipitation reactions and not from adsorption reactions.

Rodgers (1976) conducted plutonium adsorption studies on clay and silt fractions from a glacial till soil from DOE's Mound Facility in Ohio. He noted that  $K_d$  values ranged from about 50 to 166,700 ml/g. The highest  $K_d$  values were observed between pH values of 5 to 6.

The effects of strong chelating agents such as ethylenediaminetetraacetic acid (EDTA) and diethylenetriaminepentaacetic acid (DTPA) on Pu(IV) adsorption by 3 different soils were investigated by Relyea and Brown (1978). The soils used for the adsorption were a sand (Fuquay from South Carolina), a loamy sand (Burbank from Washington), and a silt loam (Muscatine from Illinois) with initial concentrations of Pu(IV) fixed at about  $5 \times 10^{-8}$  M. Without the chelating ligands, the  $K_d$  values were 316, 6,000, and 8,000 ml/g for the sand, the loamy sand, and the silt loam respectively. When  $10^{-3}$  M of EDTA was present in the matrix solution, the measured  $K_d$  values were 120, 94.5 and 338 ml/g for the sand, the loamy sand, and the silt loam respectively. These significant reductions in adsorption were attributed to the limited affinity of Pu-EDTA complexes to adsorb onto the soil mineral surfaces. Increasing the EDTA concentration by an order of magnitude resulted in reductions in  $K_d$  values from about 1 order (for silt loam) to 2 orders (for sand) of magnitude. Using a stronger chelating agent ( $10^{-3}$  M DTPA) resulted in very low  $K_d$  values (0.12 ml/g for sand, 1.06 ml/g for loamy sand, and 0.24 ml/g for silt loam) which were about 3 to 4 orders of magnitude smaller as compared to the values from chelate-free systems. The results obtained from desorption experiments (using EDTA and DTPA ligands) showed that the  $K_d$  values were 1 to 2 orders of magnitude higher than the values calculated from adsorption experiments leading to the conclusion that some fraction of plutonium in soil was specifically adsorbed (not exchangeable). These data showed

that Pu(IV) adsorption on soils would be significantly reduced if the equilibrating solutions contain strong chelating ligands, such as EDTA and DTPA.

The reduction of plutonium adsorption on soils by strong synthetic chelating agents was also confirmed by experiments conducted by Delegard *et al.* (1984). These investigators conducted tests to identify tank waste components that could significantly affect sorption of plutonium on 3 typical shallow sediments from the the DOE Hanford Site. They found that sorption was decreased by the chelating agents, 0.05 M EDTA and 0.1 M HEDTA (N-2-hydroxyethylethylenediaminetriacetate) but not by low concentrations of carbonate (0.05 M). Delegard's data also showed that roughly a twofold increase in ionic strength caused an order of magnitude decrease in plutonium adsorption.

Based on an adsorption study of plutonium on basalt interbed sediments from the vicinity of Hanford site, Barney (1984) reported a  $K_d$  value of about 500 ml/g. This relatively lower  $K_d$  value may have resulted from the relatively enhanced concentration of 215 mg/l of carbonate (a complex forming ligand) which was present in the groundwater used in the experiments. Later, sorption of plutonium in +4, +5, and +6 redox states on a Hanford Site shallow sediment was studied by Barney (1992) to elucidate any differences in rate and amount of adsorption of plutonium in different redox states. The initial plutonium concentrations used in these experiments varied between about  $10^{-11}$  to  $10^{-9}$  M with synthetic ground water as a background electrolyte. The data indicated that the  $K_d$  values ranged from 2,100 to 11,600, 2,700 to 4,600, and 1,000 to 4,600 ml/g for plutonium in +4, +5, and +6 redox states, respectively. The data also indicated that Pu(V) and Pu(VI) upon adsorption was reduced to the tetravalent state. In these experiments, the  $K_d$  data obtained at lower initial concentrations ( $\sim 1 \times 10^{-11}$  M) of plutonium are reliable because the dominant plutonium removal mechanism from solution was adsorption.

Using batch equilibration techniques, Bell and Bates (1988) measured  $K_d$  values for plutonium which ranged from 32 to 7,600 ml/g. The soils used in these experiments were obtained from the Sellafield and Drigg sites in England and their texture ranged from clay to sand. Ground water spiked with about  $2.1 \times 10^{-8}$  M of plutonium was used in these adsorption experiments. The data also showed that the adsorption of plutonium on these soils varied as a function of pH, with maximum adsorption occurring at a pH value of about 6.

A number of studies indicate that  $K_d$  values for plutonium adsorption on river, oceanic, and lake sediments range from about  $1 \times 10^3$  to  $1 \times 10^6$  ml/g. Duursma and coworkers calculated that  $K_d$  for marine sediments was about  $1 \times 10^4$  ml/g (Duursma and Eisma, 1973; Duursma and Gross, 1971; Duursma and Parsi, 1974). Studies by Mo and Lowman (1975) on plutonium-contaminated calcareous sediments in aerated and anoxic seawater medium yielded  $K_d$  values from  $1.64 \times 10^4$  to  $3.85 \times 10^5$  ml/g. Based on distribution of plutonium between solution and suspended particle phases in sea water, Nelson *et al.* (1987) calculated that for plutonium in oxidized states (V, VI), the  $K_d$  was  $\sim 2.5 \times 10^3$  ml/g, and  $\sim 2.8 \times 10^6$  ml/g for plutonium in reduced states (III, IV). Based on a number of observations of lake and sea water samples, Nelson *et al.* (1987) reported that  $K_d$  values for lake particulates ranged from 3,000 to  $4 \times 10^5$  ml/g, and for oceanic particulates ranged from  $1 \times 10^5$  to  $4 \times 10^5$  ml/g.

## G.2.0 Data Set for Soils

The most detailed data set on plutonium  $K_d$  measurements were obtained by Glover *et al.* (1976). These data set were based on 17 soil samples from 9 different sites that included 7 DOE sites. The characterization of the soil included measurements of CEC, electrical conductivity, pH and soluble carbonate of the soil extracts, inorganic and organic carbon content, and the soil texture (wt.% of sand, silt, and clay content). The textures of these soils ranged from clay to fine sand. Three different initial concentrations of plutonium ( $10^{-8}$ ,  $10^{-7}$ , and  $10^{-6}$  M) were used in these experiments. This data set is the most extensive as far as the determination of a number of soil properties therefore, it can be examined for correlative relationships between  $K_d$  values and the measured soil parameters. The data set generated at initial plutonium concentrations of  $10^{-8}$  M were chosen for statistical analyses because the data sets obtained at higher initial concentrations of plutonium may have been affected by precipitation reactions (Table G.1).

### **G.3.0 Approach and Regression Models**

The most detailed data set on plutonium  $K_d$  measurements were obtained by Glover *et al.* (1976). This data set was based on 17 soil samples from 9 different sites that included 7 DOE sites. The characterization of the soil included measurements of CEC, electrical conductivity, pH and soluble carbonate of the soil extracts, inorganic and organic carbon content, and the soil texture (wt.% of sand, silt, and clay content). The textures of these soils ranged from clay to fine sand. Three different initial concentrations of plutonium ( $10^{-8}$ ,  $10^{-7}$ , and  $10^{-6}$  M) were used in these experiments. This data set is the most extensive as far as the determination of a number of soil properties therefore, it can be examined for correlative relationships between  $K_d$  values and the measured soil parameters. The data set generated at an initial plutonium concentration of  $10^{-8}$  M was chosen for statistical analyses because the data sets obtained at higher initial concentrations of plutonium may have been confounded by precipitation reactions

In developing regression models, initially it is assumed that all variables are influential. However, based on theoretical considerations or prior experience with similar models, one usually knows that some variables are more important than others. As a first step, all the variables are plotted in a pairwise fashion to ascertain any statistical relationship that may exist between these variables. This is typically accomplished by the use of scatter diagrams in which the relationship of each variable with other variables is examined in a pair-wise fashion and displayed as a series of 2-dimensional graphs. This was accomplished by using the Statistica™ software. The variables graphed included the distribution coefficient ( $K_d$  in ml/g), pH, CEC (in meq/100g), electrical conductivity of soil extract (EC in mmhos/cm), dissolved carbonate concentration in soil extract (DCARB in meq/l), inorganic carbon content (IC as percent  $\text{CaCO}_3$ ), organic carbon content (OC as wt.%), and the clay content (CLAY as wt.%).

**Table G.1.** Plutonium adsorption data for soil samples. [Data taken from results reported by Glover *et al.* (1976) for measurements conducted at an initial plutonium concentrations of  $10^{-8}$  M.]

Soil Sample	$K_d$ (ml/g)	pH	CEC <sup>1</sup> (meq/100 g)	EC <sup>1</sup> (mmhos/cm)	DCARB <sup>1</sup> (meq/l)	IC % <sup>1</sup> CaCO <sub>3</sub>	OC <sup>1</sup> (% mass)	CLAY <sup>1</sup> (% mass)
CO-A	2,200	5.7	20.0	3.6	5.97	0.4	2.4	36
CO-B	200	5.6	17.5	0.4	0.97	0.3	3.4	22
CO-C	1,900	7.9	29.6	0.4	1.98	2.4	0.7	64
ID-A	1,700	7.8	15.5	0.5	2.71	17.2	0.8	34
ID-B	320	8.3	13.8	0.8	2.51	7.9	0.2	32
ID-C	690	8.0	8.2	1.0	2.52	5.2	0.3	23
ID-D	2,100	7.5	17.5	1.2	4.90	0.0	0.1	3
WA-A	100	8.0	6.4	0.9	2.60	0.6	0.3	14
WA-B	430	8.2	5.8	0.4	2.30	0.0	0.1	14
SC	280	5.4	2.9	0.4	0.50	0.2	0.7	20
NY	810	5.4	16.0	1.2	1.40	0.0	2.7	36
NM	100	6.4	7.0	1.7	2.80	0.2	0.7	18
AR-A	710	6.2	34.4	0.5	0.10	0.9	3.2	56
AR-B	80	4.8	3.8	0.4	0.10	0.7	0.6	9
AR-C	430	2.3	16.2	0.3	0.10	0.6	2.3	37
IL	230	3.6	17.4	0.5	0.10	0.7	3.6	16

<sup>1</sup> CEC: Cation exchange capacity; EC: Electrical conductivity; DCARB: Dissolved carbonate; IC: Inorganic carbon; OC: Organic carbon; CLAY: Soil clay content.

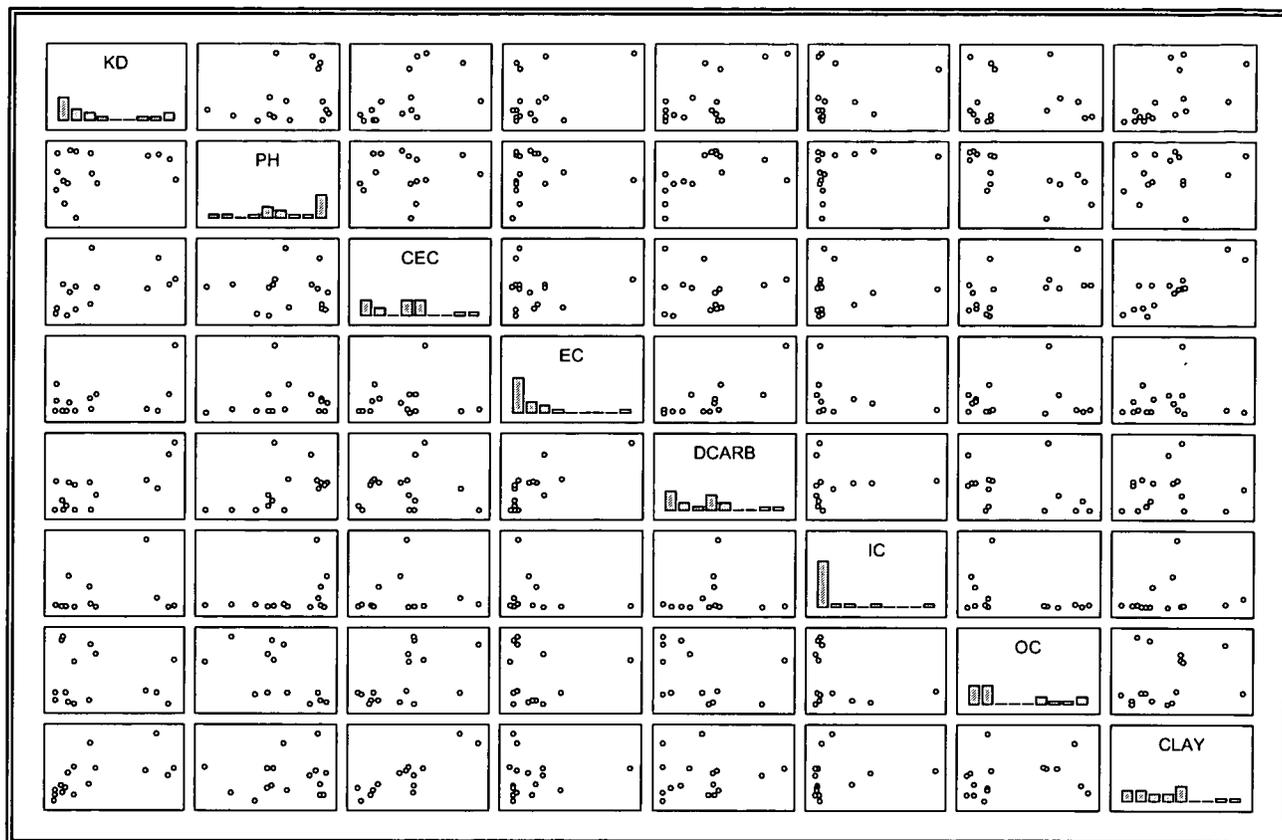
The scatterplots are typically displayed in a matrix format with columns and rows representing the dependent and independent variables respectively. For instance, the first row of plots shows the relationship between  $K_d$  as a dependent variable and other variables each in turn as selected as independent variables. Additionally, histograms displayed in each row illustrate the value distribution of each variable when it is being considered as the dependent variable.

The scatter matrix (Figure G.1) shows that regression relationships may exist between  $K_d$  and CEC, DCARB, and CLAY. Other relationships may exist between the CEC and CLAY, DCARB, and between PH, EC and DCARB. These relationships affirm that the CEC of soils depends mainly on the clay content. Similarly, the electrical conductivity of a soil solution depends on total concentrations of soluble ions and increasing dissolved carbonate concentration would contribute towards increasing EC. Also the pH of a soil solution would reflect the carbonate content of a soil with soils containing solid carbonate tending towards a pH value of  $\sim 8.3$ .

While a scatter diagram is a useful tool to initially assess the pairwise relationships between a number of variables, this concept cannot be extended to analyze multiple regression relationships (Montgomery and Peck, 1982). These authors point out that if there is 1 dominant regressive relationship, the corresponding scatter diagram would reveal this correlation. They also indicate however, that if several regressive relationships exist between a dependent variable and other independent variables, or when correlative relationships exist between independent variables themselves, the scatter diagrams cannot be used to assess multiple regressive relationships.

Typically, in regression model building, significant variables have to be selected out of a number of available variables. Montgomery and Peck (1982) indicate that regression model building involves 2 conflicting objectives. First, the models have to include as many independent variables as possible so that the influence of these variables on the predicted dependent variable is not ignored. Second, the regression model should include a minimum number of independent variables as possible so that the variance of predicted dependent variable is minimized.

Variable selection was conducted by using forward stepwise and backward stepwise elimination methods (Montgomery and Peck, 1982). In the forward stepwise method, each independent variable is added in a stepwise fashion until an appropriate model is obtained. The backward stepwise elimination method starts off by including all independent variables and in each step deletes (selects out) the least significant variables resulting in a final model which includes only the most influential independent variables.



**Figure G.1.** Scatter plot matrix of soil properties and the distribution coefficient ( $K_d$ ) of plutonium.

The variable selection with and without an intercept indicated that the 2 most significant variables for reliably forecasting the  $K_d$  values were the concentrations of dissolved carbonate (DCARB) and the clay content (CLAY) of soils (Table G.2). Using these 2 independent variables, several forms of polynomial regression models and a piecewise regression model with a breakpoint were generated. The results showed that the best regression model among all the models tested was the piecewise regression model. The relationship between the  $K_d$  values and the 2 independent variables (CLAY and DCARB) is shown as a 3-dimensional surface (Figure G.2). This graph illustrates that the highest  $K_d$  values are encountered under conditions of high clay content and dissolved carbonate concentrations. In contrast, the low  $K_d$  values are encountered in soils containing low clay content and low dissolved carbonate concentrations.

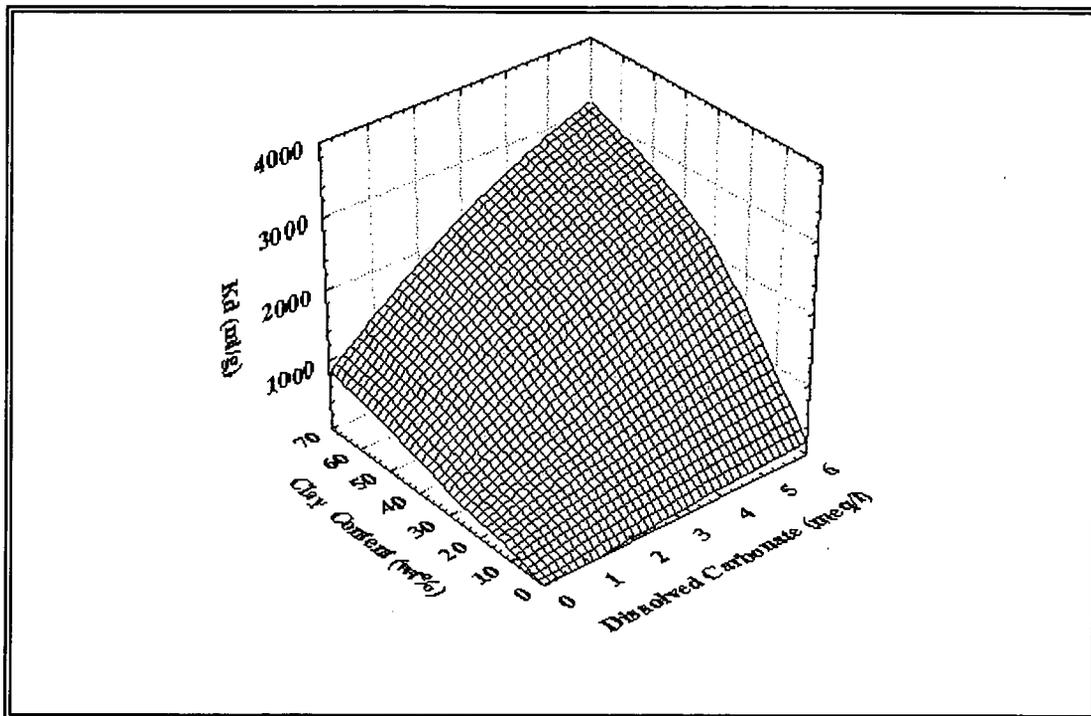
Using the piecewise regression model, a look-up table (Table G.3) was created for ranges of clay content and soluble carbonate values which are typically encountered in soils.

**Table G.2.** Regression models for plutonium adsorption.

Model Type	Forecasting Equation	R <sup>2</sup>
Linear Regression Forward Stepwise	$K_d = 284.6 (\text{DCARB}) + 27.8 (\text{CLAY}) - 594.2$	0.7305
Linear Regression Forward Stepwise	$K_d = 488.3 (\text{DCARB}) + 29.9 (\text{CLAY}) - 119.1 (\text{pH}) - 356.8 (\text{EC})$	0.8930
Linear Regression Backward Stepwise	$K_d = 284.6 (\text{DCARB}) + 27.8 (\text{CLAY}) - 594.2$	0.7305
Linear Regression Backward Stepwise	$K_d = 351.4 (\text{DCARB})$	0.7113
Piecewise Linear Regression	$K_d = 25.7 (\text{DCARB}) + 12.14 (\text{CLAY}) + 2.41$ for $K_d$ values <767.5 $K_d = 286.0 (\text{DCARB}) + 21.3(\text{CLAY}) - 81.2$ for $K_d$ values >767.5	0.9730
Polynomial	$K_d = -156.0 (\text{DCARB}) + 15.2 (\text{CLAY}) + 16.1 (\text{DCARB})^2 - 0.04 (\text{CLAY})^2 + 11.3 (\text{DCARB})(\text{CLAY}) - 87.0$	0.9222
Polynomial	$K_d = -171.1(\text{DCARB}) + 10.5 (\text{CLAY}) + 17.2(\text{DCARB})^2 + 0.02 (\text{CLAY})^2 + 11.6 (\text{DCARB})(\text{CLAY})$	0.9219
Polynomial	$K_d = -106.1(\text{DCARB}) + 11.2 (\text{CLAY}) + 12.5 (\text{DCARB})(\text{CLAY}) - 72.4$	0.9194
Polynomial	$K_d = -137.9 (\text{DCARB}) + 9.3 (\text{CLAY}) + 13.4 (\text{DCARB})(\text{CLAY})$	0.9190

**Table G.3.** Estimated range of  $K_d$  values for plutonium as a function of the soluble carbonate and soil clay content values.

$K_d$ (ml/g)	Clay Content (wt.%)								
	0 - 30			31 - 50			51 - 70		
	Soluble Carbonate (meq/l)			Soluble Carbonate (meq/l)			Soluble Carbonate (meq/l)		
	0.1 - 2	3 - 4	5 - 6	0.1 - 2	3 - 4	5 - 6	0.1 - 2	3 - 4	5 - 6
Minimum	5	80	130	380	1,440	2,010	620	1,860	2,440
Maximum	420	470	520	1,560	2,130	2,700	1,980	2,550	3,130



**Figure G.2.** Variation of  $K_d$  for plutonium as a function of clay content and dissolved carbonate concentrations.

#### G.4.0 References

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## **APPENDIX H**

### **Partition Coefficients For Strontium**

## Appendix H

### Partition Coefficients For Strontium

#### H.1.0 Background

Two simplifying assumptions underlying the selection of strontium  $K_d$  values included in the look-up table were made. These assumptions are that the adsorption of strontium occurs by cation exchange and follows a linear isotherm. These assumptions appear to be reasonable for a wide range of environmental conditions. However, these simplifying assumptions are compromised in systems with strontium concentrations greater than about  $10^{-4}$  M, humic substance concentrations greater than about 5 mg/l, ionic strengths greater than about 0.1 M, and pH levels greater than approximately 12.

Based on these assumptions and limitations, strontium  $K_d$  values and some important ancillary parameters that influence cation exchange were collected from the literature and tabulated in Section H.3. The tabulated data were from studies that reported  $K_d$  values (not percent adsorbed or Freundlich or Langmuir constants) and were conducted in systems consisting of

- Natural soils (as opposed to pure mineral phases)
- Low ionic strength ( $< 0.1$  M)
- pH values between 4 and 10
- Strontium concentrations less than  $10^{-4}$  M
- Low humic material concentrations ( $< 5$  mg/L)
- No organic chelates (such as EDTA)

The ancillary parameters included clay content, pH, CEC, surface area, solution calcium concentrations, and solution strontium concentrations. The table in Section H.3 describes 63 strontium  $K_d$  values. Strontium  $K_d$  values for soils as well as pure mineral phases are tabulated in Section H.4. This table contains 166 entries, but was not used to provide guidance regarding the selection of  $K_d$  values to be included in the look-up table.

Statistical analysis were conducted with the data collected from the literature. These analyses were used as guidance for selecting appropriate  $K_d$  values for the look-up table. The  $K_d$  values used in the look-up tables could not be based entirely on statistical consideration because the statistical analysis results were occasionally nonsensical. For instance, negative  $K_d$  values were predicted by 1 regression analysis. Thus, the  $K_d$  values included in the look-up table were not selected purely by objective reasoning. Instead, the statistical analysis was used as a tool to provide guidance for the selection of the approximate range of values to use and to identify meaningful trends between the strontium  $K_d$  values and the soil parameters.

The descriptive statistics of the strontium  $K_d$  data set for soil data only (entire data set presented in Section H.3) is presented in Table H.1. The 63 strontium  $K_d$  values in this data set ranged

from 1.6 ml/g for a measurement made on a sandy soil dominated by quartz (Lieser *et al.*, 1986) to 10,200 ml/g for a measurement made on a tuff<sup>1</sup> soil collected at Yucca Mountain, Nevada (Sample YM-38; Vine *et al.*, 1980). The average strontium  $K_d$  value was  $355 \pm 184$  ml/g. The median<sup>2</sup> strontium  $K_d$  value was 15.0 ml/g. This is perhaps the single central estimate of a strontium  $K_d$  value for this data set.

