CONF-960212--46
SAND96-0377C

SCOPING ANALYSIS OF TOXIC METAL PERFORMANCE IN DOE LOW-LEVEL WASTE DISPOSAL FACILITIES[a]

RECEIVED
FEB 27 1996
OSTI

Robert D. Waters, Dmitri A. Bougai, Phillip I. Pohl
Sandia National Laboratories, Albuquerque, New Mexico 87185-0734

ABSTRACT

This study provides a scoping safety assessment for disposal of toxic metals contained in Department of Energy (DOE) mixed low-level waste (MLLW) at six DOE sites that currently have low-level waste (LLW) disposal facilities - Savannah River Site, Oak Ridge Reservation, Los Alamos National Laboratory, Hanford Reservation, Nevada Test Site, and Idaho National Engineering Laboratory. The study has focused on the groundwater contaminant pathway, which is considered to be the dominant human exposure pathway from shallow land MLLW disposal. A simple and conservative transport analysis has been performed using site hydrological data to calculate site-specific "permissible" concentrations of toxic metals in grout-immobilized waste. These concentrations are calculated such that, when toxic metals are leached from the disposal facility by infiltrating water and attenuated in local ground-water system, the toxic metal concentrations in groundwater below the disposal facility do not exceed the Maximum Contaminant Levels as stated in the National Primary Drinking Water Regulation. The analysis shows that arid sites allow about 100 times higher toxic metal concentrations in stabilized waste leachate than humid sites. From the limited available data on toxic metal concentrations in DOE MLLW, a margin of protection appears to exist in most cases when stabilized wastes containing toxic metals are disposed of at the DOE sites under analysis. Possible exceptions to this conclusion are arsenic, chromium, selenium, and mercury when disposed of at some humid sites such as the Oak Ridge Reservation. This analysis also demonstrates that the U.S. Environmental Protection Agency's prescriptive regulatory approach that defines rigid waste treatment standards does not inherently account for the variety of disposal environments encountered nationwide and may result in either underprotection of groundwater resources (at humid sites) or an excessive margin of protection (at arid sites).

INTRODUCTION

Six DOE sites - Savannah River Site (SRS), Oak Ridge Reservation (ORR), Los Alamos National Laboratory (LANL), Hanford Reservation (HR), Nevada Test Site (NTS), and Idaho National Engineering Laboratory (INEL) currently dispose of low-level waste (LLW), and each is planning to dispose of mixed low-level waste (MLLW) [1]. This study has focused on the groundwater contaminant pathway, which is generally considered to be the dominant human exposure pathway from shallow land disposal facilities such as these [2]. A simple and conservative transport analysis has been performed to estimate site-specific "permissible" toxic metal concentrations in stabilized waste. Toxic metals present in MLLW were identified and the permissible toxic metal concentrations in groundwater at the site performance boundary (MCLs) were defined. From these concentrations the disposal facility "permissible" toxic metal concentrations in stabilized waste for the disposal facility were derived based on the estimated concentration attenuation

---

[a] This work was performed at Sandia National Laboratories, which is operated for the U.S. Department of Energy under Contract No. DE-AC04-94AL85000.

provided by the waste form and on environmental transport from the disposal facility to the performance boundary using the conceptual groundwater pathway model described below. These "permissible" stabilized waste concentrations were then compared with data on toxic metal concentrations in DOE MLLW to assess the feasibility of disposal of toxic metal contaminated wastes in DOE LLW disposal facilities. The results of the analysis are used to discuss the U.S. Environmental Protection Agency (EPA) regulation related to disposal of chemotoxic wastes.

Requirements for managing DOE LLW are established in DOE Order 5820.2A[3]. While the Order does not give specific guidance on toxic components of mixed waste, it requires that the hazardous component of DOE MLLW be managed to conform to the Resource Conservation and Recovery Act (RCRA) requirements of appropriate regional authorities. The list of toxic metals regulated under the RCRA (commonly referred to as RCRA metals) is given in Table I. The toxicity of waste is determined by the Toxicity Characteristic Leaching Procedure (TCLP), a test of the leachability of solids containing any one of a list of specific contaminants. A concentration of RCRA metal in the extract above its regulatory level (column 2 of Table I) renders the waste hazardous.

Place Table I here

The Hazardous and Solid Waste Amendments (HSWA) to the RCRA prohibit land disposal of untreated hazardous wastes. This portion of the HSWA is referred to as the Land Disposal Restrictions (LDR). Treatment standards for wastes containing toxic metals have been established by EPA in the form of maximum permissible hazardous constituent concentrations in waste TCLP extracts. Corresponding concentrations are listed in column 3 (LDRs for characteristic wastes) and column 4 (universal treatment standards (UTS) for listed wastes) of Table I. EPA has recently proposed that the metal UTS replace LDR treatment standards for characteristic metal wastes in order to reduce confusion and make compliance easier (60 FR 43654, August 22, 1995). This proposed rule is not yet finalized.

TECHNICAL APPROACH

Model Assumptions

In accordance with performance assessment guidelines established by the DOE Performance Assessment Task Team[4]: (1) a buffer zone of 100 m is used in assessing compliance with the performance objective for protection of groundwater resources, and (2) the requirement in 5820.2A to "protect groundwater resources consistent with federal, state, and local requirements" is taken to mean that the human exposure benchmarks are Maximum Contamination Levels (MCLs) based on 40 CFR 141, the National Primary Drinking Water Regulations. The same document recommends a 10,000-year period of compliance for the performance objectives for groundwater protection.

The MCL values for RCRA metals that are used as human exposure benchmarks in the analysis are listed in column 5 of Table I. Arsenic (As), lead (Pb) and silver (Ag) do not currently have MCLs. The MCL values promulgated in 40 CFR 141.11, which were effective until December 7, 1992, were used for As and Pb. Silver is not considered in the analysis because it does not have an

MCL, and the only potential adverse effect from exposure to Ag in drinking water is argyria (a discoloration of the skin), considered a cosmetic effect which does not impair health (56 FR 3526, January 30, 1991).

Immobilization/solidification techniques using cement-based grouts are considered the most common and universal option for the treatment and ultimate disposal of MLLW for variety of waste matrices [5,6]. In particular, this stabilization technique is likely to be applied to the majority of MLLW streams containing toxic metals (i.e., inorganic solids and aqueous liquids[7]). Therefore, the analysis considers cement-based grout as a reference waste form for stabilization of the MLLW.

Because the uncertainty in hydrogeologic and geochemical parameters is frequently large even at well characterized sites, a simple and conservative groundwater pathway model[6], shown in Figure 1, has been used in this analysis. The following major assumptions were used[6]:

Source Term Assumptions. (1) The waste form was grout-stabilized. (2) Effects of contaminant solubility limits were neglected. (3) Performance of the disposal facility's engineered barriers is limited in time. After failure of the engineered barriers, the infiltration through the disposal facility is equal to the natural infiltration through local soils. (4) A continuous contaminant source was assumed for the transport calculations.

Environmental Transport Assumptions. (1) A one-dimensional analysis was performed. (2) Leachate dilution with groundwater was the only concentration attenuation mechanism accounted for. The contaminant-plume mixing depth in the saturated zone used in the "mixing cell" dilution model was estimated from either (a) constraining hydrogeology (e.g., thin aquifers, sequences of units of high and low permeability) or (b) vertical transverse solute dispersion. The last method was based on a one-dimensional flow, three-dimensional transport model[8]. (3) Effects of longitudinal dispersion were neglected. (Once a continuous source model is assumed, this assumption is of minor importance.) (4) Fractured flow in the vadose zone was addressed by removing the fractured section from the stratigraphic column, implying complete and instantaneous transport through the fractured zone. (5) Fracture flow in the saturated zone was treated as an equivalent matrix porosity.

Place Figure 1 here

Calculations

The contaminant concentration attenuation in the groundwater pathway was calculated by assuming a mixing cell beneath the disposal facility in which contaminated leachate is mixed with clean groundwater (Figure 1). The resulting concentration in the groundwater is calculated based on mass balance with the following formula:

$$C_{gw} = C_l \frac{Q_l}{Q_{gw} + Q_l}, \quad Q_l = abq_l, \; Q_{gw} = d_m b q_{gw} \qquad (1)$$