**Table H.1.** Descriptive statistics of strontium  $K_d$  data set for soils.

	Sr $K_d$ (ml/g)	Clay Content (wt.%)	pH	CEC (meq/100 g)	Surface Area (m <sup>2</sup> /g)	Ca (mg/l)
Mean	355	7.1	6.8	4.97	1.4	56
Standard Error	183	1.1	0.21	1.21	0	23
Median	15	5	6.7	0.9	1.4	0
Mode	21	5	6.2	2	1.4	0
Standard Deviation	1,458	7.85	1.35	9.66	0.00	134
Kurtosis	34	10.7	-0.5	11.6	-3	3.4
Minimum	1.6	0.5	3.6	0.05	1.4	0.00
Maximum	10,200	42.4	9.2	54	1.4	400
Number of Observations	63	48	42	63	7.00	32

<sup>1</sup> Tuff is a general name applied to material dominated by pyroclastic rocks composed of particles fragmented and ejected during volcanic eruptions.

<sup>2</sup> The median is that value for which 50 percent of the observations, when arranged in order of magnitude, lie on each side.

## H.2.0 Approach and Regression Models

### H.2.1 Correlations with Strontium $K_d$ Values

A matrix of the correlation coefficients of the strontium  $K_d$  values and soil parameters are presented in Table H.2. The correlation coefficients significant at or less than the 5 percent level of probability ( $P \leq 0.05$ ) are identified in Table H.2. The highest correlation coefficient with strontium  $K_d$  values was with CEC ( $r = 0.84$ ). Also significant are the correlation coefficients between strontium  $K_d$  values and clay content ( $r = 0.82$ ) and CEC and clay content ( $r = 0.91$ ) (Table H.2).

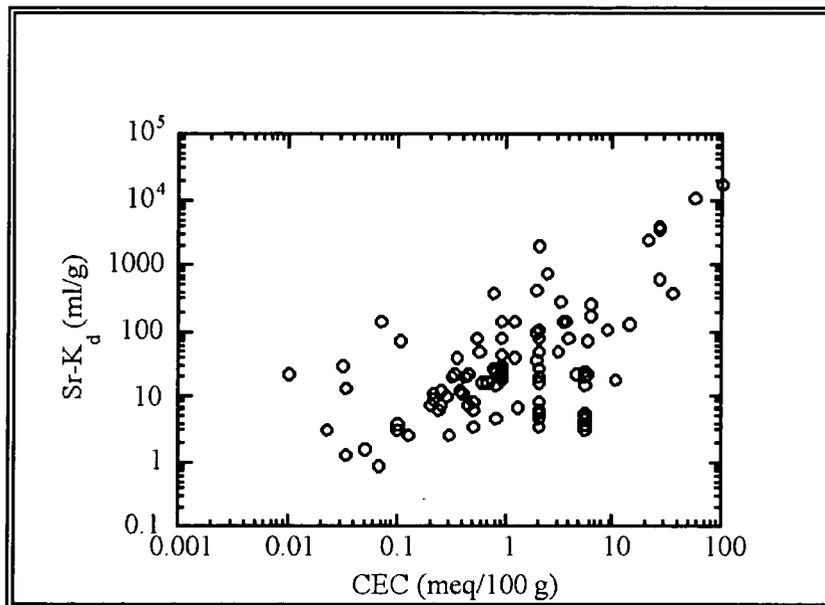
### H.2.2 Strontium $K_d$ Values as a Function of CEC and pH

The CEC and strontium  $K_d$  data are presented in Figure H.1. It should be noted that a logarithmic scale was used for the y-axis to assist in the visualization of the data and is not meant to suggest any particular model. A great deal of scatter exists in this data, especially in the lower CEC range where more data exist. For example, between the narrow CEC range of 5.5 to 6.0 meq/100 g, 9 strontium  $K_d$  values are reported (Keren and O'Connor, 1983; McHenry, 1958; Serne *et al.*, 1993). The strontium  $K_d$  values range from 3 ml/g for a surface noncalcareous sandy loam collected from New Mexico (Keren and O'Connor, 1983) to 70 ml/g for a carbonate surface soil collected from Washington (McHenry, 1958). Thus, over an order of magnitude variability in strontium  $K_d$  values may be expected at a given CEC level.

**Table H.2.** Correlation coefficients ( $r$ ) of the strontium  $K_d$  data set for soils.

	Strontium $K_d$	Clay Content	pH	CEC	Surface Area	Ca Conc.
Strontium $K_d$	1.00					
Clay Content	0.82 <sup>1</sup>	1.00				
pH	0.28	0.03	1.00			
CEC	0.84 <sup>1</sup>	0.91 <sup>1</sup>	0.28 <sup>1</sup>	1.00		
Surface Area	0.00	-1.00	0.00	1.00 <sup>1</sup>	1.00	
Ca Conc.	-0.17	0.00	-0.20	0.03	---	1.00

<sup>1</sup> Correlation coefficients significant at or less than the 5% level of probability ( $P \leq 0.05$ ).



**Figure H.1.** Relation between strontium  $K_d$  values and CEC in soils.

Another important issue regarding this data set is that 83 percent of the observations exists at CEC values less than 15 meq/100 g. The few  $K_d$  values associated with CEC values greater than 15 meq/100 g may have had a disproportionately large influence on the regression equation calculation (Neter and Wasserman, 1974). Consequently, estimates of strontium  $K_d$  values using these data for low CEC soils, such as sandy aquifers, may be especially inaccurate.

The regression equation for the data in Figure H.1 is presented as Equation 1 in Table H.3. Also presented in Table H.3 are the 95 percent confidence limits of the calculated regression coefficients, the y-intercepts, and slopes. These coefficients, when used to calculate  $K_d$  values, suggest a  $K_d$  range at a given CEC by slightly over an order of magnitude. The lower 95 percent confidence limit coefficients can provide guidance in selecting lower (or conservative)  $K_d$  values.

The large negative intercept in Equation 1 compromises its value for predicting strontium  $K_d$  values in low CEC soils, a potentially critical region of the data, because many aquifers matrix have low CEC values. At CEC values less than 2.2 meq/100 g, Equation 1 yields negative

strontium  $K_d$  values, which are clearly unrealistic.<sup>1</sup> To provide a better estimate of strontium  $K_d$  values at low CEC values, 2 approaches were evaluated. First, the data in Figure H.1 was reanalyzed such that the intercept of the regression equation was set to zero, *i.e.*, the regression equation was forced through the origin. The statistics of the resulting regression analysis are presented as Equation 2 in Table H.3. The coefficient of determination ( $R^2$ ) for Equation 2 slightly decreased compared to Equation 1 to 0.67 and remained highly significant ( $F = 2 \times 10^{-16}$ ). However, the large value for the slope resulted in unrealistically high strontium  $K_d$  values. For example at 1 meq/100 g, Equation 2 yields a strontium  $K_d$  value of 114 ml/g, which is much greater than the actual data presented in Figure H.1.

The second approach to improving the prediction of strontium  $K_d$  values at low CEC was to limit the data included in the regression analysis to those with CEC less than 15 meq/100 g. These data are redrawn in Figure H.2. The accompanying regression statistics with the y-intercept calculated and forced through the origin are presented in Table H.3 as Equations 3 and 4, respectively. The regression equations are markedly different from their respective equations describing the entire data set, Equations 1 and 2. Not surprisingly, the equations calculate strontium  $K_d$  more similar to those in this reduced data set. Although the coefficients of determination for Equations 3 and 4 decreased compared to those of Equations 1 and 2, they likely represent these low CEC data more accurately.

Including both CEC and pH as independent variables further improved the predictive capability of the equation for the full data set as well as the data set for soils with CEC less than 15 meq/100 g (Equations 5 and 6 in Table H.3). Multiple regression analyses with additional parameters did not significantly improve the model (results not presented).

### ***H.2.3 Strontium $K_d$ Values as a Function of Clay Content and pH***

Because CEC data are not always available to contaminant transport modelers, an attempt was made to use independent variables in the regression analysis that are more commonly available to modelers. Multiple regression analysis was conducted using clay content and pH as independent variables to predict CEC (Equations 7 and 8 in Table H.3) and strontium  $K_d$  values (Equations 9 and 10 in Table H.3; Figures H.3 and H.4). The values of pH and clay content were highly correlated to soil CEC for the entire data set ( $R^2 = 0.86$ ) and for those data limited to CEC less than 15 meq/100 g ( $R^2 = 0.57$ ). Thus, it is not surprising that clay content and pH were correlated to strontium  $K_d$  values for both the entire data set and for those associated with CEC less than 15 meq/100 g.

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<sup>1</sup> A negative  $K_d$  value is physically possible and is indicative of the phenomena referred to as anion exclusion or negative adsorption. It is typically and commonly associated with anions being repelled by the negative charge of permanently charged minerals.



**Table H.3.** Simple and multiple regression analysis results involving strontium  $K_d$  values, cation exchange capacity (CEC; meq/100 g), pH, and clay content (percent).

#	Equation	n <sup>2</sup>	Data Range <sup>3</sup>	95% Confidence Limits <sup>1</sup>						R <sup>2</sup> <sup>4</sup>	F Value <sup>5</sup>
				Intercept		Slope First Independent Parameter		Slope Second Independent Parameter			
				Lower	Upper	Lower	Upper	Lower	Upper		
1	$K_d = -272 + 126(\text{CEC})$	63	All	-501	-43	105	147	---	---	0.70	$1 \times 10^{-17}$
2	$K_d = 114(\text{CEC})$	63	All	---	---	95	134	---	---	0.67	$2 \times 10^{-16}$
3	$K_d = 10.0 + 4.05(\text{CEC})$	57	CEC<15	3.32	16.6	2.13	5.96	---	---	0.25	$9 \times 10^{-5}$
4	$K_d = 5.85(\text{CEC})$	57	CEC<15	---	---	4.25	7.44	---	---	0.12	$7 \times 10^{-3}$
5	$K_d = -42 + 14(\text{CEC}) + 2.33(\text{pH})$	27	All	-176	91	11.3	18.3	-17.7	22.4	0.77	$3 \times 10^{-8}$
6	$K_d = 3.53(\text{CEC}) + 1.67(\text{pH})$	25	CEC<15	---	---	0.62	6.46	-0.50	3.85	0.34	$9 \times 10^{-3}$
7	$\text{CEC} = -4.45 + 0.70(\text{clay}) + 0.60(\text{pH})$	27	All	-10.6	1.67	0.59	0.82	-0.30	1.50	0.86	$4 \times 10^{-11}$
8	$\text{CEC} = 0.40(\text{clay}) + 0.19(\text{pH})$	25	CEC<15	---	---	0.24	0.56	-0.01	0.40	0.55	$1 \times 10^{-4}$
9	$K_d = -108 + 10.5(\text{clay}) + 11.2(\text{pH})$	27	All	-270	53.3	7.32	13.6	-12.5	34.9	0.67	$2 \times 10^{-6}$
10	$K_d = 3.54(\text{clay}) + 1.67(\text{pH})$	25	CEC<15	---	---	0.62	6.46	-0.50	3.85	0.34	$9 \times 10^{-3}$
11	$\text{Clay} = 3.36 + 1.12(\text{CEC})$	48	All	2.30	4.41	0.97	1.26	---	---	0.84	$1 \times 10^{-19}$
12	$\text{Clay} = 1.34(\text{CEC})$	48	All	---	---	1.16	1.51	---	---	0.69	$2 \times 10^{-13}$

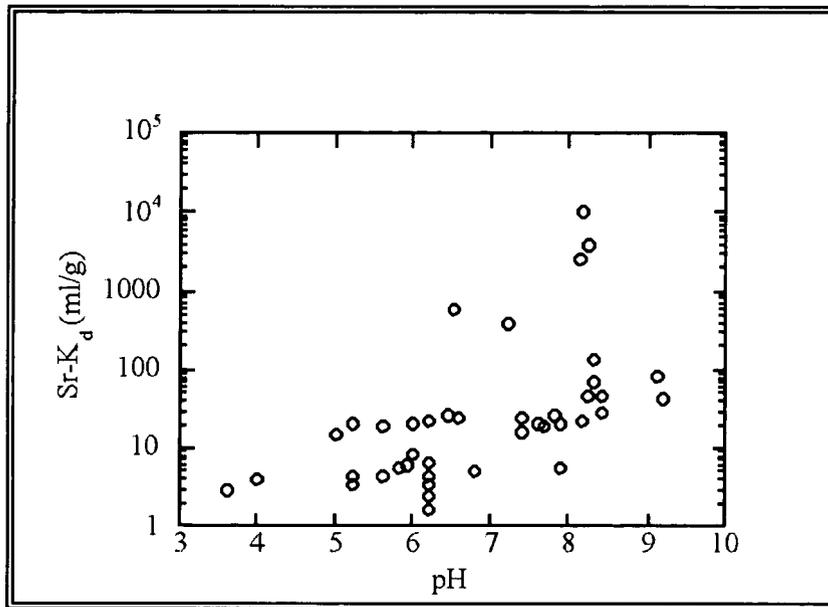
<sup>1</sup> The 95% confidence limits provides the range within which one can be 95% confident that the statistical parameter exist.

<sup>2</sup> The number of observations in the data set.

<sup>3</sup> All available observations were included in regression analysis except when noted.

<sup>4</sup> R<sup>2</sup> is the coefficient of determination and represents the proportion of the total treatment sum of squares accounted for by regression (1.00 is a perfect match between the regression equation and the data set).

<sup>5</sup> The F factor is a measure of the statistical significance of the regression analysis. The acceptable level of significance is not standardize and varies with the use of the data and the discipline. Frequently, a regression analysis with a F value of less than 0.05 is considered to describe a significant relationship.



**Figure H.4.** Relation between strontium  $K_d$  values and soil pH.

#### **H.2.4 Approach**

Two strontium  $K_d$  look up tables were created. The first table requires knowledge of the CEC and pH of the system in order to select the appropriate strontium  $K_d$  value (Table H.4). The second table requires knowledge of the clay content and pH to select the appropriate strontium  $K_d$  value (Table H.5).

A full factorial table was created that included 3 pH categories and 3 CEC categories. This resulted in 9 cells. Each cell contained a range for the estimated minimum- and maximum  $K_d$  values. A 2 step process was used in selecting the appropriate  $K_d$  values for each cell. For the first step, the appropriate equations in Table H.3 were used to calculate  $K_d$  values. The lower and upper 95 percent confidence limit coefficients were used to provide guidance regarding the minimum and maximum  $K_d$  values. For the 2 lowest CEC categories, Equation 6 in Table H.3 was used. For the highest CEC category, Equation 5 was used. For the second step, these calculated values were adjusted by “eye balling the data” to agree with the data in Figures H.2-H.4. It is important to note that some of the look-up table categories did not have any actual observations, *e.g.*, pH < 5 and CEC = 10 to 50 meq/100 g. For these categories, the regression analysis and the values in adjacent categories were used to assist in the  $K_d$  selection process.

**Table H.4.** Look-up table for estimated range of  $K_d$  values for strontium based on CEC and pH. [Tabulated values pertain to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength (< 0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.]

$K_d$ (ml/g)	CEC (meq/100 g)								
	3			3 - 10			10 - 50		
	pH			pH			pH		
	< 5	5 - 8	8 - 10	< 5	5 - 8	8 - 10	< 5	5 - 8	8 - 10
Minimum	1	2	3	10	15	20	100	200	300
Maximum	40	60	120	150	200	300	1,500	1,600	1,700

**Table H.5.** Look-up table for estimated range of  $K_d$  values for strontium based on clay content and pH. [Tabulated values pertain to systems consisting of natural soils (as opposed to pure mineral phases), low ionic strength (< 0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.]

$K_d$ (ml/g)	Clay Content (wt.%)								
	< 4%			4 - 20%			20 - 60%		
	pH			pH			pH		
	< 5	5 - 8	8 - 10	< 5	5 - 8	8 - 10	< 5	5 - 8	8 - 10
Minimum	1	2	3	10	15	20	100	200	300
Maximum	40	60	120	150	200	300	1,500	1,600	1,700

A second look-up table (Table H.5) was created from the first look-up table in which clay content replaced CEC as an independent variable. This second table was created because it is likely that clay content data will be more readily available for modelers than CEC data. To accomplish this, clay contents associated with the CEC values used to delineate the different categories were calculated using regression equations; Equation 11 was used for the high category (10 to 50 meq/100 g) and Equation 10 was used for the 2 lower CEC categories. The results of these calculations are presented in Table H.6. It should be noted that, by using either Equation 11 or 12, the calculated clay content at 15 meq/100 g of soil equaled 20 percent clay.

**Table H.6.** Calculations of clay contents using regression equations containing cation exchange capacity as a independent variable.

Equation <sup>1</sup>	Y-Intercept	Slope	CEC (meq/100 g)	Clay Content (%)
12	---	1.34	3	4
12	---	1.34	15	20
11	3.36	1.1.2	15	20
11	3.36	1.12	50	59

<sup>1</sup> Number of equation in Table H.3.

### H.3.0 $K_d$ Data Set for Soils

Table H.7 lists the available  $K_d$  values identified for experiments conducted with only soils. The  $K_d$  values are listed with ancillary parameters that included clay content, pH, CEC, surface area, solution calcium concentrations, and solution strontium concentrations.

**Table H.7.** Strontium  $K_d$  data set for soils.

Sr $K_d$ (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] ppm	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> , Comments
21	0.8	5.2	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4e2Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
19	0.8	5.6	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4e2Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
22	0.8	6.2	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4e2Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
26	0.8	6.45	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4e2Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
24	0.8	6.6	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4e2Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
30	0.8	8.4	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4e2Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
43	0.8	9.2	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4e2Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
21.4	5		0.47				Groundwater		2
25	5		0.83				Groundwater		2, CEC was estimated by adding exch. Ca,Mg,K
12.7	5		0.39				Groundwater		2, GW = 7.4Ca, 1.7Mg, 2.2Na,5.6Cl, 18ppmSO <sub>4</sub>
7.9	5		0.46				Groundwater		2, Aquifer sediments
15.6	5		0.81				Groundwater		Chalk River Nat'l Lab, Ottawa, Canada
9.4	5		0.21				Groundwater		2, Described as sand texture
7.6	5		0.25				Groundwater		2, Assumed 5% clay, mean [clay] in sandy soils
6.4	5		0.24				Groundwater		2
7.7	5		0.26				Groundwater		2
28.1	5		0.76				Groundwater		2
7.63	5		0.26				Groundwater		2

Sr $K_d$ (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] ppm	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> , Comments
11.4	5		0.41				Groundwater		2
20.1	5		0.44				Groundwater		2
13	5		0.25				Groundwater		2
9.8	5		0.29				Groundwater		2
11	5		0.22				Groundwater		2
13	5		0.39				Groundwater		2
7.8	5		0.2				Groundwater		2
3.8	5		0.1				Groundwater		2
3	5		0.1				Groundwater		2
2.5	5		0.13				Groundwater		2
4	10	4	5.5		0	1x10 <sup>-8</sup> M	0.01M NaCl	Puye soil-Na	3
15	10	5	5.5		0	1x10 <sup>-8</sup> M	0.01M NaCl	Puye soil-Na	3, Noncalcareous soils
21	10	6	5.5		0	1x10 <sup>-8</sup> M	0.01M NaCl	Puye soil-Na	3
24	10	7.4	5.5		0	1x10 <sup>-8</sup> M	0.01M NaCl	Puye soil-Na	3
3	10	3.6	5.5		400	1x10 <sup>-8</sup> M	0.01M CaCl	Puye soil-Ca	3
4.5	10	5.2	5.5		400	1x10 <sup>-8</sup> M	0.01M CaCl	Puye soil-Ca	3
5.2	10	6.8	5.5		400	1x10 <sup>-8</sup> M	0.01M CaCl	Puye soil-Ca	3
5.7	10	7.9	5.5		400	1x10 <sup>-8</sup> M	0.01M CaCl	Puye soil-Ca	3
3.5		5.2	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
4.6		5.6	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4, Carbonate system
5.8		5.8	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
6.1		5.9	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
8.3		6	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4

Sr $K_d$ (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] ppm	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> , Comments
17		7.4	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
21		7.6	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
27		7.8	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
47		8.4	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
81		9.1	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	4
19.1	4	7.66	10.4		129	100 μCi/l	Hanford Groundwater	cgs-1	5
21.5	6	7.87	5.9		58.5	100 μCi/l	Hanford Groundwater	trench-8	5, Groundwater pH = 8.3
23.2	5	8.17	4.57		35.1	100 μCi/l	Hanford Groundwater	tbs-1	5, Hanford, Richland, Washington surface and subsurface sediments
48.5		8.24	3			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-22	6, Los Alamos, New Mexico
10,200		8.17	54			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-38	6, Yucca Mountain tuff sediments
2,500		8.13	21			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM48	6, Approximate initial pH, final pH are presented
3,790		8.24	27			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-49	6, Final pH 8.1- 8.5
3,820		8.24	27			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-50	6, Sediments = 106-500 μm fractions
1.6	0.5	6.2	0.05			10x10 <sup>-6</sup> M	Groundwater	Sediments	7
2.6	3	6.2	0.3			10x10 <sup>-6</sup> M	Groundwater	Sediments	7, Added kaolinite to sand
3.4	5	6.2	0.5			10x10 <sup>-6</sup> M	Groundwater	Sediments	7, CEC estimated based on kaolinite = 10 meq/100 g
4.6	8	6.2	0.8			10x10 <sup>-6</sup> M	Groundwater	Sediments	7
6.7	13	6.2	1.3			10x10 <sup>-6</sup> M	Groundwater	Sediments	7
400	42.4	7.2	34		0		Water	Ringhold Soil	8, soil from Richland, Washington

Sr $K_d$ (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] ppm	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> , Comments
135	26.9	8.3	13.6		0		Water	Bowdoin Soil	8, soil from Montana
600	33.5	6.5	26.3		0		Water	Hall soil	8, soil from Nebraska
70	3.5	8.3	5.8		0		Water	Composite Soil	8, soil from Hanford Site, Richland, Washington

<sup>1</sup> References: 1 = Ohnuki, 1994, 2 = Patterson and Spoel, 1981; 3 = Keren and O'Connor, 1983; 4 = Rhodes and Nelson, 1957; 5 = Serne *et al.*, 1993; 6 = Vine *et al.*, 1980; 7 = Lieser and Steinkopff, 1989; 8 = McHenry, 1958

#### H.4.0 $K_d$ Data Set for Pure Mineral Phases and Soils

Table H.8 lists the available  $K_d$  values identified for experiments conducted with pure mineral phases as well as soils. The  $K_d$  values are listed with ancillary parameters that included clay content, pH, CEC, surface area, solution calcium concentrations, and solution strontium concentrations.