where C_l is the leachate concentration (M/L^3), Q_l is the leachate volumetric flux (L^3/T), Q_{gw} is the groundwater volumetric flux (L^3/T), a is the length of the disposal facility parallel to the groundwater flow (L), b is the width of the disposal facility perpendicular to groundwater flow (L), q_l is the infiltration rate through the disposal facility (L/T), d_m is the mixing depth in the aquifer (L), and q_{gw} is the groundwater Darcy velocity (L/T). The parameters a and b were assumed to be 50 m[6]. Attenuation caused by environmental transport was represented by a dimensionless concentration reduction factor (CRF_{DW})

$$CRF_{DW} = \frac{C_l}{C_{gw}} = \frac{d_m q_{gw} + a q_l}{a q_l} \quad (2)$$

The contaminant arrival time at the 100-m performance boundary was calculated as the sum of the retarded contaminant travel time in the vadose (t_v) and saturated zones (t_s):

$$t = t_v + t_s, \; t_v = \frac{l_v \theta_v R_v}{q_l}, \; t_s = \frac{l_s n R_s}{q_{gw}}, \quad (3)$$

where l_v is the depth from the disposal facility to the saturated zone (L), l_s=100 m is the distance between the disposal facility and performance boundary, the θ_v and n are the volumetric moisture content in the vadose zone and the porosity in the saturated zone, respectively (dimensionless), and the retardation factors R_v and R_s (dimensionless) for the vadose and saturated zones are given by

$$R_v = 1 + \frac{K_{d,v}\,\rho_{b,v}}{\theta_v}, \quad R_s = 1 + \frac{K_{d,s}\,\rho_{b,s}}{n} \qquad (4)$$

where $K_{d,v}$ and $K_{d,s}$ are the metal-soil distribution coefficients (L^3/M) (discussed in the next section), and $\rho_{b,v}$ and $\rho_{b,s}$ are the soil bulk densities (M/L^3) for the vadose and saturated zones, respectively.

Similar to the attenuation for environmental transport, the toxic metal concentration attenuation provided by the waste form is represented by the source concentration reduction factor, CRF_S. The CRF_S was estimated based on the assumption that the toxic metal concentration in the waste form leachate is determined by desorption into infiltrating water[6]:

$$CRF_S = \frac{C_w}{C_l} = \frac{\theta_G + K_{d,G}\,\rho_G}{f_m} \quad (5)$$

where C_w is the toxic metal concentration in the grouted waste form averaged over the entire volume of waste in the disposal facility (M/L^3), θ_G is the volumetric water content of the grouted waste form (dimensionless), $K_{d,G}$ is the distribution coefficient of the toxic metal in the grout (L^3/M), ρ_G is the bulk density of the grout waste form (M/L^3), and f_m is the mixing fraction, defined as the ratio of the volume of disposed waste in a unit volume of the facility (dimensionless). The following parameter values have been used in the calculations for equation (5)[6]: $\theta_G = 0.3$, $\rho_G = 1.76$ g/cm^3, $f_m = 0.66$.

Once the CRF_{DW} and CRF_S are calculated for each site and each toxic metal, the permissible waste form leachate concentrations (C_{pl}) and permissible concentrations in stabilized waste (C_{pw}) were calculated for the toxic metals as

$$C_{pl} = \text{MCL} \times CRF_{DW} \quad (6)$$
$$C_{pw} = C_{pl} \times CRF_S = \text{MCL} \times CRF_{DW} \times CRF_S \quad (7)$$

Sorption Parameters for Toxic Metals

The RCRA toxic metals can be divided into three broad classes: (1) cationic elements (Ba, Cd, Pb, Hg), (2) anionic elements (As), and (3) redox-sensitive elements (Cr, Se)[9]. In most cases, adsorption and precipitation cause very low mobility of *cationic* species in subsurface environments. For *anionic* species, adsorption and ion exchange will cause relatively little retardation in the subsurface, though oxyanions (e.g., AsO_3^{3-}, CrO_4^{2-}, SeO_3^{2-}) are adsorbed specifically by mineralogical surfaces which carry a positive charge. Chemical speciation and mobility of the *redox sensitive elements* are strongly dependent on specific hydrochemical conditions.

Whenever site-specific RCRA metal sorption parameters were available, these were used in transport calculations. However, for a number of metals, site-specific sorption data were lacking. To provide the model with input parameters in the latter case, a generic toxic metal distribution coefficient (K_d) list was compiled (Table II). Literature review revealed applicable quantitative sorption data for some RCRA metals that are very limited for soils and especially for grout matrixes. In the latter case, values for sorption parameters have been assigned based on modeling or empirical relationships. The mercury (Hg) geosphere and grout K_d values are based on predictions of the MINTEQA2 geochemical model for the "medium pH of solution, medium in iron oxides and natural organic content matrix" conditions and "high pH of solution, low in iron oxides and natural organic content matrix" conditions respectively [13]. As and Cr were assigned grout K_d values based on empirical pH-dependent "metal-sediment" partitioning relationships[15] assuming pH=12.5. Grout K_d values for the toxic metals are rounded to the nearest order of magnitude so as not to imply more precision in estimated values than is justified [6].

Place Table II here

RESULTS AND DISCUSSION

Toxic Metal Concentrations in DOE MLLW

Very limited data are available on actual concentrations of toxic metals in DOE MLLW. Waste concentrations of RCRA toxic metals in several Oak Ridge Reservation MLLW waste streams are presented in Table III. Whether these data are representative of the entire DOE MLLW inventory is unknown, although similar processes should provide similar waste streams. The final column of Table III gives the weighted averages of all the waste streams.

Place Table III here

Site-Specific Analyses

The summary[6] of the site-specific models and data for the six DOE sites considered in this analysis is presented in Table IV. Due to the extremely arid nature of the Nevada Test Site hydrogeological environment, a groundwater pathway is not believed to exist, implying total groundwater protection at this site[6]. Therefore, no water pathway analysis was included in this work for the NTS. Calculated site-specific concentration reduction factors due to waste leachate mixing with groundwater (Table IV) show that leachate dilution provided by arid sites (INEL and LANL) is about two orders of magnitude greater than for the humid sites (ORR and SRS). The semi-arid Hanford Site has an intermediate dilution value. Estimated site-specific CRF_{DW} (Table IV) and CRF_S (Table II) were multiplied by the RCRA metal MCLs (equations 6 and 7) to produce site-specific permissible leachate concentrations and stabilized waste concentrations for the RCRA metals (Table V). Values in Table V can be used as the basis for guidance to establish site-specific waste form performance criteria and waste acceptance criteria.

Place Table IV here

The travel time calculations (Table V) show that, except for a few estimates (e.g., Ba and Se travel times for the Hanford Site), characteristic RCRA metal travel times for the arid and semi-arid sites exceed the DOE recommended LLW disposal compliance period of 10,000 years[4]. For humid sites, the contaminant arrival times at the performance boundary will occur at least 500 years beyond the assumed performance of engineered barriers, with most of the contaminant travel times falling within the higher range of thousands and tens of thousands years.

The last column of Table V shows estimated toxic metal concentrations in the example stabilized ORR waste. These estimates are based on data in Table II, assuming that the grout to MLLW mixing ratio is 1:1 and that the stabilized waste density is the same as the grout density. Assuming that RCRA metal concentrations in these example waste streams are typical for DOE sites, the data in Table V show that disposal of MLLW at the sites under analysis will satisfy or nearly satisfy groundwater protection requirements for most toxic metals. Exceptions are the anionic and redox-sensitive elements As, Cr and Se, and Hg. The concentrations of these metals are approximately an order of magnitude higher than the estimated permissible concentrations at some humid sites (e.g., ORR). Although this analysis is conservative, it indicates that care must be taken when disposing of As, Cr, Se, and Hg in humid environments. Waste streams containing a large amount of these elements may require enhanced stabilization (other than grout stabilization) prior to disposal, or even relocation for disposal to a semi-arid or arid environment. However, more detailed analyses that take credit for additional attenuation factors may provide higher permissible waste concentrations.