**Table H.8.** Strontium  $K_d$  data set for pure mineral phases and soils.

Sr $K_d$ (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> and Comments
21	0.8	5.2	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, Ohnuki, 1994
19	0.8	5.6	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
22	0.8	6.2	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
26	0.8	6.45	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
24	0.8	6.6	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
30	0.8	8.4	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
43	0.8	9.2	0.9	1.4	0	*	NaClO <sub>4</sub>	Soil A	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
0		5.5				*		Quartz	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
290		5.5	3.3	26.4	0	*		Kaolinite	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
140		5.5	3.6	43.9	0	*		Halloysite	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
17		5.5	0.6	1.4	0	*		Chlorite	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
37		5.5	1.9	2.2	0	*		Sericite	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
8		5.5	0.5	0.7	0	*		Oligoclase	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>
6		5.5	0.5		0	*		Hornblend	1, * = 4.4x10 <sup>2</sup> Bq/ml 85-Sr in 2.4x10 <sup>-8</sup> M SrCl <sub>2</sub>

Sr $K_d$ (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> and Comments
16		5.5	0.7		0	*		Pyroxene	1, * = $4.4 \times 10^2$ Bq/ml 85-Sr in $2.4 \times 10^{-8}$ M SrCl <sub>2</sub>
110		5.5	8.5	19.3	0	*		MnO <sub>2</sub>	1, * = $4.4 \times 10^2$ Bq/ml 85-Sr in $2.4 \times 10^{-8}$ M SrCl <sub>2</sub>
7.7		5.8			24	113 $\mu$ Ci/l	Groundwater	AA 45/1	2 Jackson and Inch, 1989
9.9		6.1			25	105 $\mu$ Ci/l	Groundwater	AA45/3	2, $K_d = -.38Ca + 0.82$ . $r^2 = 0.19$
12.6		6.1			23	105 $\mu$ Ci/l	Groundwater	AA45/4	2, Ca not important to Sr $K_d$
13.7		5.8			22	123 $\mu$ Ci/l	Groundwater	AA45/5	2
10.1		6			24	99 $\mu$ Ci/l	Groundwater	AA45/7	2
15.8		5.8			21	143 $\mu$ Ci/l	Groundwater	AA38/1	2
13.8		5.8			27	113 $\mu$ Ci/l	Groundwater	AA38/2	2
11		5.9			21	114 $\mu$ Ci/l	Groundwater	AA38/3	2
14.2		5.6			21	124 $\mu$ Ci/l	Groundwater	AA38/4	2
6		5.8			24	115 $\mu$ Ci/l	Groundwater	AA38/5	2
7.5		5.9			21	117 $\mu$ Ci/l	Groundwater	AA38/6	2
6.9		5.9			17	108 $\mu$ Ci/l	Groundwater	AA38/8	2
8.3		6.1			24	68 $\mu$ Ci/l	Groundwater	AA27/1	2
8		6.2			21	71 $\mu$ Ci/l	Groundwater	AA27/2	2
6.7		6.2			28	72 $\mu$ Ci/l	Groundwater	AA27/3	2
6.8		6.2				84 $\mu$ Ci/l	Groundwater	AA27/4	2
4.9		6.2			18	84 $\mu$ Ci/l	Groundwater	AA27/5	2
5.1		6.2			19	87 $\mu$ Ci/l	Groundwater	AA27/6	2
8.5		6.2			17	88 $\mu$ Ci/l	Groundwater	AA27/7	2
8.8		6.2			18	90 $\mu$ Ci/l	Groundwater	AA27/8	2
5.6		6.3			20	77 $\mu$ Ci/l	Groundwater	AA34/1	2
5.3		6.4			16	79 $\mu$ Ci/l	Groundwater	AA34/2	2
7.2		6.4			18	65 $\mu$ Ci/l	Groundwater	AA34/3	2
5.1		6.3			18	72 $\mu$ Ci/l	Groundwater	AA34/4	2
6.5		6.4			17	75 $\mu$ Ci/l	Groundwater	AA34/5	2
6		6.2			14	79 $\mu$ Ci/l	Groundwater	AA34/6	2

Sr K <sub>d</sub> (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference' and Comments
6.5		6.2			15	107 μCi/l	Groundwater	AA34/7	2
7.6		6.2			17	107 μCi/l	Groundwater	AA34/8	2
21.4			0.47				Groundwater		3 Patterson and Spoel, 1981
25			0.83				Groundwater		3, CEC was approximated by adding exch. Ca,Mg,K
12.7			0.39				Groundwater		3, Groundwater =7.4 ppm Ca, 1.7 ppm Mg, 2.2 ppm Na, 5.6 ppm Cl, 18 ppm SO <sub>4</sub>
7.9			0.46				Groundwater		3
15.6			0.81				Groundwater		3
9.4			0.21				Groundwater		3
7.6			0.25				Groundwater		3
6.4			0.24				Groundwater		3
7.7			0.26				Groundwater		3
28.1			0.76				Groundwater		3
7.63			0.26				Groundwater		3
11.4			0.41				Groundwater		3
20.1			0.44				Groundwater		3
13			0.25				Groundwater		3
9.8			0.29				Groundwater		3
11			0.22				Groundwater		3
13			0.39				Groundwater		3
7.8			0.2				Groundwater		3
3.8			0.1				Groundwater		3
3			0.1				Groundwater		3
2.5			0.13				Groundwater		3
4	10	4	5.5		0	1x10 <sup>-8</sup> M	.01M NaCl	Puye soil-Na	4
15	10	5	5.5		0	1x10 <sup>-8</sup> M	.01M NaCl		4, Noncalcareous soils
21	10	6	5.5		0	1x10 <sup>-8</sup> M	.01M NaCl		4
24	10	7.4	5.5		0	1x10 <sup>-8</sup> M	.01M NaCl		4

Sr K <sub>d</sub> (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> and Comments
3	10	3.6	5.5		400	1x10 <sup>-8</sup> M	.01M CaCl <sub>2</sub>	Puye soil-Ca	4
4.5	10	5.2	5.5		400	1x10 <sup>-8</sup> M	.01M CaCl <sub>2</sub>		4
5.2	10	6.8	5.5		400	1x10 <sup>-8</sup> M	.01M CaCl <sub>2</sub>		4
5.7	10	7.9	5.5		400	1x10 <sup>-8</sup> M	.01M CaCl <sub>2</sub>		4
7.2		3			0	0.1 ppm	2,000 ppm Na	Hanford Soil	5
12.7		5			0	0.1 ppm	2,000 ppm Na	Hanford Soil	5
14.9		7			0	0.1 ppm	2,000 ppm Na	Hanford Soil	5
12.9		9			0	0.1 ppm	2,000 ppm Na	Hanford Soil	5
25.1		11			0	0.1 ppm	2,000 ppm Na	Hanford Soil	5
40.6				0.98				C-27	6
48.6				0.96				C-27	6
35				0.88				C-97	6
39.2				0.8				C-55	6
25.2				0.73				C-81	6
16.4				0.39				C-62	6
10.3				0.36				C-71	6
8.2				0.32				C-85	6
7.6				0.25				C-77	6
7.8				0.51				MK-4	6
11.2				0.38				TK3	6
10.5				0.34				RK2	6
3.7				0.34				NK2	6
3.5		5.2	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
4.6		5.6	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
5.8		5.8	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
6.1		5.9	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
8.3		6	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7

Sr K <sub>d</sub> (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> and Comments
17		7.4	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
21		7.6	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
27		7.8	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
47		8.4	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
81		9.1	2		0	1x10 <sup>-10</sup> M	NaOH/HCl	Hanford soil	7
140	70	2.4		70	0	1x10 <sup>-8</sup> M	Water	Bentonite	8
160	70	2.4		70		1x10 <sup>-8</sup> M	Groundwater	Bentonite	8
1500	70	9.3		70	0	1x10 <sup>-8</sup> M	Water	Bentonite	8
1100	70	9.3		70		1x10 <sup>-8</sup> M	Groundwater	Bentonite	8
1800	10	6.1		130	0	1x10 <sup>-8</sup> M	Water	Takadate Loam	8, hydrohalloysite=10%, 70% silt
950	10	8		130		1x10 <sup>-8</sup> M	Groundwater	Takadate Loam	8, hydrohalloysite=10%, 70% silt
550	10	6.5		60	0	1x10 <sup>-8</sup> M	Water	Hachinohe Loam	8, hydrohalloysite = 10%, 90% silt
260	10	8.2		60		1x10 <sup>-8</sup> M	Groundwater	Hachinohe Loam	8, hydrohalloysite = 10%, 90% silt
19.1	4	7.66	10.4		129	100 µCi/l	Hanford Groundwater	cgs-1	9
21.5	6	7.87	5.9		58.5	100 µCi/l	Hanford Groundwater	trench-8	9, Groundwater pH = 8.3
23.2	5	8.17	4.57		35.1	100 µCi/l	Hanford Groundwater	tbs-1	9
48.5	0	8.24	3			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-22	10, Los Alamos, New Mexico
10200	0	8.17	54			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-38	10, Yucca Mt tuff sediments
2500	0	8.13	21			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM48	10, Approximate initial pH, final pH are presented
3790	0	8.24	27			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-49	10, Final pH 8.1- 8.5
3820	0	8.24	27			3.8x10 <sup>-8</sup> M	Yucca Groundwater	YM-50	10, Sediments = 106-500 µm fractions
27000	0	8.4		31	10	3.8x10 <sup>-8</sup> M	Yucca Groundwater	JA-18	10

Sr $K_d$ (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> and Comments
4850	0	8.63		31	50	$3.8 \times 10^{-8} M$	Yucca Groundwater	JA-19	10
85	0	8.25		8	10	$3.8 \times 10^{-8} M$	Yucca Groundwater	JA-32	10
17.7	0	8.5		8	50	$3.8 \times 10^{-8} M$	Yucca Groundwater	JA-33	10
385	0	8.39		105	10	$3.8 \times 10^{-8} M$	Yucca Groundwater	JA-37	10
149	0	8.45		105	50	$3.8 \times 10^{-8} M$	Yucca Groundwater	JA-38	10
25000		12				10 nCi/ml		kaolinite	13
530		12				10 nCi/ml		chlorite	13
71,000		12				10 nCi/ml		FeOOH	13
1.6	0.5	6.2	0.05			$10 \times 10^{-6} M$	Groundwater	Sediments	14
2.6	3	6.2	0.3			$10 \times 10^{-6} M$	Groundwater	Sediments	14, Added Kaolinite to sand
3.4	5	6.2	0.5			$10 \times 10^{-6} M$	Groundwater	Sediments	14, CEC estimated based on kaolinite = 10 meq/100 g
4.6	8	6.2	0.8			$10 \times 10^{-6} M$	Groundwater	Sediments	14
6.7	13	6.2	1.3			$10 \times 10^{-6} M$	Groundwater	Sediments	14
17,000			97			$1 \times 10^{-10} M$		Ohya tuff	14, Akiba and Hashimoto, 1990
150			3.4			$1 \times 10^{-10} M$		Pyrophyllite	14, $\log K_d = \log \text{CEC} +$ constant: for trace [Sr]
780			2.4			$1 \times 10^{-10} M$		Sandstone	14, pH not held constant, ranged from 6 to 9.
95			1.9			$1 \times 10^{-10} M$		Shale	14, 1g solid:50ml sol'n, centrifuged, 32- 60mesh
440			1.9			$1 \times 10^{-10} M$		Augite Andesite	14, CEC of Cs and $K_d$ of Sr
39			1.2			$1 \times 10^{-10} M$		Plagiorhyolite	14
380			0.75			$1 \times 10^{-10} M$		Olivine Basalt	14
50			0.57			$1 \times 10^{-10} M$		Vitric Massive Tuff	14
82			0.54			$1 \times 10^{-10} M$		Inada granite	14

Sr K <sub>d</sub> (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> and Comments
22			0.35			1x10 <sup>-10</sup> M		Rokko Granite	14
1.3			0.033			1x10 <sup>-10</sup> M		Limestone	14
2,000			2			1x10 <sup>-10</sup> M		Muscovite	14
140			0.93			1x10 <sup>-10</sup> M		Chlorite	14
40			0.36			1x10 <sup>-10</sup> M		Hedenbergite	14
20			0.33			1x10 <sup>-10</sup> M		Hornblende	14
71			0.11			1x10 <sup>-10</sup> M		Grossular	14
150			0.07			1x10 <sup>-10</sup> M		Microcline	14
0.92			0.067			1x10 <sup>-10</sup> M		Forsterite	14
14			0.034			1x10 <sup>-10</sup> M		K-Feldspar	14
30			0.032			1x10 <sup>-10</sup> M		Albite	14
3			0.022			1x10 <sup>-10</sup> M		Epidote	14
23			0.0098			1x10 <sup>-10</sup> M		Quartz	14
400	42.4	7.2	34		0		Water	Ringhold Soil	11, Soil from Richland WA
135	26.9	8.3	13.6		0		Water	Bowdoin Soil	11, from Montana
600	33.5	6.5	26.3		0		Water	Hall Soil	11, from Nebraska
70	3.5	8.3	5.8		0		Water	Composite Soil	11, from Hanford Site
2.4		4					Groundwater	Eolian Sand	12
4.7		5						Eolian Sand	12, Belgian soils
6		7						Eolian Sand	12, Composition of Groundwater was not given
2.3		4						Mol White Sand	12, Compared static vs. dynamic K <sub>d</sub>
5.5		5						Mol White Sand	12
4.8		7						Mol White Sand	12

Sr K <sub>d</sub> (ml/g)	Clay Content (%)	pH	CEC (meq/ 100 g)	Surface Area (m <sup>2</sup> /g)	[Ca] (ppm)	[Sr]	Background Solution	Soil ID	Reference <sup>1</sup> and Comments
2.6		4						Mol Lignitic Sand	12
5.3		5						Mol Lignitic Sand	12
7.2		7						Mol Lignitic Sand	12

<sup>1</sup> References: 1 = Ohnuki, 1994; 2 = Jackson and Inch, 1989; 3 = Patterson and Spoel, 1981; 4 = Keren and O'Connor, 1983; 5 Nelson, 1959; 6 = Inch and Killey, 1987; 7 = Rhodes and Nelson, 1957; 8 = Konishi *et al.*, 1988; 9 = Serne *et al.*, 1993; 10 = Vine *et al.*, 1980; 11 = McHenry, 1958; 12 = Baetsle *et al.*, 1964; 13 = Ohnuki, 1991; 14 = Lieser and Steinkopff, 1989

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## **APPENDIX I**

### **Partition Coefficients For Thorium**

## Appendix I

### Partition Coefficients For Thorium

#### I.1.0 BACKGROUND

Two generalized, simplifying assumptions were established for the selection of thorium  $K_d$  values for the look-up table. These assumptions were based on the findings of the literature review conducted on the geochemical processes affecting thorium sorption. The assumptions are as follows:

- Thorium adsorption occurs at concentrations less than  $10^{-9}$  M. The extent of thorium adsorption can be estimated by soil pH.
- Thorium precipitates at concentrations greater than  $10^{-9}$  M. This concentration is based on the solubility of  $\text{Th}(\text{OH})_4$  at pH 5.5. Although (co)precipitation is usually quantified with the solubility construct, a very large  $K_d$  value will be used in the look-up table to approximate thorium behavior in systems with high thorium concentrations.

These assumptions appear to be reasonable for a wide range of environmental conditions. However, these simplifying assumptions are clearly compromised in systems containing high alkalinity (LaFlamme and Murray, 1987), carbonate (LaFlamme and Murray, 1987), or sulfate (Hunter *et al.*, 1988) concentrations, and low or high pH values (pH values less than 3 or greater than 8) (Hunter *et al.*, 1988; LaFlamme and Murray, 1987; Landa *et al.*, 1995). These assumptions will be discussed in more detail in the following sections.

Thorium  $K_d$  values and some important ancillary parameters that influence sorption were collected from the literature and tabulated. Data included in this table were from studies that reported  $K_d$  values (not percent adsorbed or Freundlich or Langmuir constants) and were conducted in systems consisting of:

- Low ionic strength ( $< 0.1$  M)
- pH values between 4 and 10.5
- Dissolved thorium concentrations less than  $10^{-9}$  M
- Low humic material concentrations ( $< 5$  mg/l)
- No organic chelates (such as EDTA)

These aqueous chemistry constraints were selected to limit the thorium  $K_d$  values evaluated to those that would be expected to exist in a far-field. The ancillary parameters included in these tables were clay content, calcite concentration, pH, and CEC. Attempts were also made to include the concentrations of organic matter and aluminum/iron oxides in the solid phase in the data set. However, these latter ancillary parameters were rarely included in the reports evaluated during the compilation of the data set. The data set included 17 thorium  $K_d$  values.

The descriptive statistics of the thorium  $K_d$  data set are presented in Table I.1. The lowest thorium  $K_d$  value was 100 ml/g for a measurement made on a pH 10 soil (Rancon, 1973). The largest thorium  $K_d$  value was 500,000 ml/g for a measurement made on a silt/quartz soil of schist origin (Rancon, 1973). The average thorium  $K_d$  value for the 17 observations was  $54,000 \pm 29,944$  ml/g.

**Table I.1.** Descriptive statistics of thorium  $K_d$  value data set presented in Section I.3.

	Thorium $K_d$ (ml/g)	Clay Content (wt.%)	pH	CEC (meq/100 g)	Calcite (wt.%)	Al/Fe- Oxides (wt.%)	Organic Matter (wt.%)
Mean	54,000	26.8	6.1	13.7	29	--	--
Standard Error	29,944	6.3	0.4	11.2	13.4	--	--
Median	5,000	30	6	2.9	25	--	--
Mode	100,000	40	6	2.9	0	--	--
Standard Deviation	123,465	14.1	1.5	29.8	30.1	--	--
Sample Variance	$1.5 \times 10^{10}$	199.2	2.1	886.2	905	--	--
Minimum	100	12	4	1.7	0	--	--
Maximum	500,000	40	10	81.2	60	--	--
No. Observations	17	5	17	7	5	0	0

## I.2.0 Approach and Regression Models

### I.2.1 Correlations with Thorium $K_d$ Values

A matrix of the correlation coefficients for thorium  $K_d$  values with soil parameters is presented in Table I.2. The correlation coefficients that are significant at or less than the 1 percent or 5 percent level of probability are identified. The parameter with the largest correlation coefficient with thorium  $K_d$  was pH ( $r = 0.58$ ,  $n = 16$ ,  $P \leq 0.01$ , where  $r$ ,  $n$ , and  $P$  represent correlation coefficient, number of observations, and level of probability, respectively). The pH range for this data set is 4 to 7.6. When  $K_d$  data for pH 10 is included in the regression analysis, the correlation coefficient decreases to 0.14 ( $n = 17$ ,  $P \leq 0.22$ ). The nonsignificant correlations with clay content, CEC, and calcite may in part be attributed to the small number of values in the data sets.

**Table I.2.** Correlation coefficients (r) of the thorium  $K_d$  value data set presented in Section I.3.

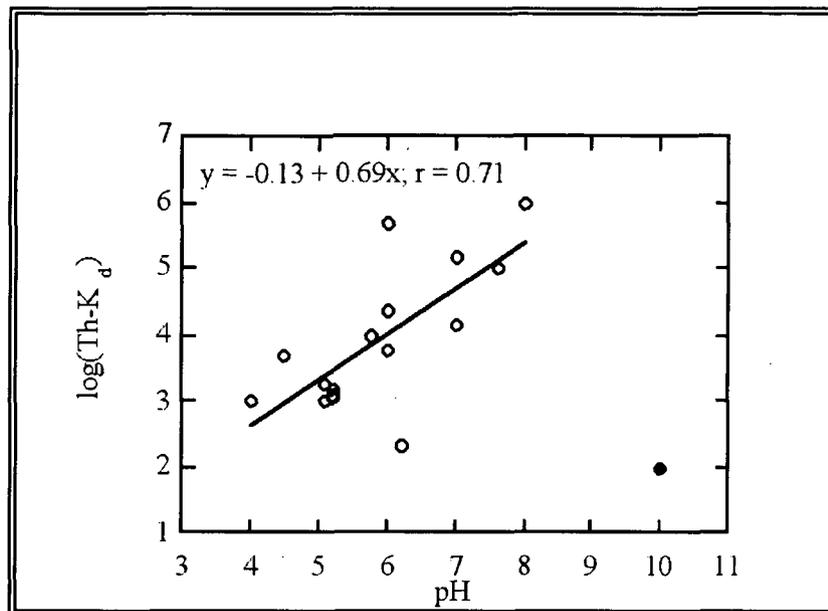
	Thorium $K_d$	Clay Content	pH	CEC
Thorium $K_d$	1			
Clay Content	-0.79	1		
pH	0.58 <sup>2</sup> (0.14) <sup>3</sup>	-0.84 <sup>1</sup>	1	
CEC	-0.15	--	-0.21	1
Calcite	0.76	-0.998 <sup>2</sup>	0.85 <sup>1</sup>	--

<sup>1,2</sup> Correlation coefficient is significant at the 5 percent ( $P \leq 0.05$ ) (indicated by footnote a) or 1 percent ( $P \leq 0.01$ ) (indicated by footnote b) level of significance, respectively. Significance level is in part dependent on the number of observations, n, (more specifically, the degrees of freedom) and variance of each correlation comparison (Table I.1). Thus, it is possible for thorium  $K_d$ /clay correlation coefficient of -0.79 to be not significant and the thorium  $K_d$ /pH correlation coefficient of 0.58 to be significant because the former has 4 degrees of freedom and the latter has 15 degrees of freedom.

<sup>3</sup> Excluding the  $K_d$  values at the highest pH value (pH 10), the correlation is 0.58 ( $n = 16$ ). Including this  $K_d$  value, the correlation coefficient decreases to 0.14.

### 1.2.2 Thorium $K_d$ Values as a Function of pH

Thorium  $K_d$  values were significantly correlated to pH between the pH range of 4 to 8, but were not correlated to pH between the range 4 to 10 (Figure I.1 and Table I.2). The pH dependence of thorium sorption to solid phases has been previously demonstrated with pure mineral phases (Hunter *et al.*, 1987; LaFlamme and Murray, 1987). The pH dependence can be explained in part by taking into consideration the aqueous speciation of thorium in groundwater. Thorium aqueous speciation changes greatly as a function of groundwater pH (Table I.3). As the pH increases, the thorium complexes become more anionic or neutral, thereby becoming less prone to be electrostatically attracted to a negatively charged solid phase. This decrease in electrostatic attraction would likely result in a decrease in  $K_d$  values. Figure I.1 shows an increase in thorium  $K_d$  values between pH 4 and 8. This may be the result of the pH increasing the number of exchange sites in the soil. At pH 10, the large number of neutral or anionic thorium complexes may have reduced the propensity of thorium to sorb to the soil.



**Figure I.1.** Linear regression between thorium  $K_d$  values and pH for the pH range from 4 to 8. [The single  $K_d$  value at pH 10 is identified by the filled circle.]

**Table I.3.** Calculated aqueous speciation of thorium as a function of pH. [The composition of the water and details of the aqueous speciation calculations are presented in Chapter 5. Total thorium concentration used in the aqueous speciation calculations is 1 ng/ml.]

pH	Dominant Aqueous Species	Percent (%) of Total Dissolved Thorium
3	$\text{ThF}_2^{2+}$	54
	$\text{ThF}_3^+$	42
7	$\text{Th}(\text{HPO}_4)_3^{2-}$	98
9	$\text{Th}(\text{OH})_4^0$ (aq)	99

The regression equation between the pH range of 4 to 8 that is shown in Figure I.1 is

$$\log (\text{Th } K_d) = -0.13 + 0.69(\text{pH}). \quad (\text{I.1})$$

The statistics for this equation are presented in Table I.4. The fact that the P-value for the intercept coefficient is  $\geq 0.05$  indicates that the intercept is not significantly ( $P \geq 0.05$ ) different than 0. The fact that the P-value for the slope coefficient is  $\leq 0.05$  indicates that the slope is significantly ( $P \geq 0.05$ ) different than 1. The lower and upper 95 percent coefficients presented in Table I.4 reflect the 95 percent confidence limits of the coefficients. They were used to calculate the upper and lower limits of expected thorium  $K_d$  values at a given pH value.

### ***1.2.3 Approach***

Linear regression analyses were conducted with data collected from the literature. These analyses were used as guidance for selecting appropriate  $K_d$  values for the look-up table. The  $K_d$  values used in the look-up tables could not be based entirely on statistical consideration because the statistical analysis results were occasionally nonsensible. For example, the data showed a negative correlation between clay content and thorium  $K_d$  values. This trend contradicts well established principles of surface chemistry. Instead, the statistical analysis was used to provide guidance as to the approximate range of values to use and to identify meaningful trends between the thorium  $K_d$  values and the solid phase parameters. Thus, the  $K_d$  values included in the look-up table were in part selected based on professional judgment. Again, only low-ionic strength solutions similar to that expected in far-field ground waters were considered in these analyses.

**Table I.4.** Regression coefficient and their statistics relating thorium  $K_d$  values and pH.  
[ $\log (\text{Th } K_d) = -0.13 + 0.69(\text{pH})$ , based on data presented in Figure I.1.]

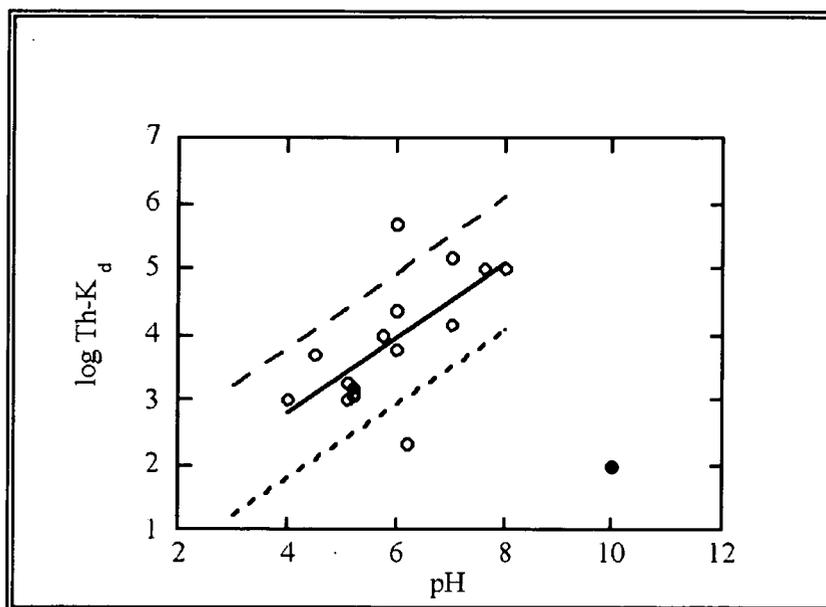
	<b>Coefficients</b>	<b>Standard Error</b>	<b>t-Statistic</b>	<b>P-value</b>	<b>Lower 95%</b>	<b>Upper 95%</b>
Intercept Coefficient	2.22	1.06	0.47	0.64	-1.77	2.76
Slope Coefficient	0.57	0.18	3.24	0.006	0.19	0.95

The look-up table (Table I.5) for thorium  $K_d$  values was based on thorium concentrations and pH. These 2 parameters have an interrelated effect on thorium  $K_d$  values. The maximum concentration of dissolved thorium may be controlled by the solubility of hydrous thorium oxides (Felmy *et al.*, 1991; Rai *et al.*, 1995; Ryan and Rai, 1987). The dissolution of hydrous thorium oxides may in turn vary with pH. Ryan and Rai (1987) reported that the solubility of hydrous thorium oxide is  $\sim 10^{-8.5}$  to  $\sim 10^{-9}$  in the pH range of 5 to 10. The concentration of dissolved thorium increases to  $\sim 10^{-2.6}$  M (600 mg/L) as pH decreases from 5 to 3.2. Thus, 2 categories, pH 3 - 5 and pH 5 - 10, based on thorium solubility were included in the look-up table. Although precipitation is typically quantified by the solubility construct, a very large  $K_d$  value was used in Table I.5 to describe high thorium concentrations.

The following steps were taken to assign values to each category in the look-up table. For  $K_d$  values in systems with pH values less than 8 and thorium concentrations less than the estimated solubility limits, Equation I.1 was used. This regression equation is for data collected between the pH range of 4 to 8 as shown in Figure I.1 [ $\log(\text{Th } K_d) = -0.13 + 0.69(\text{pH})$ ]. pH values of 4 and 6.5 were used to estimate the "pH 3 to 5" and "pH 5 to 8" categories, respectively. The  $K_d$  values in the "pH 8 to 10" category were based on the single laboratory experiment conducted at pH 10 that had a  $K_d$  of 200 ml/g. Upper and lower estimates of thorium  $K_d$  values were calculated by adding or subtracting 1 logarithmic unit to the "central estimates" calculated above for each pH category (Figure I.2). The 1 logarithm unit estimates for the upper and lower limits are based on visual examination of the data in Figure I.1. The use of the upper and lower regression coefficient values at the 95 percent confidence limits (Table I.5) resulted in calculated ranges that were unrealistically large. At pH 4, for the "pH 3 to 5" category, the lower and upper  $\log(\text{Th } K_d)$  values were calculated to be 1 and 6.6, respectively; at pH 6.5, this range of  $K_d$  was -0.5 to 9.0). All thorium  $K_d$  values for systems containing concentrations of dissolved thorium greater than their estimated solubility limit ( $10^{-9}$  M for pH 5 to 10 and  $10^{-2.6}$  M for pH < 5) were assigned a  $K_d$  of 300,000 ml/g.

**Table I.5.** Look-up table for thorium  $K_d$  values (ml/g) based on pH and dissolved thorium concentrations. [Tabulated values pertain to systems consisting of low ionic strength (<0.1 M), low humic material concentrations (<5 mg/l), no organic chelates (such as EDTA), and oxidizing conditions.]

$K_d$ (ml/g)	pH					
	3 - 5		5 - 8		8 - 10	
	Dissolved Th (M)		Dissolved Th (M)		Dissolved Th (M)	
	< $10^{-2.6}$	> $10^{-2.6}$	< $10^{-9}$	> $10^{-9}$	< $10^{-9}$	> $10^{-9}$
Minimum	62	300,000	1,700	300,000	20	300,000
Maximum	6,200	300,000	170,000	300,000	2,000	300,000



**Figure I.2.** Linear regression between thorium  $K_d$  values and pH for the pH Range 4 to 8. [Values  $\pm 1$  logarithmic unit from the regression line are also identified. The single  $K_d$  value at pH 10 is identified by the filled circle)].

### I.3.0 $K_d$ Data Set for Soils

The data set of thorium  $K_d$  values used to develop the look-up table are listed in Table I.6.

**Table I.6.** Data set containing thorium  $K_d$  values.

Thorium $K_d$ (ml/g)	pH	Clay (wt.%)	CEC <sup>1</sup> (meq/100g)	OM <sup>1</sup> (wt.%)	Fe-Oxides (wt.%)	Th (M)	Calcite (wt.%)	Solution Chemistry	Soil ID and Characteristics	Ref <sup>2</sup>
10,000	7.6		3					Synthetic GW <sup>1</sup> , pH 6.6	Soil A	1
500,000	6	40					0	Syn. GW, <sup>232</sup> Th Competing Ion	Silt+Qtz Sed., Schist soil	2
1,000	4	40					0	Syn. GW, <sup>232</sup> Th Competing Ion	Silt+Qtz Sed., Schist soil	2
100,000	8	12					60	Syn. GW, <sup>232</sup> Th Competing Ion	Silt+Qtz+OM+calcite, Schist Soil	2
150,000	7	30					25	Syn. GW, <sup>232</sup> Th Competing Ion	Cadarache Sed.	2
100	10	12					60	Syn. GW, <sup>232</sup> Th Competing Ion	Silt+Qtz+OM+calcite, Schist Soil	2
24,000	6							Groundwater	Glacial till, Clay	3
5,800	6							Groundwater	Fine Coarse Sand	3
1,028.6	5.1		2.9						Gleyed Dystric Brunisol, Ae Horizon 4-15 cm	4
1,271	5.2		2.1						Gleyed Dystric Brunisol, Bf Horizon 1 5-45 cm	4
5,000	4.5								Jefferson City, Wyoming, Fine Sandstone and Silty Clay	5
10,000	5.8								Jefferson City, Wyoming, Fine Sandstone and Silty Clay	5
15,000	7								Jefferson City, Wyoming, Fine Sandstone and Silty Clay	5
1,578	5.2		81.2					Groundwater	Gleyed Dystric Brunisol, Ah Horizon	6
1,862.5	5.1		2.9					Groundwater	Gleyed Dystric Brunisol, Ae Horizon	6
1,153.7	5.2		2.1					Groundwater	Gleyed Dystric Brunisol, Bf Horizon	6
206.9	6.2		1.7					Groundwater	Gleyed Dystric Brunisol, C Horizon	6

<sup>1</sup> CEC = cation exchange capacity, OC = organic matter, GW = groundwater.  
<sup>2</sup> References: 1 = Legoux *et al.*, 1992; 2 = Rancon, 1973; 3 = Bell and Bates, 1988; 4 = Sheppard *et al.*, 1987; 5 = Haji-Djafari *et al.*, 1981; 6 = Thibault *et al.*, 1990.

## I.5.0 References

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## **APPENDIX J**

### **Partition Coefficients For Uranium**

## Appendix J

### Partition Coefficients For Uranium

#### J.1.0 Background

The review of uranium  $K_d$  values obtained for a number of soils, crushed rock material, and single-mineral phases (Table J.5) indicated that pH and dissolved carbonate concentrations are the 2 most important factors influencing the adsorption behavior of U(VI). These factors and their effects on uranium adsorption on soils are discussed below. The solution pH was also used as the basis for generating a look-up table of the range of estimated minimum and maximum  $K_d$  values for uranium.

Several of the studies identified in this review demonstrate the importance dissolved carbonate through the formation of strong anionic carbonate complexes on the adsorption and solubility of dissolved U(VI). This complexation especially affects the adsorption behavior of U(VI) at alkaline pH conditions. Given the complexity of these reaction processes, it is recommended that the reader consider the application of geochemical reaction codes, and surface complexation models in particular, as the best approach to predicting the role of dissolved carbonate in the adsorption behavior of uranium and derivation of  $K_d$  values when site-specific  $K_d$  values are not available for U(VI).

#### J.2.0 Availability of $K_d$ Values for Uranium

More than 20 references were identified that reported the results of  $K_d$  measurements for the sorption of uranium onto soils, crushed rock material, and single mineral phases. These studies were typically conducted to support uranium migration investigations and safety assessments associated with the genesis of uranium ore deposits, remediation of uranium mill tailings, agriculture practices, and the near-surface and deep geologic disposal of low-level and high-level radioactive wastes (including spent nuclear fuel).

A large number of laboratory uranium adsorption/desorption and computer modeling studies have been conducted in the application of surface complexation models (see Chapter 5 and Volume I) to the adsorption of uranium to important mineral adsorbates in soils. These studies are also noted below.

Several published compilations of  $K_d$  values for uranium and other radionuclides and inorganic elements were also identified during the course of this review. These compilations are also briefly described below for the sake of completeness because the reported values may have applicability to sites of interest to the reader. Some of the  $K_d$  values in these compilations are tabulated below, when it was not practical to obtain the original sources references.

### J.2.1 Sources of Error and Variability

The  $K_d$  values compiled from these sources show a scatter of 3 to 4 orders of magnitude at any pH value from pH 4 to 9. As will be explained below, a significant amount of this variation represents real variability possible for the steady-state adsorption of uranium onto soils resulting from adsorption to important soil mineral phases (e.g., clays, iron oxides, clays, and quartz) as a function of important geochemical parameters (e.g., pH and dissolved carbonate concentrations). However, as with most compilations of  $K_d$  values, those in this report and published elsewhere, reported  $K_d$  values, and sorption information in general, incorporate diverse sources of errors resulting from different laboratory methods (batch versus column versus *in situ* measurements), soil and mineral types, length of equilibration (experiments conducted from periods of hours to weeks), and the fact that the  $K_d$  parameter is a ratio of 2 concentrations. These sources of error are discussed in detail in Volume I of this report.

Taking the ratio of 2 concentrations is particularly important to uranium, which, under certain geochemical conditions, will absorb to soil at less than 5 percent (very small  $K_d$ ) or up to more than 95 percent (very large  $K_d$ ) of its original dissolved concentration. The former circumstance (<5 percent adsorption) requires the investigator to distinguish very small differences in the analyzed initial and final concentrations of dissolved uranium. On the other hand, the latter circumstance (>95 percent adsorption) requires analysis of dissolved uranium concentrations that are near the analytical minimum detection limit. When comparing very small or very large  $K_d$  values published in different sources, the reader must remember this source of uncertainty can be the major cause for the variability.

In the following summaries, readers should note that the valence state of uranium is given as that listed in the authors' publications. Typically, the authors describe their procedures and results in terms of "uranium," and do not distinguish between the different valence states of uranium [U(VI) and U(IV)] present. In most studies, it is fair for the reader to assume that the authors are referring to U(VI) because no special precautions are described for conducting the adsorption studies using a dissolved reductant and/or controlled environmental chamber under ultralow oxygen concentrations. However, some measurements of uranium sorption onto crushed rock materials may have been compromised unbeknownst to the investigators by reduction of U(VI) initially present to U(IV) by reaction with ferrous iron [Fe(II)] exposed on fresh mineral surfaces. Because a major decrease of dissolved uranium typically results from this reduction due to precipitation of U(IV) hydrous-oxide solids (*i.e.*, lower solubility), the measured  $K_d$  values can be too large as a measure of U(VI) sorption. This scenario is possible when one considers the geochemical processes associated with some *in situ* remediation technologies currently under development. For example, Fruchter *et al.* (1996) [also see related paper by Amonette *et al.* (1994)] describe development of a permeable redox barrier remediation technology that introduces a reductant (sodium dithionite buffered at high pH) into contaminated sediment to reduce Fe(III) present in the sediment minerals to Fe(II). Laboratory experiments have shown that dissolved U(VI) will accumulate, via reduction of U(VI) to U(IV) and subsequent precipitation as a U(IV) solid, when it contacts such treated sediments.

### J.2.2 Uranium $K_d$ Studies on Soils and Rock Materials

The following sources of  $K_d$  values considered in developing the uranium  $K_d$  look-up table are listed in alphabetical order. Due to their extensive length, summary tables that list the uranium  $K_d$  values presented or calculated from data given in these sources are located at the end of this appendix.

Ames *et al.* (1982) studied the adsorption of uranium on 3 characterized basalts and associated secondary smectite clay. The experiments were conducted at 23 and 60°C under oxidizing conditions using 2 synthetic groundwater solutions. The compositions of the solutions were based on those of groundwater samples taken at depth from the Columbia River basalt formations. The basalts were crushed, and the 0.85-0.33 mm size fraction used for the adsorption studies. The groundwater solutions were mixed with the basaltic material and smectite in a ratio of 10 ml/1 g, and equilibrated for 60 days prior to analysis. Four initial concentrations of uranium ( $1.0 \times 10^{-4}$ ,  $1.0 \times 10^{-5}$ ,  $1.0 \times 10^{-6}$ , and  $1.0 \times 10^{-7}$  M uranium) were used for the measurements. The pH values in the final solutions ranged from 7.65 to 8.48. Uranium  $K_d$  values listed as "D" values in Ames *et al.* (1982, Table III) for the 23°C sorption measurements are listed in Table J.5.

Bell and Bates (1988) completed laboratory uranium (and other radionuclides)  $K_d$  measurements designed to evaluate the importance of test parameters such as pH, temperature, groundwater composition, and contact time at site-relevant conditions. Materials used for the  $K_d$  measurements included a sample of borehole groundwater that was mixed in a solution-to-solid ratio of 10 ml/1 g with the <5-mm size fraction of each of 5 soil materials. For the experiments conducted as a function of pH, the initial pH of the groundwater samples was adjusted by the addition of HCl, NaOH, or  $\text{NH}_4\text{OH}$ . The soils included a glacial till clay, sand, and 3 coarse granular deposits (listed as C1:2, C.3, and C.6 by Bell and Bates). The  $K_d$  values were measured using a batch method where the test vessel was agitated continuously at a fixed temperature for a pre-determined length of time. The uranium  $K_d$  values measured for the 5 soils at pH 5.7 and 15°C sampled at 14 days are listed in Table J.5. Bell and Bates noted that steady-state conditions were seldom achieved for 14 days contact at pH 5.7 and 15°C. For the clay and C1:2 soils, which exhibited the low-sorptive properties, the uranium  $K_d$  values doubled for each temperature increase of 5°. No significant temperature dependence was observed in the uranium  $K_d$  values measured using the other 3 soil materials. The uranium  $K_d$  values measured as a function of pH showed a maximum in sorption near pH 6 and 10, for the sand and clay soils. However, these 7-day experiments were affected by kinetic factors.

Erickson (1980) measured the  $K_d$  values for several radionuclides, including uranium, on abyssal red clay. The dominant mineral in the clay was iron-rich smectite, with lesser amounts of phillipsite, hydrous iron and manganese oxides. The  $K_d$  values were measured using a batch equilibration technique with equilibration times of 2-4 days and an initial concentration of dissolved uranium of approximately  $3.1 \times 10^{-8}$  mg/ml. The uranium  $K_d$  values measured at pH values of 2.8 and 7.1 by Erickson (1980) are listed in Table J.5.

Erikson *et al.* (1993) determined the  $K_d$  values for the adsorption of uranium on soil samples from the U.S. Department of Army munition performance testing sites at Aberdeen Proving Ground, Maryland, and Yuma Proving Ground, Arizona. The soil samples included 2 silt loams (Spesutie and Transonic) from the Aberdeen Proving Ground, and sandy loam (Yuma) from the Yuma Proving Ground. The names of the soil samples were based on the sampling locations at the study sites. The  $K_d$  measurements for the Spesutie and Transonic soil samples were conducted with site-specific surface water samples. Because no representative surface water existed at the Yuma site, the soil was equilibrated with tap water. The soil samples were equilibrated in a ratio of 30 ml/1 g with water samples spiked with 200  $\mu\text{g/l}$  uranium. The water/soil mixtures were sampled at 7 and 30 days. The  $K_d$  results are given in Table J.5. The  $K_d$  values reported for the 30-day samples are 4360 (pH 6.8), 328 (pH 5.6), and 54 ml/g (pH 8.0), respectively, for the Spesutie, Transonic, and Yuma soils. The lower  $K_d$  values measured for the Yuma Soil samples were attributed to carbonate complexation of the dissolved uranium.

Giblin (1980) determined the  $K_d$  values for uranium sorption on kaolinite as a function of pH in a synthetic groundwater. The measurements were conducted at 25°C using a synthetic groundwater (Ca-Na-Mg-Cl-SO<sub>4</sub>) containing 100  $\mu\text{g/l}$  uranium. Ten milliliters of solution was mixed with 0.01 g of kaolinite for a solution-to-solid ratio of 1,000 ml/1 g. The pH of the suspension was adjusted to cover a range from 3.8 to 10. Uranium  $K_d$  values from Giblin (1980, Figure 1) are given in Table J.5.<sup>1</sup> Giblin's results indicate that adsorption of uranium on kaolinite in this water composition was negligible below pH 5. From pH 5 to 7, the uranium  $K_d$  values increase to a maximum of approximately 37,000 ml/g. At pH values from 7 to 10, the uranium adsorption decreased.

Kaplan *et al.* (1998) investigated the effects of U(VI) concentration, pH, and ionic strength on the adsorption of U(VI) to a natural sediment containing carbonate minerals. The sediments used for the adsorption measurements were samples of a silty loam and a very coarse sand taken, respectively, from Trenches AE-3 and 94 at DOE's Hanford Site in Richland, Washington. Groundwater collected from an uncontaminated part of the Hanford Site was equilibrated with each sediment in a ratio of 2 ml/1 g for 14 or 30 days. The  $K_d$  values listed in Kaplan *et al.* (1998) are given in Table J.5. The adsorption of U(VI) was determined to be constant for concentrations between 3.3 and 100  $\mu\text{g/l}$  UO<sub>2</sub><sup>2+</sup> at pH 8.3 and an ionic strength of 0.02 M. This result indicates that a linear  $K_d$  model could be used to describe the adsorption of U(VI) at these conditions. In those experiments where the pH was greater than 10, precipitation of U(VI)-containing solids occurred, which resulted in apparent  $K_d$  values greater than 400 ml/g.

Kaplan *et al.* (1996) measured the  $K_d$  values for U(VI) and several other radionuclides at geochemical conditions being considered in a performance assessment for the long-term disposal of radioactive low-level waste in the unsaturated zone at DOE's Hanford Site in Richland,

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<sup>1</sup> The uranium  $K_d$  values listed in Table J.5 for Giblin (1980) were provided by E. A. Jenne (PNNL, retired) based on work completed for another research project. The  $K_d$  values were generated from digitization of the  $K_d$  values plotted in Giblin (1980, Figure 1).

Washington. The studies included an evaluation of the effects of pH, ionic strength, moisture content, and radionuclide concentration on radionuclide adsorption behavior. Methods used for the adsorption measurements included saturated batch adsorption experiments, unsaturated batch adsorption experiments, and unsaturated column adsorption experiments based on the Unsaturated Flow Apparatus (UFA). The measurements were conducted using uncontaminated pH 8.46 groundwater and the <2-mm size fraction of sediment samples collected from the Hanford Site. The sediment samples included TBS-1 Touchet Bed sand, Trench AE-3 silty loam, Trench-8 medium coarse sand, and Trench-94 very coarse sand. Dominant minerals identified in the clay-size fraction of these sediment samples included smectite, illite, vermiculite, and plagioclase. The reader should refer to Table 2.3 in Kaplan *et al.* (1996) for a listing of the physical and mineralogical properties of these sediment samples. Uranium  $K_d$  values estimated from results plotted in Kaplan *et al.* [1996, Figure 3.1 (400-day contact), Figure 3.2 (all values as function of dissolved uranium concentrations), and Figure 3.5 (100 percent saturation values) are listed in Table J.5. Their results show that U(VI)  $K_d$  values increased with increasing contact time with the sediments. For the concentration range from 3.3 to 100  $\mu\text{g/l}$  dissolved uranium, the U(VI)  $K_d$  values were constant. The U(VI)  $K_d$  values increased from 1.1 to 2.2 ml/g for pH values of 8 and 10, respectively, for these site-specific sediments and geochemical conditions. Kaplan *et al.* noted that, at pH values above approximately 10, the measured  $K_d$  values were affected by precipitation of uranium solids. Their measurements also indicated that U(VI)  $K_d$  values varied as a function of moisture content, although the trend differed based on sediment type. For a coarse-grained sediment, Kaplan *et al.* noted the  $K_d$  values increased with increasing moisture saturation. However, the opposite trend was observed for the U(VI)  $K_d$  values for fine-grained sediments. Kaplan *et al.* proposed that this behavior was related to changes in tortuosity and effective porosity within the fine pore spaces.

Kaplan and Serne (1995, Table 6.1) report  $K_d$  values for the adsorption of uranium on loamy sand sediment taken from Trench 8 at DOE's Hanford Site in Richland, Washington. The measurements were made using a column technique at unsaturated conditions (7 to 40 percent saturated), neutral-to-high pH, low organic material concentrations, and low ionic strength ( $I \leq 0.1$ ). The aqueous solutions consisted of a sample of uncontaminated groundwater from the Hanford Site. The  $K_d$  values listed in Kaplan and Serne (1995) are given in Table J.5. The  $K_d$  values ranged from 0.08 to 2.81 ml/g, and typically increase with increasing degree of column saturation. Kaplan and Serne noted that  $K_d$  values measured using a batch technique are usually greater than those obtained using the column technique due to the greater residence time and greater mixing of the sediment and aqueous phase associated with the batch method.

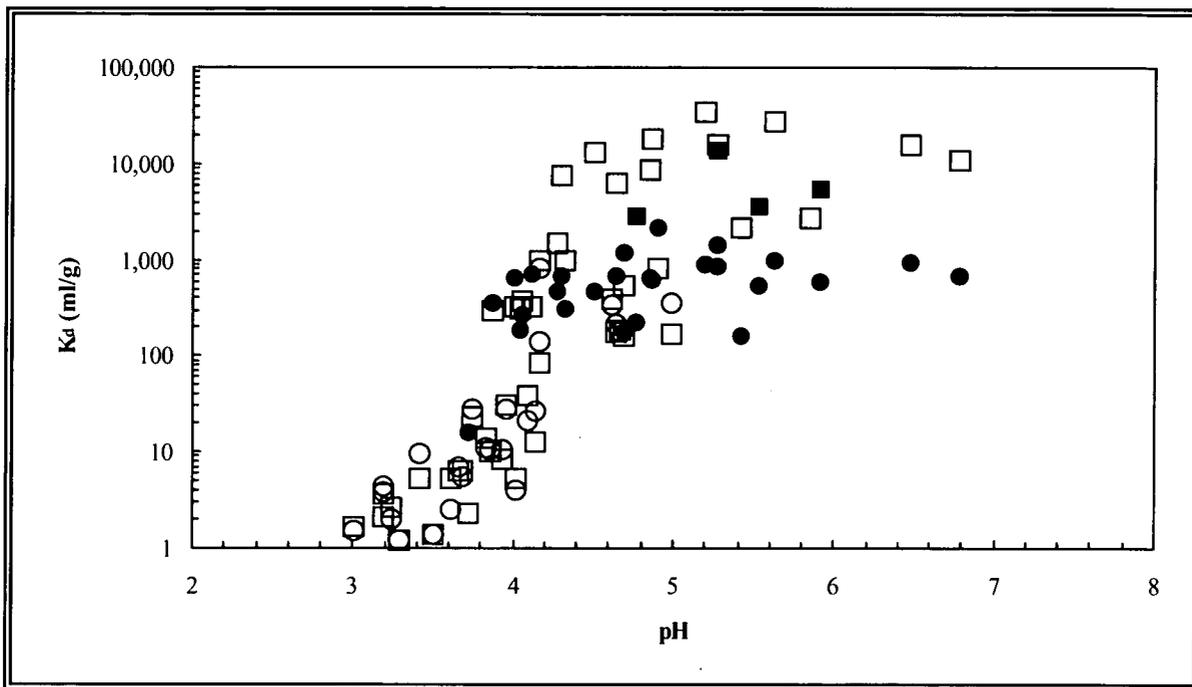
Lindenmeier *et al.* (1995) conducted a series of flow-through column tests to evaluate contaminant transport of several radionuclides through sediments under unsaturated (vadose zone) conditions. The sediments were from the Trench 8 (W-5 Burial Ground) from DOE's Hanford Site in Richland, Washington. The <2-mm size fraction of the sediment was used for the measurements. The <2-mm size fraction had a total cation exchange capacity (CEC) of 5.2 meq/100 g, and consisted of 87 percent sand, 7 percent silt, and 6 percent clay-size materials. Mineralogical analysis of <2-mm size fraction indicated that it consisted of 43.0 wt.% quartz,

26.1 wt.% plagioclase feldspar, and minor amounts of other silicate, clay, hydrous oxide, and carbonate minerals. The column tests were run using a site-specific groundwater and standard saturated column systems, commercial and modified Wierenga unsaturated column systems, and the Unsaturated Flow Apparatus (UFA). The results of the column tests indicated that the  $K_d$  values for uranium on this sediment material decrease as the sediment becomes less saturated. A  $K_d$  value of 2 ml/g was determined from a saturated column test conducted at a pore water velocity of 1.0 cm/h and residence time of 1.24 h. However, at 29 percent water saturation, the measured  $K_d$  value decreases by 70 percent to 0.6 ml/g (pore water velocity of 0.3 cm/h and residence time of 20.6 h). The  $K_d$  values listed in Lindenmeier *et al.* (1995, Table 4.1) are given in Table J.5.