Place Table V here

Implications for Regulation of Chemotoxic Waste Disposal

The rationale for the EPA TCLP-based TC, LDR, and UTS standards (Table I) is that corresponding concentration levels in the TCLP waste extract are considered to be an upper bound of contaminant concentrations in the disposed waste leachate under field conditions (55 FR 11862, March 22, 1990). The TC levels for toxic metals have been established as 100 times the MCLs which were in effect at the time of the rulemaking, with a generic 100-fold dilution/attenuation factor estimating attenuation of the toxic constituent leachate concentration in groundwater during subsurface transport from the disposal facility to the point of human exposure (45 FR 33084, May 19, 1980; 55 FR 1862, March 22, 1990). The LDR and UTS standards are also based on technology (i.e., based on performance of the Best Demonstrated Available Technology [BDAT] for hazardous waste treatment) (55 FR 22689, June 1, 1990; 59 FR 47982, September 19, 1994). However, the health risk considerations are implicit in the EPA rulemaking, because, with the exception of Se, the LDR standards for toxic metals are exactly the same as the TC levels.

Comparison of the site-specific permissible leachate concentrations estimated in this study (Table V) with TC, LDR and UTS standards (Table I) shows that, except for a few species (e.g., As), the UTS standards compare well with the estimates of permissible leachate concentrations for humid sites and will therefore ensure groundwater protection for most disposal environments, including humid ones; the LDR and TC standards will provide adequate protection at semi-arid and arid sites only. At the same time, the UTS waste treatment standards provide unnecessarily large margins of groundwater protection in arid disposal environments. Thus, the current EPA regulatory approach of setting a single waste treatment standard inherently does not account for a variety of hydrogeological disposal environments encountered nationwide. The result of such regulatory action is either potential underprotection of humid environments or unnecessarily high protection of arid sites. A performance-based regulatory approach, consisting of setting a performance objective to be fulfilled in every regulatory action, would provide a much more consistent, flexible, and cost-effective way of protecting human health and the environment [2].

CONCLUSIONS

Arid sites appear to provide a greater degree of groundwater protection from land disposal of toxic metals because of higher attenuation and longer contaminant travel times. Arid sites also allow about 100 times greater leachate concentrations for stabilized waste compared to humid sites (e.g., Los Alamos National Laboratory vs. Oak Ridge Reservation).

Most RCRA toxic metals are, in general, immobile in the subsurface environment, even at relatively humid sites. Exceptions to this conclusion are anionic species (e.g., arsenic and selenium), which tend to have lower K_d values. However, more specific analyses are needed to better determine the environmental fate of these and other metals.

From the limited available data on toxic metal concentrations in DOE MLLW and the estimated performance of grout-stabilized process waste, a margin of protection appears to be present in most cases when wastes containing RCRA toxic metals are disposed of at the evaluated DOE sites. Exceptions to this conclusion are As, Se, Cr and Hg when disposed at some humid sites. Waste streams containing large amounts of these elements may require enhanced stabilization (other than grout stabilization) prior to disposal, or relocation for disposal to semi-arid or arid sites.

The EPA's prescriptive approach of explicitly defining regulatory levels for toxic waste does not inherently account for the variety of disposal environments encountered nationwide, and may potentially result in either underprotection of groundwater resources at humid sites (e.g., TC levels) or an unnecessarily large margin of protection at arid sites (e.g., UTS levels).

ACKNOWLEDGMENTS

This work was sponsored in part by the United States National Academy of Sciences/National Research Council through the Collaboration in Basic Sciences and Engineering Program between the Eastern European countries and the United States.

REFERENCES

1. DOE (Department of Energy), "Framework for DOE Low-Level and Mixed Low-Level Waste Disposal: Current Overview", DOE/ID-10484, Washington, DC: U.S. Department of Energy, Office of Waste Management (1994).

2. NUMARC (Nuclear Management and Resources Council, Inc.), "The Management of Mixed Low-Level Radioactive Waste in the Nuclear Power Industry", Nuclear Management and Resources Council, Inc. (1990).

3. DOE (Department of Energy), "Low-Level Waste", Chap. III in "Radioactive Waste Management", Order 5820.2A (1988).

4. D.E. Wood, R.U. Curl, D.R. Armstrong, et al., "Performance Assessment Task Team Progress Report", DOE/LLW-157, Rev. 1, Idaho Falls, ID: Idaho National Engineering Laboratory (1994).

5. DOE (Department of Energy), "U.S. Department of Energy Interim Mixed Waste Inventory Report: Waste Streams, Treatment Capacities and Technologies", DOE/NBM-1100, Vol.1 (1993).