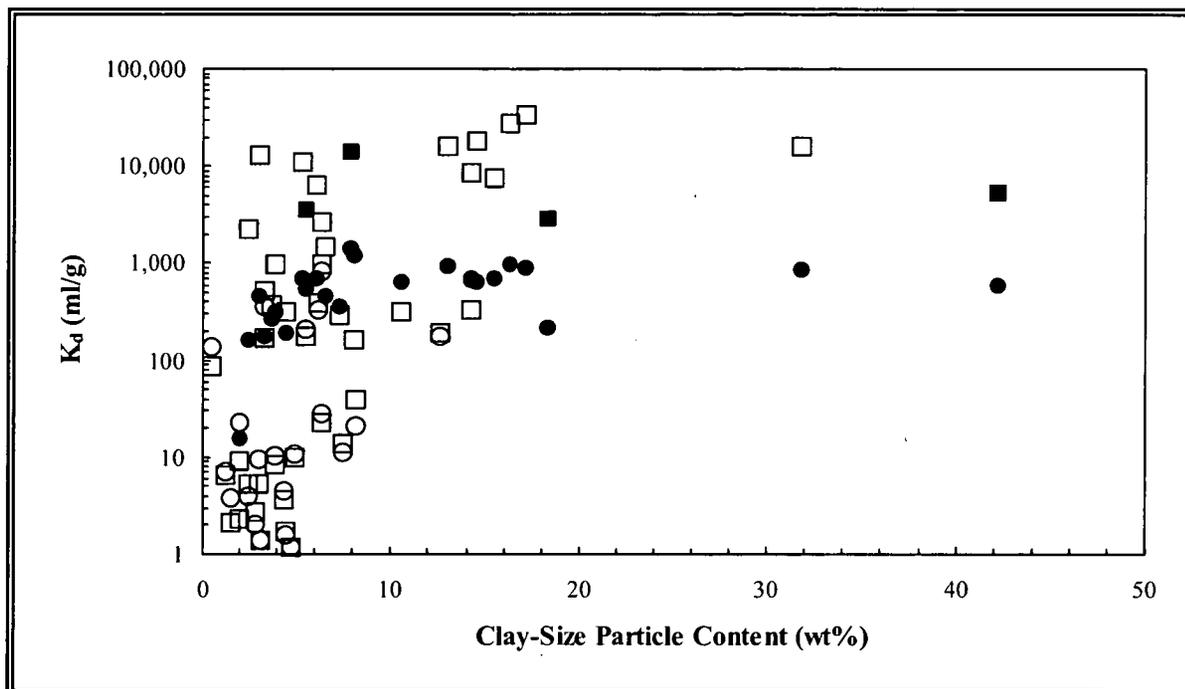
Salter *et al.* (1981) investigated the effects of temperature, pressure, groundwater composition, and redox conditions on the sorption behavior of several radionuclides, including uranium, on Columbia River basalts. Uranium  $K_d$  values were determined at 23 and 60°C under oxidizing and reducing conditions using a batch technique. The measurements were conducted with 2 synthetic groundwater solutions (GR-1 and GR-2) that have compositions representative of the groundwater present in basalt formations at DOE's Hanford Site, Richland, Washington. The GR-1 and GR-2 solutions represent a pH 8 sodium bicarbonate-buffered groundwater and a pH 10 silicic acid-buffered groundwater. The synthetic groundwater solutions were mixed with the crushed basalt material (0.03-0.85 mm size fraction) in a ratio of 10 ml/1 g. The contact time for the measurements was approximately 60 days. The  $K_d$  values were determined for initial concentrations of  $1.0 \times 10^{-4}$ ,  $1.0 \times 10^{-5}$ ,  $1.0 \times 10^{-6}$ ,  $1.0 \times 10^{-7}$ , and  $2.15 \times 10^{-8}$  M uranium. The  $K_d$  values listed in Table J.5 from Salter *et al.* (1981) include only those for 23°C under oxidizing conditions. The reader is referred to Salter *et al.* (1981) for a description of the measurement procedure and results for reducing conditions.

Serkiz and Johnson (1994) (and related report by Johnson *et al.*, 1994) investigated the partitioning of uranium on soil in contaminated groundwater downgradient of the F and H Area Seepage Basins at DOE's Savannah River Site in South Carolina. Their study included determination of an extensive set of field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  for 48 soil/porewater samples. The  $K_d$  values were determined from analyses of  $^{238}\text{U}$  and  $^{235}\text{U}$  in soil samples and associated porewaters taken from contaminated zones downgradient of the seepage basins. It should be noted that the mass concentration of  $^{235}\text{U}$  is significantly less than (*e.g.*, <1 percent) the concentration of  $^{238}\text{U}$  in the same soil sample and associated porewater. Serkiz and Johnson used the geochemical code MINTEQA2 to model the aqueous complexation and adsorption of uranium in their analysis of migration and partitioning in the contaminated soils. Soil/porewater samples were collected over a range of geochemical conditions (*e.g.*, pH, conductivity, and contaminant concentration). The field-derived uranium  $K_d$  listed for  $^{238}\text{U}$  and  $^{235}\text{U}$  by Serkiz and Johnson are given in Table J.5. The uranium  $K_d$  values varied from 1.2 to 34,000 ml/g over a pH range from approximately 3 to 6.7 (Figure J.1). The reader should note that the field-derived  $K_d$  values in Figures J.1, J.2, and J.3 are plotted on a logarithmic scale. At these site-specific conditions, the  $K_d$  values indicate that uranium adsorption increases with increasing pH over the pH range from 3 to 5.2. The adsorption of uranium is at a maximum at approximately pH 5.2, and then decreases with increasing pH over the pH range from 5.2 to 6.7.

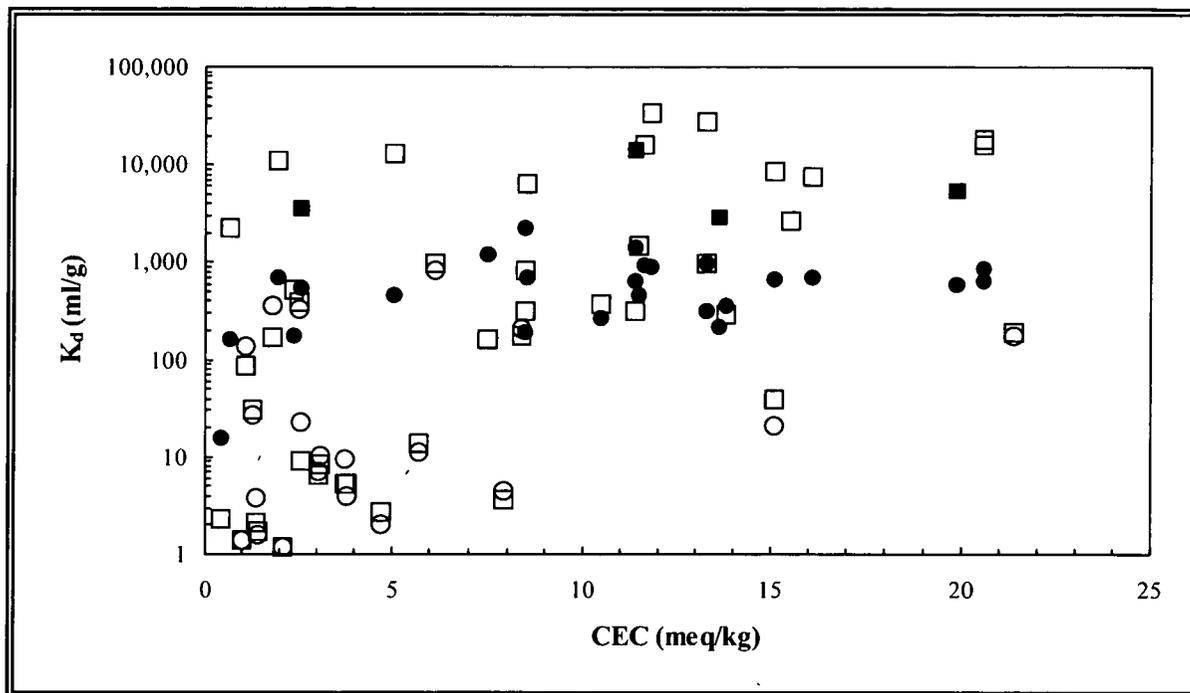
Serkiz and Johnson found that the field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  were not well correlated with the weight percent of clay-size particles (Figure J.2) or CEC (Figure J.3) of the soil samples. Based on the field-derived  $K_d$  values and geochemical modeling results, Serkiz and Johnson proposed that the uranium was not binding to the clays by a cation exchange reaction, but rather to a mineral surface coating with the variable surface charge varying due to the porewater pH.



**Figure J.1.** Field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  from Serkiz and Johnson (1994) plotted as a function of porewater pH for contaminated soil/porewater samples. [Square and circle symbols represent field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$ , respectively. Solid symbols represent minimum  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  that were based on minimum detection limit values for the concentrations for the respective uranium isotopes in porewaters associated with the soil sample.]



**Figure J.2.** Field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  from Serkiz and Johnson (1994) plotted as a function of the weight percent of clay-size particles in the contaminated soil/porewater samples. [Square and circle symbols represent field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$ , respectively. Solid symbols represent minimum  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  that were based on minimum detection limit values for the concentrations for the respective uranium isotopes in porewaters associated with the soil sample.]



**Figure J.3.** Field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  plotted from Serkiz and Johnson (1994) as a function of CEC (meq/kg) of the contaminated soil/porewater samples. [Square and circle symbols represent field-derived  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$ , respectively. Solid symbols represent minimum  $K_d$  values for  $^{238}\text{U}$  and  $^{235}\text{U}$  that were based on minimum detection limit values for the concentrations for the respective uranium isotopes in porewaters associated with the soil sample.]

Serne *et al.* (1993) determined  $K_d$  values for uranium and several other radionuclides at geochemical conditions associated with sediments at DOE's Hanford Site in Richland, Washington. The  $K_d$  values were measured using the batch technique with a well-characterized pH 8.3 groundwater and the <2-mm size fraction of 3 well-characterized sediment samples from the Hanford Site. The sediment samples included TBS-1 Touchet Bed sand, CSG-1 coarse sand/gravel, and Trench-8 medium coarse sand. The <2-mm size fraction of 3 samples consisted of approximately 70 to 90 wt.% plagioclase feldspar and quartz, and minor amounts of other silicate, clay, hydrous oxide, and carbonate minerals. The solution-to-solid ratio was fixed at 30 ml/1 g. The contact time for adsorption measurements with TBS-1, CSG-1, and Trench-8 were, 35, 35, and 44 days, respectively. The average  $K_d$  values tabulated for uranium in Serne *et al.* (1993) are given in Table J.5.

Sheppard and Thibault (1988) investigated the migration of several radionuclides, including uranium, through 3 peat<sup>1</sup> types associated with mires<sup>2</sup> typical of the Precambrian Shield in Canada. Cores of peat were taken from a floating sphagnum mire (samples designated PCE, peat-core experiment) and a reed-sedge mire overlying a clay deposit (samples designated SCE, sedge-core experiment). Uranium  $K_d$  values were determined by *in situ* and batch laboratory methods. The *in situ*  $K_d$  values were calculated from the ratio of uranium in the dried peat and associated porewater solutions. The batch laboratory measurements were conducted over an equilibration period of 21 days. The *in situ* and batch-measured uranium  $K_d$  values tabulated in Sheppard and Thibault (1988) are listed in Table J.5. Because the uranium  $K_d$  values reported by Sheppard and Thibault (1988) represent uranium partitioning under reducing conditions, which are beyond the scope of our review, these  $K_d$  values were not included in Figure J.4. Sheppard and Thibault (1988) noted that the uranium  $K_d$  for these 3 peat types varied from 2,00 to 19,000 ml/g, and did not vary as a function of porewater concentration. The laboratory measured  $K_d$  values were similar to those determined *in situ* for the SCE peat sample.

Thibault *et al.* (1990) present a compilation of soil  $K_d$  values prepared as support to radionuclide migration assessments for a Canadian geologic repository for spent nuclear fuel in Precambrian Shield plutonic rock. Thibault *et al.* collected  $K_d$  values from other compilations, journal articles, and government laboratory reports for important elements, such as uranium, that would be present in the nuclear fuel waste inventory. Some of the uranium  $K_d$  values listed by Thibault *et al.* were collected from references that were not available during the course of our review. These sources included studies described in reports by M. I. Sheppard, a coauthor of Thibault *et al.* (1990), and papers by Dahlman *et al.* (1976), Haji-Djafari *et al.* (1981), Neiheisel (1983), Rançon (1973) and Seeley and Kelmers (1984). The uranium  $K_d$  values, as listed in Thibault *et al.* (1990), taken for these sources are included in Table J.5.

Warnecke and coworkers (Warnecke *et al.*, 1984, 1986, 1988, 1994; Warnecke and Hild, 1988; and others) published several papers that summarize the results of radionuclide migration experiments and adsorption/desorption measurements ( $K_d$  values) that were conducted in support of Germany's investigation of the Gorleben salt dome, Asse II salt mine, and former Konrad iron ore mine as disposal sites for radioactive waste. Experimental techniques included batch and recirculation methods as well as flow-through and diffusion experiments. The experiments were designed to assess the effects of parameters, such as temperature, pH, Eh, radionuclide concentration, complexing agents, humic substances, and liquid volume-to-soil mass ratio, on radionuclide migration and adsorption/desorption. These papers are overviews of the work completed in their program to date, and provide very few details on the experimental designs and individual results. There are no pH values assigned to the  $K_d$  values listed in these overview

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<sup>1</sup> Peat is defined as "an unconsolidated deposit of semicarbonized plant remains in a water saturated environment" (Bates and Jackson, 1980).

<sup>2</sup> A mire is defined as "a small piece of marshy, swampy, or boggy ground" (Bates and Jackson, 1980).

papers. Warnecke *et al.* (1984) indicated that the measured pH values for the locations of soil and groundwater samples at Gorleben site studies range from 6 to 9.

Warnecke *et al.* (1994) summarize experiments conducted during the previous 10 years to characterize the potential for radionuclide migration at site-specific conditions at the Gorleben site. Characteristic, minimum, and maximum  $K_d$  values tabulated by Warnecke *et al.* (1994, Table 1) for uranium adsorbed to sandy and clayish sediments in contact with fresh or saline waters are listed below in Table J.1. No pH values were assigned to the listed  $K_d$  values. Warnecke *et al.* noted that the following progression in uranium  $K_d$  values as function of sediment type was indicated:

$$K_d (\text{Clay}) > K_d (\text{Marl}^1) > K_d (\text{Sandy}) .$$

Warnecke and Hild (1988) present an overview of the radionuclide migration experiments and adsorption/desorption measurements that were conducted for the site investigations of the Gorleben salt dome, Asse II salt mine, and Konrad iron ore mine. The uranium  $K_d$  values listed in Warnecke and Hild are identical to those presented in Warnecke *et al.* (1994). The uranium  $K_d$  values (ml/g) listed by Warnecke and Hild (1988, Table II) for sediments and different water types for the Konrad site are: 4 (Quaternary fresh water), 6 (Turonian fresh water), 6 (Cenomanian saline water), 20 [Albian (Hauterivain) saline water], 1.4 [Albian (Hils) saline water], 2.6 (Kimmeridgian saline water), 3 (Oxfordian saline water), and 3 [Bajocian (Dogger) saline water]. Warnecke and Hild (1988, Table III) list minimum and maximum uranium  $K_d$  values (0.54-15.2 ml/g) for 26 rock samples from the Asse II site. No pH values were assigned to any of the tabulated  $K_d$  values, and no descriptions were given regarding the mineralogy of the site sediment samples. Warnecke and Hild noted that sorption measurements for the Konrad sediments, especially for the consolidated material, show the same trend as those for the Gorleben sediments.

**Table J.1.** Uranium  $K_d$  values (ml/g) listed by Warnecke *et al.* (1994, Table 1).

Sediment Type	Fresh Water			Saline Water		
	Typical $K_d$ Value	Minimum $K_d$ Value	Maximum $K_d$ Value	Typical $K_d$ Value	Minimum $K_d$ Value	Maximum $K_d$ Value
Sandy	27	0.8	332	1	0.3	1.6
Clayish	17	8.6	100	14 - 1,400	14.1	1,400

<sup>1</sup> Marl is defined as “an earthy substance containing 35-65 percent clay and 65-35 percent carbonate formed under marine or freshwater conditions” (Bates and Jackson, 1980).

Warnecke *et al.* (1986) present an overview of the radionuclide migration experiments and adsorption/desorption measurements that were conducted for the Gorleben salt dome, and Konrad iron ore mine. The tabulated  $K_d$  values for the Gorleben and Konrad site sediments and waters duplicate those presented Warnecke *et al.* (1994) and Warnecke and Hild (1988).

Warnecke *et al.* (1984) present a short summary of radionuclide sorption measurements that were conducted by several laboratories in support of the Gorleben site investigation. Sediment (especially sand and silt) and water samples were taken from 20 locations that were considered representative of the potential migration path for radionuclides that might be released from a disposal facility sited at Gorleben. The minimum and maximum  $K_d$  values listed by Warnecke *et al.* (1984, Table III) are 0.5 and 3,000 ml/g, respectively (note that these values are not listed as a function of pH).

Zachara *et al.* (1992) studied the adsorption of U(VI) on clay-mineral separates from subsurface soils from 3 DOE sites. The materials included the clay separates (<2  $\mu\text{m}$  fraction) from the Kenoma Formation (Feed Materials Production Center, Fernald, Ohio), Ringold Formation (Hanford Site, Richland, Washington), and Cape Fear Formation (Savannah River Site, Aiken, South Carolina). Prior to the measurements the clay separates were treated with dithionite-citrate buffer and hydrogen peroxide to remove amorphous ferric hydroxides and organic materials. The measurements used clay suspensions ( $\approx 1$  meq of charge/l) spiked with 2 mg/l (8.6  $\mu\text{mol/l}$ ) uranium and  $\text{Ca}(\text{ClO}_4)_2$  or  $\text{NaClO}_4$  as the electrolyte. The pH values of the suspensions were adjusted over the pH range from 4.5 to 9.0 using sodium hydroxide. *The measurements were completed in a glovebox under an inert atmosphere to eliminate effects from aqueous complexation of U(VI) by dissolved carbonate.* Uranium  $K_d$  values calculated from values of percent uranium adsorbed versus pH (Zachara *et al.*, 1992, Figures 6 and 7) for the Kenoma and Ringold clays are listed in Table J.5.<sup>1</sup> The adsorption results for the Cape Fear clay isolate were essentially the same as those for the Kenoma clay (Zachara *et al.*, 1992, Figures 8). The results for the Kenoma clay isolate show a strong dependence of uranium adsorption as a function of ionic strength that is opposite to that expected for competitive sorption between uranium and the electrolyte cation. Zachara *et al.* (1992) suggest that this increase in uranium adsorption with increasing ionic strength may be due to the ionic strength dependence of the hydrolysis of the uranyl ion.

### J.2.3 Uranium $K_d$ Studies on Single Mineral Phases

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<sup>1</sup> The uranium  $K_d$  values listed in Table J.5 for Zachara *et al.* (1992) were provided by E. A. Jenne (PNNL, retired) based on work completed for another research project. The  $K_d$  values were derived from percent uranium adsorbed values generated from digitization of data plotted in Zachara *et al.* (1992, Figures 6 and 7) for the Kenoma and Ringold clay isolates. Due to the inherent uncertainty and resulting exceptionally large  $K_d$  values, Jenne did not calculate  $K_d$  values from any percent uranium adsorbed values that were greater than 99 percent.

Anderson *et al.* (1982) summarize an extensive study of radionuclides on igneous rocks and related single mineral phases. They report  $K_d$  values for U(VI) sorption on apatite, attapulgite (also known as palygorskite), biotite, montmorillonite, and quartz. The  $K_d$  values were determined using a batch technique using  $10^{-7}$ - $10^{-9}$  mol/l uranium concentrations, synthetic groundwater, and crushed (0.045-0.063 mm size fraction) mineral and rock material. The solution-to-solid ratio used in the experiments was 50 ml/1 g. The synthetic groundwater had a composition typical for a Swedish deep plutonic groundwater. Uranium  $K_d$  values from Anderson *et al.* (1982, Figure 6a) are given in Table J.5.<sup>1</sup>

Ames *et al.* (1983a,b) investigated the effects of uranium concentrations, temperature, and solution compositions on the sorption of uranium on several well-characterized secondary and sheet silicate minerals. The secondary phases studied by Ames *et al.* (1983a, oxide analyses listed in their Table 3) included clinoptilolite, glauconite, illite, kaolinite, montmorillonite, nontronite, opal, and silica gel. The sheet silicate minerals used by Ames *et al.* (1983b, oxide analyses listed in their Table 1) consisted of biotite, muscovite, and phlogopite. The sorption of uranium on each mineral phase was measured with 2 solutions (0.01 M NaCl and 0.01 M NaHCO<sub>3</sub>) using 4 initial uranium concentrations. The initial uranium concentrations used for the 25°C experiments included  $1.0 \times 10^{-4}$ ,  $1.0 \times 10^{-5}$ ,  $1.4 \times 10^{-6}$ , and  $4.4 \times 10^{-7}$  mol/l uranium. The batch experiments were conducted under oxidizing conditions at 5, 25, and 65°C in an environmental chamber. Solutions were equilibrated with the mineral solids in a ratio of 10 ml/1 g. A minimum of 30 days was required for the mineral/solution mixtures to reach steady state conditions. Uranium  $K_d$  values calculated from the 25°C sorption results given in Ames *et al.* (1983a, Table 6) are listed in Table J.5.

Ames *et al.* (1983c) studied the effects of uranium concentrations, temperature, and solution compositions on the sorption of uranium on amorphous ferric oxyhydroxide. The sorption of uranium on amorphous ferric oxyhydroxide was measured with 2 solutions (0.01 M NaCl and 0.01 M NaHCO<sub>3</sub>) using 4 initial uranium concentrations. The initial uranium concentrations used for the 25°C experiments included  $1.01 \times 10^{-4}$ ,  $1.05 \times 10^{-5}$ ,  $1.05 \times 10^{-6}$ , and  $4.89 \times 10^{-7}$  mol/l uranium for the 0.01 M NaCl solution, and  $1.01 \times 10^{-4}$ ,  $1.05 \times 10^{-5}$ ,  $1.53 \times 10^{-6}$ , and  $5.46 \times 10^{-7}$  mol/l uranium for the 0.01 M NaHCO<sub>3</sub> solution. The batch experiments were conducted under oxidizing conditions at 25 and 60°C. The solutions were equilibrated for 7 days with the amorphous ferric oxyhydroxide in a ratio 3.58 l/g of iron in the solid. Uranium  $K_d$  values calculated from the 25°C sorption results given in Ames *et al.* (1983c, Table II) are listed in Table J.5. Reflecting the high adsorptive capacity of ferric oxyhydroxide, the  $K_d$  values for the 25°C measurements range from approximately  $2 \times 10^6$  ml/g for the 0.01 M NaCl solution to approximately  $3 \times 10^4$  ml/g for the 0.01 M NaHCO<sub>3</sub> solution.

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<sup>1</sup> The uranium  $K_d$  values listed in Table J.5 for Anderson *et al.* (1982) were provided by E. A. Jenne (PNNL, retired) based on work completed for another research project. The  $K_d$  values were generated from digitization of the  $K_d$  values plotted in Anderson *et al.* (1982, Figure 6a).

Borovec (1981) investigated the adsorption of U(VI) and its hydrolytic complexes at 20°C and pH 6.0 on fine-grained kaolinite, illite, and montmorillonite. The results indicate that the  $K_d$  values increase with decreasing concentrations of dissolved uranium. At uranium concentrations less than  $10^{-4}$  mol/l, the uranium  $K_d$  values for the individual minerals were constant. The  $K_d$  values determined at 20°C and pH 6.0 ranged from 50 to 1,000. The values increased in the sequence  $K_d$  (kaolinite) <  $K_d$  (illite) <  $K_d$  (montmorillonite). Borovec presents the following linear equations for the maximum sorption capacity of uranium ( $a_m$ , in meq/100 g) on clays at 20°C and pH 6.0 with respect to CEC (in meq/100 g),

$$a_m = 0.90 \text{ CEC} + 1.56 \quad (r = 0.99522) ,$$

and specific surface ( $A$ , in  $\text{m}^2/\text{g}$ ) of clays,

$$a_m = 0.11 A + 2.05 \quad (r = 0.97232) .$$

#### ***J.2.4 Published Compilations Containing $K_d$ Values for Uranium***

Baes and Sharp (1983) present a model developed for annual-average, order-of-magnitude leaching constants for solutes in agricultural soils. As part of this model development, they reviewed and determined generic default values for input parameters, such as  $K_d$ , in their leaching model. A literature review was completed to evaluate appropriate distributions for  $K_d$  values for various solutes, including uranium. Because Baes and Sharp (1983) are cited frequently as a source of  $K_d$  values in other published  $K_d$  reviews (e.g., Looney *et al.*, 1987; Sheppard and Thibault, 1990), the uranium  $K_d$  values listed by Baes and Sharp are reported here for the sake of completeness. Based on the distribution that Baes and Sharp determined for the  $K_d$  values for cesium and strontium, they assumed a lognormal distribution for the  $K_d$  values for all other elements in their compilation. Baes and Sharp listed an estimated default  $K_d$  of 45 ml/g for uranium based on 24 uranium  $K_d$  values from 10.5 to 4,400 ml/g for agricultural soils and clays in the pH range from 4.5 to 9.0. Their compiled  $K_d$  values represent a diversity of soils, pure clays (other  $K_d$  values for pure minerals were excluded), extracting solutions, measurement techniques, and experimental error.

Looney *et al.* (1987) describe the estimation of geochemical parameters needed for environmental assessments of waste sites at DOE's Savannah River Plant in South Carolina. Looney *et al.* list  $K_d$  values for several metal and radionuclide contaminants based on values that they found in 1-5 published sources. For uranium, Looney *et al.* list a "recommended"  $K_d$  of 39.8 ( $10^{1.6}$ ) ml/g, and a range for its  $K_d$  values of 0.1 to 1,000,000 ml/g. Looney *et al.* note that their recommended values are specific to the Savannah River Plant site, and they must be carefully reviewed and evaluated prior to using them in assessments at other sites. Nonetheless, such data are often used as "default values" in radionuclide migration assessment calculations, and are therefore listed here for the sake of completeness. It should be noted that the work of Looney *et al.* (1987) predates the uranium-migration and field-derived uranium  $K_d$  study reported for contaminated soils at the Savannah River Site by Serkiz and Johnston (1994) (described above).

McKinley and Scholtis (1993) compare radionuclide  $K_d$  sorption databases used by different international organizations for performance assessments of repositories for radioactive wastes. The uranium  $K_d$  values listed in McKinley and Scholtis (1993, Tables 1, 2, and 4) are listed in Table J.2. The reader should refer to sources cited in McKinley and Scholtis (1993) for details regarding their source, derivation, and measurement. Radionuclide  $K_d$  values listed for cementitious environments in McKinley and Scholtis (1993, Table 3) are not included in Table J.2. The organizations listed in the tables in McKinley and Scholtis (1993) include: AECL (Atomic Energy of Canada Limited); GSF (Gesellschaft für Strahlen- und Umweltforschung m.b.H., Germany); IAEA (International Atomic Energy Agency, Austria); KBS (Swedish Nuclear Safety Board); NAGRA [Nationale Genossenschaft für die Lagerung radioaktiver Abfälle (Swiss National Cooperation for Storage of Radioactive Waste), Switzerland]; NIREX (United Kingdom Nirex Ltd.); NRC (U.S. Nuclear Regulatory Commission); NRPB (National Radiological Protection Board, United Kingdom); PAGIS [Performance Assessment of Geological Isolation Systems, Commission of the European Communities (CEC), Belgium; as well as PAGRIS SAFIR (Safety Assessment and Feasibility Interim Report)]; PSE (Projekt Sicherheitsstudien Entsorgung, Germany); RIVM [Rijksinstituut voor Volksgezondheid en Milieuhygiene (National Institute of Public Health and Environment Protection), Netherlands]; SKI [Statens Kärnkraftinspektion (Swedish Nuclear Power Inspectorate)]; TVO [Teollisuuden Voima Oy (Industrial Power Company), Finland]; and UK DoE (United Kingdom Department of the Environment).

**Table J.2.** Uranium  $K_d$  values listed by McKinley and Scholtis (1993, Tables 1, 2, and 4) from sorption databases used by different international organizations for performance assessments of repositories for radioactive wastes.

Organization	Argillaceous (Clay)		Crystalline Rock		Soil/Soil	
	Sorbing Material	$K_d$ (ml/g)	Sorbing Material	$K_d$ (ml/g)	Sorbing Material	$K_d$ (ml/g)
AECL	Bentonite-Sand	100	Granite	5	Soil/Sediment	20
GSF	Sediment	2				
IAEA	Pelagic Clay	500				
KBS-3	Bentonite	120	Granite	5,000		
NAGRA	Bentonite	1,000	Granite	1,000	Soil/Sediment	20
	Clay	5,000			Soil/Sediment	100
NIREX	Clay Mudstone	10				
NRC	Clay, Soil Shale	20	Granite	5		
			Basalt	4		
			Tuff	300		
NRPB	Clay	300			Soil/Sediment	300
PAGIS	Bentonite	90			Soil/Sediment	1,700
	Subseabed	100				
PAGIS SAFIR	Clay	600				
PSE	Sediment	0.02				
RIVM	Sandy Clay	10				
SKI	Bentonite	200	Granite	5,000		
TVO	Bentonite	90	Crystalline Rock, Reducing	200	Soil/Sediment	500
	Baltic Sea Sediment	500	Crystalline Rock, Real.	5		
	Ocean Sediment	500				
	Lake Sediment	500				
UK DoE	Clay	200			Soil/Sediment	50
	Coastal Marine Water	1000				

In a similar comparison of sorption databases for use in performance assessments of radioactive waste repositories, Stenhouse and Pöttinger (1994) list "realistic"  $K_d$  values (ml/g) for uranium

in crystalline rock/water systems of 1,000 (NAGRA), 5,000 [Svensk Kärnbränslehantering AB (Nuclear Fuel and Waste Management Company), Sweden; SKB], 1000 (TVO), and 6 (Canadian Nuclear Fuel Waste Management Programme, CNFWM). For bentonite/groundwater systems, they list 5,000 (NAGRA), 3,000 (SKB), and 500 (TVO). The reader should refer to sources cited in Stenhouse and Pöttinger for details regarding the source, derivation, and measurement of these values.

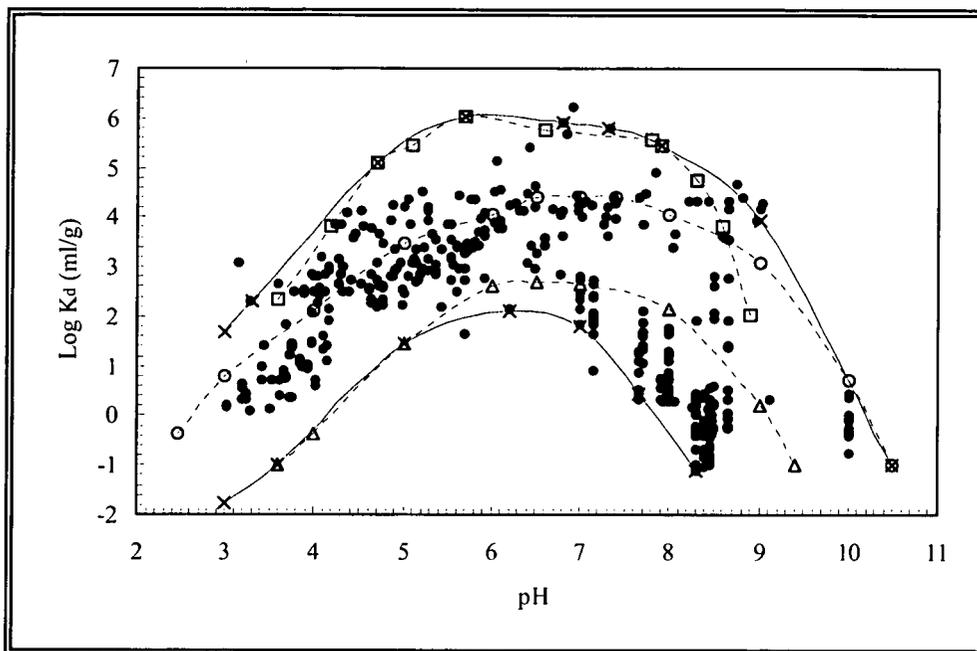
Thibault *et al.* (1990) [also summarized in Sheppard and Thibault (1990)] updated a compilation of soil  $K_d$  values that they published earlier (Sheppard *et al.*, 1984). The compilations were completed to support the assessment(s) of a Canadian geologic repository for spent nuclear fuel in Precambrian Shield plutonic rock. Thibault *et al.* collected  $K_d$  values from other compilations, journal articles, and government laboratory reports for important elements, such as uranium, that would be present in the inventory associated with Canada's nuclear fuel wastes. Because Thibault *et al.* (1990) and Sheppard and Thibault (1990) are frequently cited, their derived uranium  $K_d$  values are reported here for the sake of completeness. The  $K_d$  values for each element were categorized according to 4 soil texture types. These included sand (*i.e.*, contains  $\geq 70$  percent sand-size particles), clay (*i.e.*, contains  $\geq 35$  percent clay-size particles), loam (*i.e.*, contains an even distribution of sand-, clay-, and silt-size particles, or  $\leq 80$  percent silt-size particles), and organic (*i.e.*, contains  $>30$  percent organic matter and are either classic peat or muck sediments, or the litter horizon of a mineral sediment). Based on their previous evaluations, Thibault *et al.* ln-transformed and averaged the compiled  $K_d$  values to obtain a single geometric mean  $K_d$  value for each element for each soil type. The  $K_d$  values for each soil type and the associated range of  $K_d$  values listed for uranium by Thibault *et al.* (1990) are given in Table J.3.

**Table J.3.** Geometric mean uranium  $K_d$  values derived by Thibault *et al.* (1990) for sand, loam, clay, and organic soil types.

Soil Type	Geometric Mean $K_d$ Values (ml/g)	Observed Range of $K_d$ Values (ml/g)	Number of $K_d$ Values
Sand	35	0.03 - 2,200	24
Loam	15	0.2 - 4,500	8
Clay	1,600	46 - 395,100	7
Organic	410	33 - 7,350	6

### J.3.0 Approach in Developing $K_d$ Look-Up Table

The uranium  $K_d$  values listed in Table J.5 are plotted in Figure J.4 as a function of pH. The  $K_d$  values exhibit large scatter. This scatter increases from approximately 3 orders of magnitude at pH values below pH 5, to approximately 3 to 4 orders of magnitude from pH 5 to 7, and approximately 4 to 5 orders of magnitude at pH values from pH 7 to 9. This comparison can be somewhat misleading. At the lowest and highest pH regions, it should be noted that 1 to 2 orders of the observed variability actually represent uranium  $K_d$  values that are less than 10 ml/g. At pH values less than 3.5 and greater than 8, this variability includes extremely small  $K_d$  values of less than 1 ml/g.



**Figure J.4.** Uranium  $K_d$  values used for development of  $K_d$  look-up table. [Filled circles represent  $K_d$  values listed in Table J.5. Open symbols (joined by dotted line) represent  $K_d$  maximum and minimum values estimated from uranium adsorption measurements plotted by Waite *et al.* (1992) for ferrihydrite (open squares), kaolinite (open circles), and quartz (open triangles). The limits for the estimated maximum and minimum  $K_d$  values based on the values in Table J.5 and those estimated from Waite *et al.* (1992) are given by the "x" symbols joined by a solid line.]

### ***J.3.1 $K_d$ Values as a Function of pH***

Although the uranium  $K_d$  values in Figure J.4 exhibit a great deal of scatter at any fixed pH value, the  $K_d$  values show a trend as a function of pH. In general, the adsorption of uranium by soils and single-mineral phases is low at pH values less than 3, increases rapidly with increasing pH from pH 3 to 5, reaches a maximum in adsorption in the pH range from pH 5 to 8, and then decreases with increasing pH at pH values greater than 8. This trend is similar to the *in situ*  $K_d$  values reported by Serkiz and Johnson (1994) (see Figure J.1), and percent adsorption values measured for uranium on single mineral phases as described above and those reported for iron oxides (Duff and Amrhein, 1996; Hsi and Langmuir, 1985; Tripathi, 1984; Waite *et al.*, 1992, 1994; and others), clays (McKinley *et al.*, 1995; Turner *et al.*, 1996; Waite *et al.*, 1992; and others), and quartz (Waite *et al.*, 1992). The adsorption data are similar to those of other hydrolyzable metal ions with a sharp pH edge separating low adsorption at low pH from high adsorption at higher pH values. As discussed in the surface complexation laboratory and modeling studies [*e.g.*, Tripathi (1984), Hsi and Langmuir (1985), Waite *et al.* (1992, 1994), and Duff and Amrhein (1996)], this pH-dependent behavior is related to the pH-dependent surface charge properties of the soil minerals and complex aqueous speciation of dissolved U(VI), especially near and above neutral pH conditions where dissolved U(VI) forms strong anionic uranyl-carbonate complexes with dissolved carbonate.

### ***J.3.2 $K_d$ Values as a Function of Mineralogy***

In addition to the sources of error and variability discussed above, the scatter in  $K_d$  values in Figure J.4 is also related to heterogeneity in the mineralogy of the soils. Soils containing larger percentages of iron oxide minerals and mineral coatings and/or clay minerals will exhibit higher sorption characteristics than soils dominated by quartz and feldspar minerals. This variability in uranium adsorption with respect to mineralogy is readily apparent in uranium  $K_d$  values calculated from adsorption measurements (reported as percent uranium adsorbed versus pH) for ferrihydrite, kaolinite, and quartz by Waite *et al.* (1992).

Uranium  $K_d$  values were estimated<sup>1</sup> from the plots of percent uranium adsorption given for ferrihydrite, kaolinite, and quartz by Waite *et al.* (1992). To estimate the maximum variability that should be expected for the adsorption of uranium by different mineral substrates,  $K_d$  values were calculated from plots of uranium adsorption data for ferrihydrite and kaolinite (minerals with high adsorptive properties) that exhibited the maximum adsorption at any pH from 3 to 10, and for quartz (a mineral with low adsorptive properties) that exhibited the minimum adsorption

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<sup>1</sup> The reader is cautioned that significant uncertainty may be associated with  $K_d$  values estimated in this fashion because of the extreme solution-to-solid ratios used in some of these studies, especially for highly adsorptive iron-oxide phases, and errors related to estimating the concentrations of sorbed and dissolved uranium based on values for the percent of absorbed uranium near 0 or 100 percent, respectively.

at any pH. These estimated  $K_d$  values are shown, respectively, as open squares, circles, and triangles (and joined by dotted lines) in Figure J.4. The difference in the maximum and minimum  $K_d$  values is nearly 3 orders of magnitude at any fixed pH value in the pH range from 3 to 9.5. At pH values less than 7, the uranium  $K_d$  values for ferrihydrite and quartz calculated from data in Waite *et al.* (1992) bound more than 95 percent of the uranium  $K_d$  values gleaned from the literature. Above pH 7, the calculated uranium  $K_d$  values for ferrihydrite and kaolinite effectively bound the maximum uranium  $K_d$  values reported in the literature. In terms of bounding the minimum  $K_d$  values, the values calculated for quartz are greater than several data sets measured by Kaplan *et al.* (1996, 1998), Lindenmeier *et al.* (1995), and Serne *et al.* (1993) for sediments from the Hanford Site in Richland, Washington which typically contain a significant quantity of quartz and feldspar minerals. It should also be noted that some of the values listed from these studies represent measurements of uranium adsorption on Hanford sediments under partially saturated conditions.

### ***J.3.3 $K_d$ Values As A Function Of Dissolved Carbonate Concentrations***

As noted in several studies summarized above and in surface complexation studies of uranium adsorption by Tripathi (1984), Hsi and Langmuir (1985), Waite *et al.* (1992, 1994), McKinley *et al.* (1995), Duff and Amrhein (1996), Turner *et al.* (1996), and others, dissolved carbonate has a significant effect on the aqueous chemistry and solubility of dissolved U(VI) through the formation of strong anionic carbonate complexes. In turn, this complexation affects the adsorption behavior of U(VI) at alkaline pH conditions. Even differences in partial pressures of  $\text{CO}_2$  have a major effect on uranium adsorption at neutral pH conditions. Waite *et al.* (1992, Figure 5.7), for example, show that the percent of U(VI) adsorbed onto ferrihydrite decreases from approximately 97 to 38 percent when  $\text{CO}_2$  is increased from ambient (0.03 percent) to elevated (1 percent) partial pressures. In those adsorption studies that were conducted in the absence of dissolved carbonate (see surface complexation modeling studies listed above), uranium maintains a maximum adsorption with increasing pH as opposed to decreasing with increasing pH at pH values near and above neutral pH. Although carbonate-free systems are not relevant to natural soil/groundwater systems, they are important to understanding the reaction mechanisms affecting the aqueous and adsorption geochemistry of uranium.

It should be noted that it is fairly common to see figures in the literature or at conferences where uranium adsorption plotted from pH 2 to 8 shows maximum adsorption behavior even at the highest pH values. Such plots may mislead the reader into thinking that uranium adsorption continues this trend (*i.e.*, maximum) to even higher pH conditions that are associated with some groundwater systems and even porewaters derived from leaching of cementitious systems. Based on the uranium adsorption studies discussed above, the adsorption of uranium decreases rapidly, possibly to very low values, at pH values greater than 8 for waters in contact with  $\text{CO}_2$  or carbonate minerals.

No attempt was made to statistically fit the  $K_d$  values summarized in Table J.5 as a function of dissolved carbonate concentrations. Typically carbonate concentrations were not reported and/or discussed, and one would have to make assumptions about possible equilibrium between the

solutions and atmospheric or soil-related partial pressures of CO<sub>2</sub> or carbonate phases present in the soil samples. As will be discussed in a later section, the best approach to predicting the role of dissolved carbonate in the adsorption behavior of uranium and derivation of K<sub>d</sub> values is through the use of surface complexation modeling techniques.

### J.3.4 K<sub>d</sub> Values as a Function of Clay Content and CEC

No attempt was made to statistically fit the K<sub>d</sub> values summarized in Table J.5 as a function of CEC or concentrations of clay-size particles. The extent of clay concentration and CEC data, as noted from information included in Table J.5, is limited to a few studies that cover somewhat limited geochemical conditions. As discussed above, Serkiz and Johnson (1994) found no correlation between their uranium *in situ* K<sub>d</sub> values and the clay content (Figure J.2) or CEC (Figure J.3) of their soils. Their systems covered the pH conditions from 3 to 7.

As noted in the studies summarized above, clays have an important role in the adsorption of uranium in soils. Attempts have been made (*e.g.*, Borovec, 1981) to represent this functionality with a mathematical expression, but such studies are typically for limited geochemical conditions. Based on the studies by Chisholm-Brause (1994), Morris *et al.* (1994), McKinley *et al.* (1995), Turner *et al.* (1996), and others, uranium adsorption onto clay minerals is complicated and involves multiple binding sites, including exchange and edge-coordination sites. The reader is referred to these references for a detailed treatment of the uranium adsorption on smectite clays and application of surface complexation modeling techniques for such minerals.

### J.3.5 Uranium K<sub>d</sub> Look-Up Table

Given the orders of magnitude variability observed for reported uranium K<sub>d</sub> values, a subjective approach was used to estimate the minimum and maximum K<sub>d</sub> values for uranium as a function of pH. These values are listed in Table J.4. For K<sub>d</sub> values at non-integer pH values, especially given the rapid changes in uranium adsorption observed at pH values less than 5 and greater than 8, the reader should assume a linear relationship between each adjacent pair of pH-K<sub>d</sub> values listed in Table J.4.

**Table J.4.** Look-up table for estimated range of K<sub>d</sub> values for uranium based on pH.

K <sub>d</sub> (ml/g)	pH							
	3	4	5	6	7	8	9	10
Minimum	<1	0.4	25	100	63	0.4	<1	<1
Maximum	32	5,000	160,000	1,000,000	630,000	250,000	7,900	5

The minimum and maximum  $K_d$  values listed in Table J.4 were taken from the solid lines plotted in Figure F.4. The area between the 2 solid lines contains more than 95 percent of uranium  $K_d$  values collected in this review. The curve representing the minimum limit for uranium  $K_d$  values is based on  $K_d$  values calculated (described above) for quartz from data given in Waite *et al.* (1992) and the  $K_d$  values reported by Kaplan *et al.* (1996, 1998), Lindenmeirer *et al.* (1995), and Serne *et al.* (1993). It is unlikely that actual  $K_d$  values for U(VI) can be much lower than those represented by this lower curve. At the pH extremes along this curve, the uranium  $K_d$  values are already very small. Moreover, if one considers potential sources of error resulting from experimental methods, it is difficult to rationalize uranium  $K_d$  values much lower than this lower boundary.

The curve representing the maximum limit for uranium  $K_d$  values is based on  $K_d$  values calculated (described above) for ferrihydrite and kaolinite from data given in Waite *et al.* (1992). It is estimated that the maximum boundary of uranium  $K_d$  values plotted in Figure J.4 is conservatively high, possibly by an order of magnitude or more especially at pH values greater than 5. This estimate is partially based on the distribution of measured  $K_d$  values plotted in Figure J.4, and the assumption that some of the very large  $K_d$  measurements may have included precipitation of uranium-containing solids due to starting uranium solutions being oversaturated. Moreover, as noted previously, measurements of uranium adsorption onto crushed rock samples may include U(VI)/U(IV) redox/precipitation reactions resulting from contact of dissolved U(VI) with Fe(II) exposed on the fresh mineral surfaces.

#### **J.4.0 Use of Surface Complexation Models to Predict Uranium $K_d$ Values**

As discussed in Chapter 4 and in greater detail in Volume I of this report, electrostatic surface complexation models (SCMs) incorporated into chemical reaction codes, such as EPA's MINTEQA2, may be used to predict the adsorption behavior of some radionuclides and other metals and to derive  $K_d$  values as a function of key geochemical parameters, such as pH and carbonate concentrations. Typically, the application of surface complexation models is limited by the availability of surface complexation constants for the constituents of interest and competing ions that influence their adsorption behavior.

The current state of knowledge regarding surface complexation constants for uranium adsorption onto important soil minerals, such as iron oxides, and development of a mechanistic understanding of these reactions is probably as advanced as those for any other trace metal. In the absence of site-specific  $K_d$  values for the geochemical conditions of interest, the reader is encouraged to apply this technology to predict bounding uranium  $K_d$  values and their functionality with respect to important geochemical parameters.

Numerous laboratory surface complexation studies for uranium have been reported in the literature. These include studies of uranium adsorption onto iron oxides (Duff and Amrhein, 1996; Hsi and Langmuir, 1985; Tripathi, 1984; Waite *et al.*, 1992, 1994; and others), clays (McKinley *et al.*, 1995; Turner *et al.*, 1996; Waite *et al.*, 1992; and others), and quartz (Waite *et*

*al.*, 1992; and others). These references include derivation of the surface complexation constants for surface coordination sites determined to be important.

In addition to these laboratory studies, there are numerous examples in the literature of the application of surface complexation models and published binding constants to predict and evaluate the migration of uranium in soil/groundwater systems. For example, Koß (1988) describes the use of a surface complexation adsorption model to calculate the sorption of uranium for soil-groundwater systems associated with the proposed site for a German geologic radioactive waste repository at Gorleben. An apparent constant (*i.e.*, apparent surface complex formation constant based on bulk solution concentrations,  $K_d^{app}$ ) was derived for uranium sorption using the MINEQL geochemical code and site-specific geochemical data for soil CEC values, groundwater compositions, and measured uranium  $K_d$  values. Quartz ( $\text{SiO}_2$ ) was the main constituent in the soils considered in this study. Because the model incorporates the aqueous speciation of uranium, it may be used to compare  $K_d$  values for different soil systems having equal sorption sites. The modeling results indicated that CEC, pH, ionic strength, and dissolved carbonate concentrations were the main geochemical parameters affecting the sorption of uranium in groundwater systems.

Puigdomènech and Bergström (1994) evaluated the use of surface complexation models for calculating radionuclide sorption and  $K_d$  values in support of performance assessments studies of geologic repositories for radioactive wastes. They used a triple layer surface complexation model to predict the amount of uranium sorbed to a soil as a function of various environmental parameters. They then derived  $K_d$  values based on the concentrations of adsorbed and dissolved uranium predicted by the model. For the surface complexation modeling, they assumed (1) a total uranium concentration of  $10^{-5}$  mol/l, and (2) the adsorption of uranium on soil was controlled by the soil concentration of iron oxyhydroxide solid, which was assumed to be 5 percent goethite [ $\alpha\text{-FeO(OH)}$ ]. Their modeling results indicated that pH, inorganic carbon (*i.e.*, dissolved carbonate), and Eh (redox conditions) are major parameters that affect uranium  $K_d$  values. Under oxidizing conditions at pH values greater than 6, their derived  $K_d$  values were approximately 100 ml/g. At high concentrations of dissolved carbonate, and pH values greater than 6, the  $K_d$  values for uranium decrease considerably. Their results indicate that the triple layer surface complexation model using constants obtained under well controlled laboratory conditions on well characterized minerals can easily be applied to estimate the dependence of uranium adsorption and uranium  $K_d$  values as a function of a variety of important site environmental conditions.

Efforts have also been made to compile site binding constants for radionuclides and other metals to create "sorption databases" for use with geochemical codes such as MINTEQA2. For example, Turner *et al.* (1993) and Turner (1993, 1995) describe the application of the surface-complexation models (SCMs) [*i.e.*, the diffuse layer model (DLM), constant capacitance model (CCM), and triple layer model (TLM)] in the geochemical reaction code MINTEQA2 to simulate potentiometric titration and adsorption data published for U(VI) and other radionuclides on several single mineral phases. Their studies were conducted in support of developing a uniform approach to using surface complexation models to predict radionuclide migration

behavior associated with disposal of high-level radioactive waste in a geologic repository. The parameter optimization code FITEQL was used for fitting and optimization of the adsorption binding constants that were used in conjunction with MINTEQA2 and its thermodynamic database. For those radionuclides having sufficient data, the surface-complexation models were used to examine the effects of changing geochemical conditions (*e.g.*, pH) on radionuclide adsorption. Turner *et al.* (1993) and Turner (1993, 1995) include a detailed listing and documentation of the adsorption reactions and associated binding constants used for the MINTEQA2 DLM, CCM, and TLM calculations. Although all 3 models proved capable of simulating the available adsorption data, the DLM was able to do so using the fewest parameters (Turner, 1995). Compared to empirical approaches (*e.g.*,  $K_d$ ) for predicting contaminant adsorption, Turner notes that surface complexation models based on geochemical principles have the advantage of being used to extrapolate contaminant adsorption to environmental conditions beyond the range measured experimentally.