6. DOE (Department of Energy), "Performance Evaluation of the Capabilities of DOE Sites for Disposal of Mixed Low-Level Waste", Predecisional Draft, DOE/ID-10521/1-3 (3 volumes), Albuquerque, NM: U.S. Department of Energy, Office of Waste Management (1995).

7. W.A. Ross, M.R.Elmore, C.L.Warner et al., "Locations, Volumes, and Characteristics of DOE's Mixed Low-Level Wastes", Waste Management-92, Vol.2, Tucson, AZ, March 1-5, 1992, p.1127 (1992).

8. M.S.Y. Chu, M.W. Kozak, J.E. Campbell, and B.M. Thompson, "A Self-Teaching Curriculum for the NRC/SNL Low-Level Waste Performance Assessment Methodology", NUREG/CR-5539, SAND90-0585, Albuquerque, NM: Sandia National Laboratories (1991).

9. D.Rai, and J.M.Zachara, "Chemical Attenuation Rates, Coefficients, and Constants in Leachate Migration. Volume 1: A Critical Review", EPRI EA-3356, Vol.1, Palo Alto, California: Electric Power Research Institute (1984).

10. III. C.F. Baes, and R.D.Sharp, "A Proposal for Estimation of Soil Leaching, and Leaching Constants for Use in Assessment Models", J.Environ.Qual., Vol.12, pp.17-28 (1983).

11. J. Ashton, and T.J. Sumerling, "Biosphere Database for Assessments of Radioactive Waste Disposal", UK DOE rep. DOE/RW/88.083, Dept. of Environment, London, UK (1988).

12. M.I. Sheppard, and D.H.Thibault, "Default Soil Solid/Liquid Partition Coefficients, Kd-s, for Four Major Soil Types: A Compendium", Health Physics, Vol.59, pp.471-482 (1990).

13. "Background Document for Finite Source Methodology for Waste Containing Metals", Washington, DC 20460: U.S. Environmental Protection Agency, Office of Solid Waste (1992).

14. T.T.Vandergraaf, and K.V. Ticknor, "A Compilation of Sorption Coefficients Used in the Geosphere Model of SYVAC for the 1990 Assessment of the Whiteshell Research Area", AECL-10546, Pinawa, Manitoba: Whitshell Laboratories (1994).

15. N.T. Loux, C.R.Chafin, and S.M.Hassan, "Statistics of Aquifer Material Properties and Empirical pH-dependent Partitioning Relationships for As, Ba, Be, Cd, Cr, Cu, Hg, Ni, Pb, Sb, Se, Tl and Zn", Athens, GA: U.S. Environmental Protection Agency (1990).

Table I. Regulatory levels for RCRA toxic metals

1	2	3	4	5
RCRA Metal	Toxicity Characteristics (TC)[a] (TCLP extr.;mg/L)	Land Disposal Restrictions (LDR)[b] (TCLP extr.; mg/L)	Universal Treatment Standards (UTS)[c] (TCLP extr.; mg/L)	Maximum Contaminant Level (MCL)[d] (mg/L)
Arsenic, As	5.0	5.0	5.0	0.05 [e]
Barium, Ba	100.0	100.0	7.6	2
Cadmium, Cd	1.0	1.0	0.19	0.005
Chromium, Cr	5.0	5.0	0.86	0.1
Lead, Pb	5.0	5.0	0.37	0.05 [e]
Mercury, Hg	0.2	0.2	0.025	0.002
Selenium, Se	1.0	5.7	0.16	0.05
Silver, Ag	5.0	5.0	0.3	-

(a) 40 CFR 261.24; (b) 40 CFR 268.41; (c) 40 CFR 268.48; (d) 40 CFR 141.62 unless noted; (e) 40 CFR 141.11

Table II. Sorption parameters for the RCRA toxic metals

RCRA Metal	Geosphere K_d (mL/g)	Reference for the geosphere K_d	Grout K_d (mL/g)	Reference for the grout K_d	Estimated CRF_S
As	6	[10]	10	[15]	27
Ba	500	[11]	10	[6]	27
Cd	80	[12]	100	[6]	270
Cr	70	[12]	10	[15]	27
Pb	270	[12]	100	[6]	270
Hg	140	[13]	10	[13]	27
Se	8	[14]	1	[6]	3.1

Table III. RCRA toxic metal concentrations in selected waste streams at the Oak Ridge Reservation (concentrations in ppm) (From Y. McClellan, Sandia National Laboratories, 1995, work in preparation)