### J.5.0 Other Studies of Uranium

The following studies and adsorption reviews were identified during the course of this study. Although they typically do not contain uranium  $K_d$  data, they discuss aspects of uranium adsorption behavior in soils that might be useful to some readers searching for similar site conditions. These studies and reviews are briefly discussed below.

Ames and Rai (1978) reviewed and evaluated the processes influencing the mobility and retention of radionuclides. Their review for uranium discussed the following published adsorption studies. The following descriptions are paraphrased from in their report.<sup>1</sup>

- Dementyev and Syromyatnikov (1968) determined that the maximum adsorption observed for uranium in the pH 6 region is due to the boundary between the dominant uranium aqueous species being cationic and anionic at lower and higher pH values, respectively.
- Goldsztaub and Wey (1955) determined that 7.5 and 2.0 g uranium could be adsorbed per 100 g of calcined montmorillonite and kaolinite, respectively.
- Horr ath (1960) measured an average enrichment factor of 200 to 350 for the adsorption of uranium on peat.
- Kovalevskii (1967) determined that the uranium content of western Siberian noncultivated soils increased as a function of their clay content and that clay soils contained at least 3 times more uranium than sands.

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<sup>1</sup> The full citations listed for these references at the end of this appendix are provided exactly as given by Ames and Rai (1978).

- Manskaya *et al.* (1956) studied adsorption of uranium on fulvic acids as a function of pH. Results indicate a maximum removal of uranium of approximately 90 percent at pH 6, and 30 percent removal at pH values of 4 and 7.
- Masuda and Yamamoto (1971) showed that uranium from 1 to 100 mg/l uranium solutions was approximately completely adsorbed by volcanic ash, alluvial, and sandy soils.
- Rancon (1973) investigated the adsorption of uranium on several soils and single minerals. The  $K_d$  values reported by Rancon (1973) are (in ml/g): 39 for river sediment (quartz, clay, calcite, and organic matter); 33 for river peat; 16 for soil (quartz, clay, calcite, and no organic matter); 270 for quartz-clay soil developed from an altered schist; 0 for quartz; 7 for calcite; and 139 for illite.
- Ritchie *et al.* (1972) determined that the uranium content of a river sediment increased with decreasing particle size.
- Rozhkova *et al.* (1959) showed a maximum adsorption of uranium on lignite and humic acids between pH 5 and 6.
- Rubtsov (1972) found that approximately 58 percent of the total uranium was associated with the <1- $\mu$ m size fraction of forest podzolic mountain soils.
- Starik *et al.* (1958) studied adsorption of uranium on ferric hydroxide as a function of pH. Adsorption was a maximum at pH 5 with 50 percent uranium adsorption and decreased at pH values greater and less than pH 5.
- Szalay (1954, 1957) showed high adsorption of uranium by decomposing plant debris, peat, lignite, and brown coal.
- Yakobenchuck (1968) showed correlations of total uranium content with the silica, iron, and alumina oxide contents in sodpodzolic soils.
- Yamamoto *et al.* (1973) showed that uranium in 1 to 50 mg/l uranium solutions was almost completely adsorbed on 3 solids in carbonate waters.

Brindley and Bastovano (1982) studied the interaction of dissolved U(VI) with commercially available, synthetic zeolites of type A saturated with  $\text{Na}^+$ ,  $\text{K}^+$ , and  $\text{Ca}^{2+}$ . The experiments consisted of mixing 30 ml of uranyl acetate solution with 50 mg of finely powdered zeolite sample for an equilibration period of 4 days. The initial concentrations and pH values of the uranyl acetate solutions ranged from 100 to 3,700 ppm, and 3.5-3.8, respectively. The reaction of the zeolite with the uranyl acetate solution resulted in pH values in the range from 6 to 8 by exchange of  $\text{H}^+$  for exchangeable  $\text{Na}^+$ ,  $\text{K}^+$ , and  $\text{Ca}^{2+}$ . Examination of the reaction products using X-ray powder diffraction (XRD) indicated the formation of uranium-containing phases

accompanied by unreacted zeolite. The products of the reactions involving Na- and K-A zeolites contained a phase similar to compreignacite ( $K_2O \cdot 6UO_3 \cdot 11H_2O$ ). Those experiments conducted with Ca-A zeolite contained a phase similar to becquerelite ( $CaO \cdot 6UO_3 \cdot 11H_2O$ ).

Ho and coworkers studied the adsorption of U(VI) on a well-characterized, synthetic hematite ( $\alpha\text{-Fe}_2\text{O}_3$ ) sol.<sup>1</sup> Characterization data listed for the hematite sol by Ho and Doern (1985) and cited in other studies by Ho and coworkers included a particle size of 0.12  $\mu\text{m}$ , surface area of 34  $\text{m}^2/\text{g}$ , isoelectric point<sup>2</sup> of pH 7.6, and composition of >98 percent  $\alpha\text{-Fe}_2\text{O}_3$  and <2 percent  $\beta\text{-FeO(OH)}$ . Ho and Doern (1985) studied the adsorption of U(VI) on the hematite sol as a function of dissolved U(VI) concentration. Their procedure consisted of mixing 10 ml of the hematite sol (*i.e.*, constant particle concentration of 0.2 g/l) with 10 ml of uranyl nitrate solution. The uranyl solutions and hematite sol were previously prepared at the required concentration, pH, and ionic strength. The mixtures were equilibrated for 16 hr at 25°C. Over the pH range from 3 to 6.2, Ho and Doern determined that adsorption of U(VI) on the hematite sol increased with increasing concentrations of dissolved U(VI). Even though the particles of hematite sol had a net positive charge in the pH range from 3 to 6.2, significant adsorption of U(VI) was measured. The adsorption of U(VI) was greatest at pH of approximately 6.2, and decreased significantly at lower pH values. Ho and Miller (1986) investigated the adsorption of U(VI) from bicarbonate solutions as a function of initial U(VI) concentration over the pH range from 6.5 to 9.1 using the hematite sol described previously. Their experimental procedure was similar to that described by Ho and Doern, except that the measurements were completed using a  $1 \times 10^{-3}$  mol/l  $\text{NaHCO}_3$  solution in which its pH was adjusted by the addition of dilute HCl. Over the pH range from 6.5 to 9.1, Ho and Miller determined that the adsorption of uranium decreased abruptly with increasing pH. In experiments conducted with an initial U(VI) concentration of  $5 \times 10^{-6}$  mol/l, the reported percentages of U(VI) adsorbed on the hematite sol were approximately 98, 47, and 26 percent, respectively, at pH values of 7.1, 8.4, and 9.1. Ho and Miller (1985) evaluated the effect of dissolved humic acid on the adsorption of U(VI) by the hematite sol described in Ho and Doern (1985) over the pH range from approximately 4.3 to 6.4. As used by Ho and Miller, the term "humic acid" referred to the "fraction of humic substances soluble in water at  $\text{pH} \geq 4.30$ ." The results of Ho and Miller (1985) indicate that the adsorption of U(VI) by hematite is affected by the addition of humic acid and that the magnitude of this effect varies with pH and concentration of humic acid. At low humic acid concentration of 3 mg/l, the surface coverage of the hematite by the humic acid is low and the U(VI) adsorption by the hematite sol is similar to that observed for bare hematite particles. However, as the concentration of humic acid increases, the adsorption behavior of U(VI) changes. In the extreme case of a high humic acid concentration of 24 mg/l, the U(VI) adsorption is opposite that observed for bare hematite sol. At intermediate concentrations of humic acid, there is a change

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<sup>1</sup> A sol is defined as "a homogeneous suspension or dispersion of colloidal matter in a fluid" (Bates and Jackson, 1980).

<sup>2</sup> The isoelectric point (iep) is defined as "the pH where the particle is electrokinetically uncharged" (Stumm and Morgan, 1981).

from enhanced U(VI) adsorption at low pH to reduced adsorption at high pH for the pH range from 4.3 to 6.4.

Tsunashima *et al.* (1981) investigated the sorption of U(VI) by Wyoming montmorillonite. The experiments consisted of reacting, at room temperature, the <2- $\mu\text{m}$  size fraction of montmorillonite saturated with  $\text{Na}^+$ ,  $\text{K}^+$ ,  $\text{Mg}^{2+}$ ,  $\text{Ca}^{2+}$ , and  $\text{Ba}^{2+}$  with U(VI) nitrate solutions containing 1 to 300 ppm U(VI). The tests included systems with fixed volumes and variable uranyl concentrations [50 mg of clay dispersed in 200 ml of U(VI) nitrate solutions with 1-40 ppm U(VI)] and systems with variable volumes and fixed amounts of U(VI) [100 mg clay dispersed in 100 ml of solution]. The duration of the contact period for the clay-solution suspensions was 5 days. Based on the conditions of the constant volume/constant ionic strength experiments, the results indicated that adsorption of uranyl ions ( $\text{UO}_2^{2+}$ ) was strongly preferred over  $\text{Na}^+$  and  $\text{K}^+$  by the clay, and less strongly preferred versus  $\text{Mg}^{2+}$ ,  $\text{Ca}^{2+}$ , and  $\text{Ba}^{2+}$ .

Vochten *et al.* (1990) investigated the adsorption of U(VI) hydrolytic complexes on well-characterized samples of natural zeolites in relation to the double-layer potential of the minerals. The zeolite samples included chabazite ( $\text{CaAl}_2\text{Si}_4\text{O}_{12}\cdot 6\text{H}_2\text{O}$ ), heulandite [ $(\text{Ca},\text{Na}_2)\text{Al}_2\text{Si}_7\text{O}_{18}\cdot 6\text{H}_2\text{O}$ ], scolecite ( $\text{CaAl}_2\text{Si}_3\text{O}_{10}\cdot 3\text{H}_2\text{O}$ ), and stilbite [ $(\text{Ca},\text{Na}_2,\text{K}_2)\text{Al}_2\text{Si}_7\text{O}_{18}\cdot 7\text{H}_2\text{O}$ ]. The adsorption measurements were conducted at  $25^\circ\text{C}$  over a pH range from 4 to 7.5 using 0.1 g of powdered (35-75  $\mu\text{m}$ ) zeolite added to a 50 ml solution of  $2\times 10^{-5}$  mol/l U(VI). The suspension was shaken for 1 week in a nitrogen atmosphere to avoid the formation of U(VI) carbonate complexes. Given the relatively small dimension of the channels in the zeolite crystal structure and ionic diameter of the non-hydrated  $\text{UO}_2^{2+}$  ion (3.84 Å), Vochten concluded that the adsorption of U(VI) was on the external surfaces of the zeolites. The results indicate low adsorption of U(VI) to the 4 zeolites from pH 4 to 5. The amount of U(VI) adsorption increases rapidly from pH 5 to 7 with the maximum rate of increase being between pH 6 to 7.<sup>1</sup> The adsorption results indicate that chabazite and scolecite had higher sorptive capacities for U(VI) than heulandite and stilbite.

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<sup>1</sup> Based on experimental solubility [e.g., as Krupka *et al.* (1985) and others] and geochemical modeling studies, the authors of this document suspect that Vochten *et al.* (1990) may have exceeded the solubility of U(VI) above pH 5 and precipitated a U(VI) solid, such as schoepite ( $\text{UO}_3\cdot 2\text{H}_2\text{O}$ ), during the course of their adsorption measurements conducted in the absence of (or minimal) dissolved carbonate.