Stream ID	dp-w006	dp-w011	dp-w019	dp-w008	dp-w046	dp-2042	yp-w001	yp-w005	Avg. (ppm)
Matrix	Sludge	Sludge	Pond Sludge	Organics	Tank Sludge	Tank Sludge	Soils	Sludges	
Mass (kg)	316,000	24,600,000	9,800,000	15,000	854,000	200,000	664,000	9,911,000	
As		0.5	109	8	44	3.18	2.51	20	28.5
Ba	20.5	1	61.025	12	69	241	86.96	131.35	45.2
Cd	6.8	0.03	2.3	0.48	18	6.64	0.28	32.4	7.8
Cr	93	0.1	171	1.6	93	519	21.7	160.4	75.4
Pb	27	0.5	77.4	20	262	676.44	15.42	361.7	102.1
Hg	2.4	0.02	21.95	3.7	30	118	0.39	1.65	6.1
Se		0.05	3.275	8		1.18	2.51	15	4.0

Table IV. Site-specific hydrogeological parameters and resulting CRF_{DW}. [6]

Parameter	Site					
	HR	INEL	LANL		ORR	SRS
Unsaturated zone			Bandelier Tuff	Puye Formation		
Infiltration rate, m/y	0.05	0.07	0.02		0.18	0.4
Vadoze zone thickness, m	72	50	155	178	2	17
Volumetric Moisture Content	0.09	0.17	0.05	0.05	0.39	0.2
Soil bulk density, g/cm^3	1.5	1.5	1.2	1.9	1.35	1.8
Saturated zone				Puye Formation		
Darcy velocity, m/y	22	56		23	2.9	8.1
Porosity	0.31	0.1		0.3	0.035	0.3
Soil bulk density, g/cm^3	1.6	1.9		1.5	1.35	1.8
Mixing Depth, m	5	12		9	3	10
Resulting CRF_{DW}	68	290	310		3	8
Unsaturated/Saturated Zone K_d, *mL/g*						
Ba	1	60	950		3000	5
Cd	100	100/10	80		200	8
Pb	100		25			100
Se	0		2			10

Table V. Site-specific permissible waste form leachate concentrations (in bold, mg/L), permissible stabilized waste concentrations (in bold italic, mg/L), and contaminant travel times from the disposal facility to the performance boundary (in parentheses, years).

Toxic Metal	Site					ORR Stabilized Waste (mg/L)
	HR	INEL	LANL	ORR	SRS	
As	**3.4**; ***92*** (13,000)	**14.5**; ***390*** (6,600)	**15.5**; ***420*** (160,000)	**0.15**; ***4*** (400)	**0.4**; ***11*** (600)	*36*
Ba	**135**; ***3700*** (2,300)	**580**; ***15700*** (65,000)	**620**; ***16700*** (17,000,000)	**6**; ***162*** (180,000)	**15**; ***432*** (500)	*56*
Cd	**0.34**; ***92*** (220,000)	**1.45**; ***390*** (86,000)	**1.55**; ***420*** (2,100,000)	**0.015**; ***4*** (12,000)	**0.04**; ***11*** (800)	*9.8*
Cr	**6.8**; ***180*** (150,000)	**29**; ***780*** (75,000)	**31**; ***840*** (1,800,000)	**0.30**; ***8.1*** (4,300)	**0.76**; ***22*** (6,900)	*94*
Pb	**3.4**; ***920*** (220,000)	**14.5**; ***3900*** (110,000)	**15.5**; ***4200*** (4,800,000)	**0.15**; ***40*** (17,000)	**0.38**; ***110*** (9,900)	*130*
Hg	**0.14**; ***3.7*** (300,000)	**0.58**; ***15.7*** (150,000)	**0.62**; ***16.7*** (3,700,000)	**0.006**; ***0.16*** (8,600)	**0.015**; ***0.43*** (14,000)	*7.6*
Se	**3.4**; ***10.5*** (100)	**14.5**; ***45*** (8,700)	**15.5**; ***48*** (160,000)	**0.15**; ***0.5*** (500)	**0.38**; ***1.2*** (500)	*5.0*

Note: Travel time estimates do not account for the performance of engineering barriers.