**Table J.5.** Uranium  $K_d$  values selected from literature for development of look-up table.

pH	U $K_d$ (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
8.3	1.98				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 40%)
8.3	0.49				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 40%)
8.3	2.81				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 38%)
8.3	0.62				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 22%)
8.3	0.45				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 30%)
8.3	0.54				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 23%)
8.3	0.62				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 25%)
8.3	0.40				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 17%)
8.3	0.10				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 7%)
8.3	0.08				Hanford Groundwater	Trench 8 Loamy Sand	Kaplan and Serne (1995, Part. Sat. Column, 7%)
8.3	2.0		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Saturated Column 1)
8.3	0.5		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Saturated Column 1)
8.3	2.7		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Saturated Column 1)
8.3	1.0		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Unsat. Column 1, 65%)
8.3	0.5		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Unsat. UFA 1, 70%)
8.3	0.2		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Unsat. UFA 2, 24%)
8.3	1.1		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Unsat. Column 1, 63%)
8.3	1.1		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Unsat. Column 2, 43%)
8.3	0.6		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Unsat. UFA 1A, 29%)
8.3	0.6		5.2		Hanford Groundwater	Trench 8 Loamy Sand	Lindenmeir <i>et al.</i> (1995, Unsat. UFA 1C, 29%)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
8.4	0.20		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1998, Batch)
8.4	0.15		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1998, Batch)
8.4	0.09		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1998, Batch)
8.4	0.15		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1998, Batch)
8.4	0.14		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1998, Batch)
7.92	1.99		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
8.05	1.92		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
7.99	1.91		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
7.99	2.10		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
7.98	2.25		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
7.97	2.44		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
8.48	1.07		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
8.26	1.46		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
8.44	1.37		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
9.12	2.12		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1998, Batch)
8.46	0.90		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996, 100% Unsaturated Batch)
8.46	1.70		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1996, 100% Unsaturated Batch)
8.46	1.00		6.0	6.3	Hanford Groundwater	TSB-1	Kaplan <i>et al.</i> (1996, 100% Unsaturated Batch)
8.46	1.10		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996, Batch)
8.46	3.50		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1996, Batch)
8.46	2.10		6.0	6.3	Hanford Groundwater	TSB-1	Kaplan <i>et al.</i> (1996, Batch)
8.46	0.24		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996)
8.46	0.64		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996)
8.46	0.51		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996)
8.46	0.46		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996)
8.46	0.35		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996)
8.46	0.53		6.4	14.8	Hanford Groundwater	Trench AE-3	Kaplan <i>et al.</i> (1996)
8.46	0.23		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1996)
8.46	0.15		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1996)
8.46	0.1		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1996)
8.46	0.16		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1996)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
8.46	0.12		5.3	6.3	Hanford Groundwater	Trench 94	Kaplan <i>et al.</i> (1996)
	2	8				Sand	Neiheisel [1983, as listed in Thibault <i>et al.</i> (1990)]
	1	7				Sand	Neiheisel [1983, as listed in Thibault <i>et al.</i> (1990)]
	3	15				Sand	Neiheisel [1983, as listed in Thibault <i>et al.</i> (1990)]
	750	36				Clayey Sand	Neiheisel [1983, as listed in Thibault <i>et al.</i> (1990)]
	770	21				Clayey Sand	Neiheisel [1983, as listed in Thibault <i>et al.</i> (1990)]
	550	19				Clayey Sand	Neiheisel [1983, as listed in Thibault <i>et al.</i> (1990)]
2.00	100					Fine Sandstone and Silty Sand	Haji-Djafari <i>et al.</i> [1981, as listed in Thibault <i>et al.</i> (1990)]
4.50	200					Fine Sandstone and Silty Sand	Haji-Djafari <i>et al.</i> [1981, as listed in Thibault <i>et al.</i> (1990)]
5.75	1,000					Fine Sandstone and Silty Sand	Haji-Djafari <i>et al.</i> [1981, as listed in Thibault <i>et al.</i> (1990)]
7.00	2,000					Fine Sandstone and Silty Sand	Haji-Djafari <i>et al.</i> [1981, as listed in Thibault <i>et al.</i> (1990)]
5.6	25,000					Red-Brown Clayey	Seeley and Kelmers [1984, as listed in Thibault <i>et al.</i> (1990)]
5.6	250					Red-Brown Clayey	Seeley and Kelmers [1984, as listed in Thibault <i>et al.</i> (1990)]
5.20	58.4						Thibault <i>et al.</i> (1990, values determined by coworkers)
5.10	294.9						Thibault <i>et al.</i> (1990, values determined by coworkers)
5.20	160						Thibault <i>et al.</i> (1990, values determined by coworkers)
6.20	45.4						Thibault <i>et al.</i> (1990, values determined by coworkers)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
7.00	450	36	28.0			Silty Loam Clay	Thibault <i>et al.</i> (1990, values determined by coworkers)
7.30	1.2	15	17.0			Loam	Thibault <i>et al.</i> (1990, values determined by coworkers)
4.90	0.03	2	5.8			Medium Sand	Thibault <i>et al.</i> (1990, values determined by coworkers)
5.50	2900	1	120.0			Organic	Thibault <i>et al.</i> (1990, values determined by coworkers)
7.40	1.9	10	9.1			Fine Sandy Loam	Thibault <i>et al.</i> (1990, values determined by coworkers)
7.40	2.4	11	8.7			Fine Sandy Loam	Thibault <i>et al.</i> (1990, values determined by coworkers)
6.60	590	10	10.8			Fine Sandy Loam	Thibault <i>et al.</i> (1990, values determined by coworkers)
6.50	4500	10	12.6			Fine Sandy Loam	Thibault <i>et al.</i> (1990, values determined by coworkers)
7.10	15	12	13.4			Fine Sandy Loam	Thibault <i>et al.</i> (1990, values determined by coworkers)
7.00	16					Sand	Rancon [1973, as listed in Thibault <i>et al.</i> (1990)]
7.00	33					Organic Peat	Rancon [1973, as listed in Thibault <i>et al.</i> (1990)]
6.50	4400					Clay Fraction	Dahlman <i>et al.</i> [1976, as listed in Thibault <i>et al.</i> (1990)]
2.80	200					Abyssal Red Clay	Erickson (1980)
7.10	790,000					Abyssal Red Clay	Erickson (1980)
8.3	1.70		2.6		Hanford Groundwater	CGS-1 sand (coarse gravel sand)	Serne <i>et al.</i> (1993, Batch)
8.3	2.30		5.2		Hanford Groundwater	Trench 8 Loamy Sand (medium/coarse sand)	Serne <i>et al.</i> (1993, Batch)
8.3	79.30		6.0		Hanford Groundwater	TBS-1 Loamy Sand (Touchet Bed sand)	Serne <i>et al.</i> (1993, Batch)
8.00	56.0				Hanford Groundwater, GR-1	Umtanum Basalt	Salter <i>et al.</i> (1981)
8.00	7.5				Hanford Groundwater, GR-1	Umtanum Basalt	Salter <i>et al.</i> (1981)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
8.00	13.2				Hanford Groundwater, GR-1	Umtanum Basalt	Salter <i>et al.</i> (1981)
8.00	17.8				Hanford Groundwater, GR-1	Umtanum Basalt	Salter <i>et al.</i> (1981)
8.00	20.2				Hanford Groundwater, GR-1	Umtanum Basalt	Salter <i>et al.</i> (1981)
8.00	13.0				Hanford Groundwater, GR-1	Flow E Basalt	Salter <i>et al.</i> (1981)
8.00	2.7				Hanford Groundwater, GR-1	Flow E Basalt	Salter <i>et al.</i> (1981)
8.00	2.2				Hanford Groundwater, GR-1	Flow E Basalt	Salter <i>et al.</i> (1981)
8.00	3.2				Hanford Groundwater, GR-1	Flow E Basalt	Salter <i>et al.</i> (1981)
8.00	2.9				Hanford Groundwater, GR-1	Flow E Basalt	Salter <i>et al.</i> (1981)
8.00	16.0				Hanford Groundwater,GR-1	Pomona Basalt	Salter <i>et al.</i> (1981)
8.00	2.2				Hanford Groundwater,GR-1	Pomona Basalt	Salter <i>et al.</i> (1981)
8.00	3.5				Hanford Groundwater,GR-1	Pomona Basalt	Salter <i>et al.</i> (1981)
8.00	5.2				Hanford Groundwater,GR-1	Pomona Basalt	Salter <i>et al.</i> (1981)
8.00	5.8				Hanford Groundwater,GR-1	Pomona Basalt	Salter <i>et al.</i> (1981)
10.00	2.8				Hanford Groundwater,GR-2	Umtanum Basalt	Salter <i>et al.</i> (1981)
10.00	2.3				Hanford Groundwater,GR-2	Umtanum Basalt	Salter <i>et al.</i> (1981)
10.00	2.8				Hanford Groundwater,GR-2	Umtanum Basalt	Salter <i>et al.</i> (1981)
10.00	2.8				Hanford Groundwater,GR-2	Umtanum Basalt	Salter <i>et al.</i> (1981)
10.00	2.5				Hanford Groundwater,GR-2	Umtanum Basalt	Salter <i>et al.</i> (1981)
10.00	1.0				Hanford Groundwater,GR-2	Flow E Basalt	Salter <i>et al.</i> (1981)
10.00	0.5				Hanford Groundwater,GR-2	Flow E Basalt	Salter <i>et al.</i> (1981)
10.00	0.4				Hanford Groundwater,GR-2	Flow E Basalt	Salter <i>et al.</i> (1981)
10.00	0.8				Hanford Groundwater,GR-2	Flow E Basalt	Salter <i>et al.</i> (1981)
10.00	0.2				Hanford Groundwater,GR-2	Flow E Basalt	Salter <i>et al.</i> (1981)
10.00	0.9				Hanford Groundwater,GR-2	Pomona Basalt	Salter <i>et al.</i> (1981)
10.00	0.6				Hanford Groundwater,GR-2	Pomona Basalt	Salter <i>et al.</i> (1981)
10.00	0.8				Hanford Groundwater,GR-2	Pomona Basalt	Salter <i>et al.</i> (1981)
10.00	0.5				Hanford Groundwater,GR-2	Pomona Basalt	Salter <i>et al.</i> (1981)
10.00	0.4				Hanford Groundwater,GR-2	Pomona Basalt	Salter <i>et al.</i> (1981)
7.66	7.5		1.83	17.7	Hanford Groundwater,GR-1	Umtanum Basalt	Ames <i>et al.</i> (1982)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
7.66	13		1.83	17.7	Hanford Groundwater,GR-1	Umtanum Basalt	Ames <i>et al.</i> (1982)
7.66	18		1.83	17.7	Hanford Groundwater,GR-1	Umtanum Basalt	Ames <i>et al.</i> (1982)
7.66	20		1.83	17.7	Hanford Groundwater,GR-1	Umtanum Basalt	Ames <i>et al.</i> (1982)
8.38	2.4		1.83	17.7	Hanford Groundwater,GR-2	Umtanum Basalt	Ames <i>et al.</i> (1982)
8.38	2.9		1.83	17.7	Hanford Groundwater,GR-2	Umtanum Basalt	Ames <i>et al.</i> (1982)
8.38	2.9		1.83	17.7	Hanford Groundwater,GR-2	Umtanum Basalt	Ames <i>et al.</i> (1982)
8.38	2.5		1.83	17.7	Hanford Groundwater,GR-2	Umtanum Basalt	Ames <i>et al.</i> (1982)
7.65	2.7		1.5	10.3	Hanford Groundwater,GR-1	Flow E Basalt	Ames <i>et al.</i> (1982)
7.65	2.2		1.5	10.3	Hanford Groundwater,GR-1	Flow E Basalt	Ames <i>et al.</i> (1982)
7.65	3.2		1.5	10.3	Hanford Groundwater,GR-1	Flow E Basalt	Ames <i>et al.</i> (1982)
7.65	2.9		1.5	10.3	Hanford Groundwater,GR-1	Flow E Basalt	Ames <i>et al.</i> (1982)
8.38	0.55		1.5	10.3	Hanford Groundwater,GR-2	Flow E Basalt	Ames <i>et al.</i> (1982)
8.38	0.38		1.5	10.3	Hanford Groundwater,GR-2	Flow E Basalt	Ames <i>et al.</i> (1982)
8.38	0.78		1.5	10.3	Hanford Groundwater,GR-2	Flow E Basalt	Ames <i>et al.</i> (1982)
8.38	0.19		1.5	10.3	Hanford Groundwater,GR-2	Flow E Basalt	Ames <i>et al.</i> (1982)
7.90	2.2		4.84	31.2	Hanford Groundwater,GR-1	Pomona Basalt	Ames <i>et al.</i> (1982)
7.90	3.5		4.84	31.2	Hanford Groundwater,GR-1	Pomona Basalt	Ames <i>et al.</i> (1982)
7.90	5.2		4.84	31.2	Hanford Groundwater,GR-1	Pomona Basalt	Ames <i>et al.</i> (1982)
7.90	5.8		4.84	31.2	Hanford Groundwater,GR-1	Pomona Basalt	Ames <i>et al.</i> (1982)
8.48	0.57		4.84	31.2	Hanford Groundwater,GR-2	Pomona Basalt	Ames <i>et al.</i> (1982)
8.48	0.83		4.84	31.2	Hanford Groundwater,GR-2	Pomona Basalt	Ames <i>et al.</i> (1982)
8.48	0.47		4.84	31.2	Hanford Groundwater,GR-2	Pomona Basalt	Ames <i>et al.</i> (1982)
8.48	0.42		4.84	31.2	Hanford Groundwater,GR-2	Pomona Basalt	Ames <i>et al.</i> (1982)
7.7	27		71.66	646	Hanford Groundwater,GR-1	Smectite, secondary	Ames <i>et al.</i> (1982)
7.7	39		4.84	31.2	Hanford Groundwater,GR-1	Smectite, secondary	Ames <i>et al.</i> (1982)
7.7	127		4.84	31.2	Hanford Groundwater,GR-1	Smectite, secondary	Ames <i>et al.</i> (1982)
7.7	76		4.84	31.2	Hanford Groundwater,GR-1	Smectite, secondary	Ames <i>et al.</i> (1982)
7.7	12		4.84	31.2	Hanford Groundwater,GR-2	Smectite, secondary	Ames <i>et al.</i> (1982)
7.7	42		4.84	31.2	Hanford Groundwater,GR-2	Smectite, secondary	Ames <i>et al.</i> (1982)
7.7	48		4.84	31.2	Hanford Groundwater,GR-2	Smectite, secondary	Ames <i>et al.</i> (1982)
7.7	22		4.84	31.2	Hanford Groundwater,GR-2	Smectite, secondary	Ames <i>et al.</i> (1982)
6.85	477,285				0.01 NaCl	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)
6.80	818,221				0.01 NaCl	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
6.90	1,739,877				0.01 NaCl	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)
6.90	1,690,522				0.01 NaCl	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)
8.60	4,313				0.01 NaHCO <sub>3</sub>	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)
8.65	14,098				0.01 NaHCO <sub>3</sub>	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)
8.65	21,362				0.01 NaHCO <sub>3</sub>	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)
8.80	26,269				0.01 NaHCO <sub>3</sub>	Amor Fe(III) Hydroxide	Ames <i>et al.</i> (1983c)
7.15	8.4		15.3	1.59	0.01 NaCl	Biotite	Ames <i>et al.</i> (1983b)
7.15	43.9		15.3	1.59	0.01 NaCl	Biotite	Ames <i>et al.</i> (1983b)
7.15	253.5		15.3	1.59	0.01 NaCl	Biotite	Ames <i>et al.</i> (1983b)
7.15	544.3		15.3	1.59	0.01 NaCl	Biotite	Ames <i>et al.</i> (1983b)
7.15	113.7		0.95	1.88	0.01 NaCl	Muscovite	Ames <i>et al.</i> (1983b)
7.15	251.0		0.95	1.88	0.01 NaCl	Muscovite	Ames <i>et al.</i> (1983b)
7.15	459.7		0.95	1.88	0.01 NaCl	Muscovite	Ames <i>et al.</i> (1983b)
7.15	68.2		0.95	1.88	0.01 NaCl	Muscovite	Ames <i>et al.</i> (1983b)
7.15	67.9		1.17	1.22	0.01 NaCl	Phlogopite	Ames <i>et al.</i> (1983b)
7.15	85.4		1.17	1.22	0.01 NaCl	Phlogopite	Ames <i>et al.</i> (1983b)
7.15	95.4		1.17	1.22	0.01 NaCl	Phlogopite	Ames <i>et al.</i> (1983b)
8.65	0.9		15.3	1.59	0.01 NaHCO <sub>3</sub>	Biotite	Ames <i>et al.</i> (1983b)
8.65	3.4		15.3	1.59	0.01 NaHCO <sub>3</sub>	Biotite	Ames <i>et al.</i> (1983b)
8.65	23.0		15.3	1.59	0.01 NaHCO <sub>3</sub>	Biotite	Ames <i>et al.</i> (1983b)
8.65	80.8		15.3	1.59	0.01 NaHCO <sub>3</sub>	Biotite	Ames <i>et al.</i> (1983b)
8.65	2.2		0.95	1.88	0.01 NaHCO <sub>3</sub>	Muscovite	Ames <i>et al.</i> (1983b)
8.65	26.9		0.95	1.88	0.01 NaHCO <sub>3</sub>	Muscovite	Ames <i>et al.</i> (1983b)
8.65	602.5		0.95	1.88	0.01 NaHCO <sub>3</sub>	Muscovite	Ames <i>et al.</i> (1983b)
8.65	3489.6		0.95	1.88	0.01 NaHCO <sub>3</sub>	Muscovite	Ames <i>et al.</i> (1983b)
8.65	0.6		1.17	1.22	0.01 NaHCO <sub>3</sub>	Phlogopite	Ames <i>et al.</i> (1983b)
8.65	1.1		1.17	1.22	0.01 NaHCO <sub>3</sub>	Phlogopite	Ames <i>et al.</i> (1983b)
8.65	0.6		1.17	1.22	0.01 NaHCO <sub>3</sub>	Phlogopite	Ames <i>et al.</i> (1983b)
7	544.5		25	116.1	0.01 NaCl	Illite, only lowest U conc	Ames <i>et al.</i> (1983a)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
8.5	90.5		25	116.1	0.01 NaHCO <sub>3</sub>	Illite, only lowest U conc	Ames <i>et al.</i> (1983a)
7	657.8		12.2	68.3	0.01 NaCl	Kaolinite, only lowest U conc	Ames <i>et al.</i> (1983a)
8.5	400.8		12.2	68.3	0.01 NaHCO <sub>3</sub>	Kaolinite, only lowest U conc	Ames <i>et al.</i> (1983a)
7	542.0		120	747	0.01 NaCl	Montmorillonite, only lowest U conc	Ames <i>et al.</i> (1983a)
8.5	1.8		120	747	0.01 NaHCO <sub>3</sub>	Montmorillonite, only lowest U conc	Ames <i>et al.</i> (1983a)
7	299.9		95	861	0.01 NaCl	Nontronite, only lowest U conc	Ames <i>et al.</i> (1983a)
8.5	4.1		95	861	0.01 NaHCO <sub>3</sub>	Nontronite, only lowest U conc	Ames <i>et al.</i> (1983a)
7	138.0		16.03	137.3	0.01 NaCl	Glauconite, only lowest U conc	Ames <i>et al.</i> (1983a)
8.5	114.2		16.03	137.3	0.01 NaHCO <sub>3</sub>	Glauconite, only lowest U conc	Ames <i>et al.</i> (1983a)
7	66.5		140.2	20	0.01 NaCl	Clinoptilolite, only lowest U conc	Ames <i>et al.</i> (1983a)
8.5	0.6		140.2	20	0.01 NaHCO <sub>3</sub>	Clinoptilolite, only lowest U conc	Ames <i>et al.</i> (1983a)
7	225.7		3.18	46.8	0.01 NaCl	Opal, only lowest U conc	Ames <i>et al.</i> (1983a)
8.5	1.7		3.18	46.8	0.01 NaHCO <sub>3</sub>	Opal, only lowest U conc	Ames <i>et al.</i> (1983a)
7	300.5		2.79	626.3	0.01 NaCl	Silica Gel., only lowest U conc	Ames <i>et al.</i> (1983a)
8.5	639.9		2.79	626.3	0.01 NaHCO <sub>3</sub>	Silica Gel., only lowest U conc	Ames <i>et al.</i> (1983a)
7.3	4200.0		4.36			Spesutie (silt loam)	Erikson <i>et al.</i> (1993)
6.2	136.0		1.29			Transonic (silt loam)	Erikson <i>et al.</i> (1993)
8.0	44		9.30			Yuma (sandy loam)	Erikson <i>et al.</i> (1993)
6.8	4360		4.36			Spesutie (silt loam)	Erikson <i>et al.</i> (1993)
5.6	328		1.29			Transonic (silt loam)	Erikson <i>et al.</i> (1993)
8.0	54		9.30			Yuma (sandy loam)	Erikson <i>et al.</i> (1993)
	39					River Sediment (Quartz, clay, calcite, organic matter)	Rancon (1973) as cited by Ames and Rai (1978)
	33					River Peat	Rancon (1973) as cited by Ames and Rai (1978)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
	16					River Sediment (Quartz, clay, calcite)	Rancon (1973) as cited by Ames and Rai (1978)
	270					Soil (Quartz and Clay, from Altered Schist)	Rancon (1973) as cited by Ames and Rai (1978)
	0					Quartz	Rancon (1973) as cited by Ames and Rai (1978)
	7					Calcite	Rancon (1973) as cited by Ames and Rai (1978)
	139					Illite	Rancon (1973) as cited by Ames and Rai (1978)
	27 (0.8-332)				Fresh Water	Gorleben Salt Dome, Sandy Sediment	Warnecke <i>et al.</i> (1984, 1986, 1994), Warnecke and Hild (1988)
	1 (0.3-1.6)				Fresh Water	Gorleben Salt Dome, Sandy Sediment	Warnecke <i>et al.</i> (1984, 1986, 1994), Warnecke and Hild (1988)
	17 (8.5-100)				Saline Water	Gorleben Salt Dome, Clayish Sediment	Warnecke <i>et al.</i> (1984, 1986, 1994), Warnecke and Hild (1988)
	14-1,400				Saline Water	Gorleben Salt Dome, Clayish Sediment	Warnecke <i>et al.</i> (1984, 1986, 1994), Warnecke and Hild (1988)
	4				Quaternary fresh water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
	6				Turonian fresh water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
	6				Cenomanian saline water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
	20				Albian (Hauterivain) saline water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
	1.4				Albian (Hils) saline water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
	2.6				Kimmeridgian saline water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
	3				Oxfordian saline water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
	3				Bajocian (Dogger) saline water	Former Konrad Iron Ore Mine	Warnecke <i>et al.</i> (1986), Warnecke and Hild (1988)
3.83	310				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
3.90	235				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
3.94	741				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
3.96	211				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.03	694				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.13	720				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.28	898				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.33	630				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.36	247				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.53	264				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.58	903				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.61	324				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.71	522				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.81	1,216				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.95	1,185				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
4.84	3,381				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.00	2,561				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.10	2,635				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.11	3,807				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.19	4,293				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.52	4,483				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.15	4,574				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.24	5,745				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.16	7,423				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
5.28	3,214				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.52	5,564				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.44	6,687				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.54	6,185				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.58	6,615				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.85	7,124				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.45	8,146				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.56	8,506				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.74	9,332				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.50	10,462				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.69	10,681				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.54	11,770				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.66	13,616				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.81	14,675				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.86	14,417				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.75	20,628				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.01	24,082				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.20	22,471				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
5.95	26,354				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.35	26,078				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.40	25,601				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.35	27,671				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
6.46	30,529				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.13	31,477				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.26	33,305				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.80	37,129				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.86	37,657				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
6.81	32,312				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
7.10	29,390				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
7.85	33,583				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
7.67	26,518				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
8.40	30,523				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
8.51	19,632				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
9.45	23,177				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
9.80	17,763				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
9.90	14,499				Synthetic Groundwater, function of pH	Kaolinite	Giblin (1980)
3.8	2				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
3.5	5				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
3.7	8				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
3.7	69				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
4.0	116				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
6.4	1,216				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
6.5	1,824				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
6.6	2,679				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
7.7	7,379				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
8.0	2,506				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
8.3	21,979				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
8.6	3,999				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
9.0	14,689				Synthetic Groundwater, function of pH	Quartz	Andersson <i>et al.</i> (1982)
3.4	27				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
4.4	326				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
4.4	522				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
4.7	418				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
5.1	1,489				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
5.2	2,512				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
6.4	2,812				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
7.3	7,228				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
7.3	16,634				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
7.4	9,840				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
8.1	4,732				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
9.0	8,337				Synthetic Groundwater, function of pH	Biotite	Andersson <i>et al.</i> (1982)
3.3	207				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
3.8	324				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
4.0	726				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
4.0	668				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
4.4	3,767				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
4.5	4,732				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
5.0	16,218				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
5.3	8,241				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
6.0	140,605				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
7.7	24,660				Synthetic Groundwater, function of pH	Apatite	Andersson <i>et al.</i> (1982)
3.6	460				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
4.1	1,514				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
4.2	7,194				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
4.5	6,471				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
4.7	4,753				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
5.1	23,335				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
5.9	12,531				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
6.4	266,686				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
7.3	645,654				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
7.8	82,224				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
8.7	46,132				Synthetic Groundwater, function of pH	Attapulgitite (Palygorskite)	Andersson <i>et al.</i> (1982)
3.2	1,175				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
4.4	12,503				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
6.6	3,917				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
7.0	10,139				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
7.0	28,054				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
7.3	10,715				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
8.2	21,528				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
8.4	20,370				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
9.0	18,621				Synthetic Groundwater, function of pH	Montmorillonite	Andersson <i>et al.</i> (1982)
5.1	7,391		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.0	1,177		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.1	2,180		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.4	3,680		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.3	4,437		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.5	7,265		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.5	7,108		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.8	23,603		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.8	22,948		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
4.7	176		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
4.8	176		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.0	283		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.0	297		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.4	708		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.7	1,961		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
5.6	2,367		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.9	4,283		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.9	4,936		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
6.0	7,936		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
6.1	8,586		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
6.2	17,631		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
6.3	19,553		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
6.4	30,963		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
6.5	43,756		45	99	Ca Electrolyte, CO <sub>2</sub> Free	Kenoma Clay, <2um fraction	Zachara <i>et al.</i> (1992, Fig 6)
5.1	508		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.2	554		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.2	676		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.4	874		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.4	1,136		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.6	1,136		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.7	2,143		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.8	2,363		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.9	9,829		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.9	11,966		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.0	33,266		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.1	37,596		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
4.8	377		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
4.8	399		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.1	620		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.0	637		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.5	1,476		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.5	1,603		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.8	3,091		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.1	6,047		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.1	5,823		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.3	13,713		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.4	13,341		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
4.9	918		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.1	1,168		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.1	1,251		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.6	2,719		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
5.7	2,928		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.7	14,848		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.8	13,036		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
7.0	13,827		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
7.0	18,042		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
7.0	19,150		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
7.1	21,771		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
7.1	18,097		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
7.4	26,008		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
7.4	19,488		59	112	Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
7.7	31,032				Ca Electrolyte, CO <sub>2</sub> Free	Ringold Clay Isolate, <2um Fraction	Zachara <i>et al.</i> (1992, Fig 7)
6.28	3,400				Reducing Conditions	PCE Surface Core, 0-8 cm	Sheppard and Thibault (1988, In Situ)
6.28	2,800				Reducing Conditions	PCE Surface Core, 9-16 cm	Sheppard and Thibault (1988, In Situ)
6.28	3,000				Reducing Conditions	PCE Surface Core, 17-24 cm	Sheppard and Thibault (1988, In Situ)
6.28	11,600				Reducing Conditions	PCE Surface Core, 25-32 cm	Sheppard and Thibault (1988, In Situ)
6.28	18,600				Reducing Conditions	PCE Surface Core, 33-40 cm	Sheppard and Thibault (1988, In Situ)
6.09	3,200				Reducing Conditions	PCE Deep Core, 9-16 cm	Sheppard and Thibault (1988, In Situ)
6.09	8,900				Reducing Conditions	PCE Deep Core, 17-24 cm	Sheppard and Thibault (1988, In Situ)
6.09	9,400				Reducing Conditions	PCE Deep Core, 25-32 cm	Sheppard and Thibault (1988, In Situ)
6.09	12,500				Reducing Conditions	PCE Deep Core, 33-40 cm	Sheppard and Thibault (1988, In Situ)
5.94	3,000				Reducing Conditions	SCE Surface Core, 0-5 cm	Sheppard and Thibault (1988, In Situ)
6.82	8,800				Reducing Conditions	SCE Surface Core, 6-20 cm	Sheppard and Thibault (1988, In Situ)
7.28	2,600				Reducing Conditions	SCE Surface Core, 21-25 cm	Sheppard and Thibault (1988, In Situ)
7.28	1,700				Reducing Conditions	SCE Surface Core, 26-30 cm	Sheppard and Thibault (1988, In Situ)
7.28	700				Reducing Conditions	SCE Surface Core, 31-40 cm	Sheppard and Thibault (1988, In Situ)
	1,300				Reducing Conditions	PCE Surface Core, 0-40 cm	Sheppard and Thibault (1988, Batch)
	2,100				Reducing Conditions	PCE Deep Core, 40-80 cm	Sheppard and Thibault (1988, Batch)
	2,000				Reducing Conditions	SCE Surface Core, 1-10 cm	Sheppard and Thibault (1988, Batch)
	2,900				Reducing Conditions	SCE Surface Core, 10-30 cm	Sheppard and Thibault (1988, Batch)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
	870				Reducing Conditions	SCE Surface Core, 30-40 cm	Sheppard and Thibault (1988, Batch)
5.7	46		2.3		Site Borehole Groundwater	Clay (Glacial Till, Less Than 5 mm)	Bell and Bates (1988)
5.7	46		3.0		Site Borehole Groundwater	C1:2 (Brown, Slightly Silty, Less Than 5 mm)	Bell and Bates (1988)
5.7	900		2.7		Site Borehole Groundwater	C3 (Dark Brown Coarse Granular Deposit, Less Than 5 mm)	Bell and Bates (1988)
5.7	2,200		2.9		Site Borehole Groundwater	C6 (Brown Coarse Granular Deposit, Less Than 5 mm)	Bell and Bates (1988)
5.7	560		0.8		Site Borehole Groundwater	Sand (Light Brown Coarse Granular Deposit, Less Than 5 mm)	Bell and Bates (1988)
4.16	85.0	0.5	1.11			A12	Serkiz and Johnson (1994)
4.99	170.0	3.3	1.82			A13	Serkiz and Johnson (1994)
3.42	5.3	3	3.74			A13R	Serkiz and Johnson (1994)
3.19	2.1	1.5	1.39			A22	Serkiz and Johnson (1994)
3.01	1.7	4.5	1.4			A23	Serkiz and Johnson (1994)
3.19	3.7	4.4	7.92			A31	Serkiz and Johnson (1994)
3.5	1.4	3.1	1			A32	Serkiz and Johnson (1994)
3.29	1.2	4.7	2.1			A42	Serkiz and Johnson (1994)
5.42	2,200.0	2.5	0.68			A52	Serkiz and Johnson (1994)
3.72	2.3	2	0.42			A53	Serkiz and Johnson (1994)
3.24	2.7	2.8	4.71			B13	Serkiz and Johnson (1994)
3.93	8.5	3.9	3.06			B14	Serkiz and Johnson (1994)
3.86	10.1	4.9				B23	Serkiz and Johnson (1994)
4.02	5.2	2.5	3.8			B23R	Serkiz and Johnson (1994)
3.83	14.0	7.5	5.69			B24	Serkiz and Johnson (1994)
4.62	390.0	6.2	2.5			B32	Serkiz and Johnson (1994)
4.64	180.0	5.5	8.42			B33	Serkiz and Johnson (1994)
4.67	190.0	12.6	21.4			B42	Serkiz and Johnson (1994)
3.66	6.4	1.2	3.02			B43	Serkiz and Johnson (1994)
4.09	39.0	8.2	15.1			B51	Serkiz and Johnson (1994)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
3.61	5.3					B52	Serkiz and Johnson (1994)
4.69	530.0	3.3	2.39			B52R	Serkiz and Johnson (1994)
3.68	6.4					C13	Serkiz and Johnson (1994)
3.75	23.0	6.4				C14	Serkiz and Johnson (1994)
3.96	30.0		1.28			C22	Serkiz and Johnson (1994)
4.17	980.0	6.4	6.12			C23	Serkiz and Johnson (1994)
5.53	3,600.0	5.5	2.54			C32	Serkiz and Johnson (1994)
4.64	6,300.0	6.1	8.54			C33	Serkiz and Johnson (1994)
5.27	14,000.0	7.9	11.4			C42	Serkiz and Johnson (1994)
4.51	13,000.0	3	5.04			C43	Serkiz and Johnson (1994)
6.78	11,000.0	5.3	1.96			D13	Serkiz and Johnson (1994)
4.14	13.0					D13RA	Serkiz and Johnson (1994)
	9.3	2	2.55			D13RB	Serkiz and Johnson (1994)
4	320.0	10.5	11.4			E13	Serkiz and Johnson (1994)
4.04	310.0	4.5	8.5			E14	Serkiz and Johnson (1994)
5.85	2,700.0	6.4	15.5			E23	Serkiz and Johnson (1994)
4.32	980.0	3.9	13.3			E23R	Serkiz and Johnson (1994)
3.87	290.0	7.3	13.8			E24	Serkiz and Johnson (1994)
4.27	1,500.0	6.5	11.5			E33	Serkiz and Johnson (1994)
4.05	380.0	3.7	10.5			E34	Serkiz and Johnson (1994)
5.27	16,000.0	31.8	20.6			E41	Serkiz and Johnson (1994)
4.87	18,000.0	14.5	20.6			E42	Serkiz and Johnson (1994)
4.3	7,500.0	15.5	16.1			F12	Serkiz and Johnson (1994)
4.9	830.0		8.51			F13	Serkiz and Johnson (1994)
4.69	160.0	8.1	7.48			F22	Serkiz and Johnson (1994)
6.48	16,000.0	13	11.6			F23	Serkiz and Johnson (1994)
4.85	8,700.0	14.2	15.1			F32	Serkiz and Johnson (1994)
4.77	2,900.0	18.3	13.6			F33	Serkiz and Johnson (1994)
5.2	34,000.0	17.2	11.8			F42	Serkiz and Johnson (1994)
4.12	330.0	14.2				F43	Serkiz and Johnson (1994)
5.91	5,500.0	42.2	19.9			F52	Serkiz and Johnson (1994)
5.63	27,000.0	16.3	13.3			F53	Serkiz and Johnson (1994)
4.16	139.0	0.5	1.11			A12	Serkiz and Johnson (1994)
4.99	361.0	3.3	1.82			A13	Serkiz and Johnson (1994)
3.42	9.46	3	3.74			A13R	Serkiz and Johnson (1994)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
3.19	3.79	1.5	1.39			A22	Serkiz and Johnson (1994)
3.01	1.55	4.5	1.4			A23	Serkiz and Johnson (1994)
3.19	4.43	4.4	7.92			A31	Serkiz and Johnson (1994)
3.5	1.38	3.1	1			A32	Serkiz and Johnson (1994)
3.29	1.19	4.7	2.1			A42	Serkiz and Johnson (1994)
5.42	160.0	2.5	0.68			A52	Serkiz and Johnson (1994)
3.72	16.0	2	0.42			A53	Serkiz and Johnson (1994)
3.24	2.0	2.8	4.71			B13	Serkiz and Johnson (1994)
3.93	10.4	3.9	3.06			B14	Serkiz and Johnson (1994)
3.86	10.7	4.9				B23	Serkiz and Johnson (1994)
4.02	4.0	2.5	3.8			B23R	Serkiz and Johnson (1994)
3.83	11.3	7.5	5.69			B24	Serkiz and Johnson (1994)
4.62	332.0	6.2	2.5			B32	Serkiz and Johnson (1994)
4.64	212.0	5.5	8.42			B33	Serkiz and Johnson (1994)
4.67	180.0	12.6	21.4			B42	Serkiz and Johnson (1994)
3.66	7.1	1.2	3.02			B43	Serkiz and Johnson (1994)
4.09	20.8	8.2	15.1			B51	Serkiz and Johnson (1994)
3.61	2.6					B52	Serkiz and Johnson (1994)
4.69	180.0	3.3	2.39			B52R	Serkiz and Johnson (1994)
3.68	5.6					C13	Serkiz and Johnson (1994)
3.75	28.3	6.4				C14	Serkiz and Johnson (1994)
3.96	27.4		1.28			C22	Serkiz and Johnson (1994)
4.17	823.0	6.4	6.12			C23	Serkiz and Johnson (1994)
5.53	540.0	5.5	2.54			C32	Serkiz and Johnson (1994)
4.64	690.0	6.1	8.54			C33	Serkiz and Johnson (1994)
5.27	1,400.0	7.9	11.4			C42	Serkiz and Johnson (1994)
4.51	460.0	3	5.04			C43	Serkiz and Johnson (1994)
6.78	690.0	5.3	1.96			D13	Serkiz and Johnson (1994)
4.14	26.6					D13RA	Serkiz and Johnson (1994)
	22.6	2	2.55			D13RB	Serkiz and Johnson (1994)
4	650.0	10.5	11.4			E13	Serkiz and Johnson (1994)
4.04	190.0	4.5	8.5			E14	Serkiz and Johnson (1994)
4.32	310.0	3.9	13.3			E23R	Serkiz and Johnson (1994)
3.87	360.0	7.3	13.8			E24	Serkiz and Johnson (1994)
4.27	470.0	6.5	11.5			E33	Serkiz and Johnson (1994)

pH	U Kd (ml/g)	Clay Cont. (wt.%)	CEC (meq/100g)	Surface Area (m <sup>2</sup> /g)	Solution	Soil Identification	Reference / Comments
4.05	270.0	3.7	10.5			E34	Serkiz and Johnson (1994)
5.27	870.0	31.8	20.6			E41	Serkiz and Johnson (1994)
4.87	630.0	14.5	20.6			E42	Serkiz and Johnson (1994)
4.3	690.0	15.5	16.1			F12	Serkiz and Johnson (1994)
4.9	2,200.0		8.51			F13	Serkiz and Johnson (1994)
4.69	1,200.0	8.1	7.48			F22	Serkiz and Johnson (1994)
6.48	950.0	13	11.6			F23	Serkiz and Johnson (1994)
4.85	660.0	14.2	15.1			F32	Serkiz and Johnson (1994)
4.77	220.0	18.3	13.6			F33	Serkiz and Johnson (1994)
5.2	910.0	17.2	11.8			F42	Serkiz and Johnson (1994)
4.12	700.0	14.2				F43	Serkiz and Johnson (1994)
5.91	600.0	42.2	19.9			F52	Serkiz and Johnson (1994)
5.63	960.0	16.3	13.3			F53	Serkiz and Johnson (1994)

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