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October 28, 1981

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Subject: Site Closure Studies, West Chicago Thorium Plant
LAW Project Number 710022

Dear Mr. Denny:

Law Engineering is pleased to submit the attached report presenting the results of closure studies at your West Chicago, Illinois facility.

This report provides a summary of our recommendations for closure of the site, and an assessment of the effects of the proposed closure on ground water quality at the site.

We appreciate the opportunity of working with you on this project. If we can be of any further assistance, please do not hesitate to contact us.

Very truly yours,

LAW ENGINEERING TESTING COMPANY

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Enclosure

Final Report

WEST CHICAGO THORIUM PLANT CLOSURE

Submitted to

KERR-McGEE CORPORATION

Denver, Colorado
October 28, 1981
Project No. 710022

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1.0 INTRODUCTION

1.1 BACKGROUND

The West Chicago plant, located in West Chicago, Illinois, operated from 1932 to 1973 milling and producing thorium and rare earth elements from ores and ore concentrates. The present plant site has been utilized for other industrial purposes by previous owners. The Union Tool Company manufactured drilling equipment at a five-acre portion of the present factory site from the mid-1880's to 1931. In 1931, the Lindsay Light Company acquired the property. Lindsay was acquired by American Potash and Chemical Corporation in 1958, which in turn was acquired by Kerr-McGee in 1967.

Lindsay Light Company began chemical operations at the West Chicago site in 1932. Lindsay acquired land for on-site disposal operations in 1952 and 1955. Thorium was produced at the site between 1932 and 1973. Various rare earth products were produced until 1973, when Kerr-McGee determined that continued plant operation was not economically viable.

Ore residues and process wastes (including low-level radioactive thorium residues) which were produced as waste products during the years of operation were deposited in mounds and ponds at a 27-acre disposal site within the 43-acre West Chicago site. The wastes are still located where they were originally deposited, although evidence suggests that they are chemically and physically stabilized.

Kerr-McGee has presented a plan to the Nuclear Regulatory Commission for decommissioning the facility and restoring the property. This plan currently is being reviewed. Based upon studies which continued after the submission of this plan, Kerr McGee intends to suggest modifications to this plan. The purposes of the modifications are to make the plan more consistent with current practices in waste disposal, and to make the disposal scheme more compatible with the hydrologic system at the proposed site. The modifications are desirable because the state-of-the art in waste disposal has advanced significantly since the preparation of the initial plan, and continuing site studies have improved the understanding of the site hydrologic system.

The modified plan is similar in concept to the original plan except for the following features:

- The modified plan provides an improved cell cover. The improved cover is designed to resist deterioration resulting from climatic conditions, and to limit the amount of infiltration into the cell. The original plan provided a thinner cell cover which might have been subject to deterioration during extreme climatic conditions.

- The modified plan provides a complete encapsulation of all waste materials by a barrier of low-permeability materials.

This encapsulation is achieved by the utilization of a cell liner beneath the entire disposal cell. Along the sides of the cell, the cell cover is joined to the cell liner. The original plan provided encapsulation of only a portion of the wastes.

The design of the cell cover and cell liner is coordinated to minimize the accumulation in the future of leachate within the cell. The original plan lacked this feature, and the thinner cell cover relative to the thick liner would have tended to encourage the accumulation of fluids within the cell.

The modified design allows the controlled release of leachate produced by controlled infiltration through the cap. The cell liner has a higher hydraulic conductivity than the cell cap, and allows normal infiltration to pass through the cell. The treatment and neutralization of the waste in the cell further protect against potentially environmental harmful discharges from the site.

The modified plan includes a leachate monitoring system within the completed cell. This system will allow the performance of the cell to be monitored for a period after closure to assure proper functioning of the disposal system.

The purpose of this report is to summarize the salient features of the proposed modifications and present the bases from which the modifications were derived. This report focuses only upon the hydrologic aspects of the proposed stabilization plan. Other studies conducted by Kerr-McGee describe additional aspects of the plan such as the control and monitoring of airborne emissions during and after plan implementation, and the control of personnel exposures during stabilization.

1.2 ADDITIONAL SITE STUDIES

Since the submission of the original stabilization plan on August, 1979, Kerr-McGee has continued a program of investigation of site and waste characteristics. The results of these studies are summarized in the following documents which were utilized in the preparation of this report.

- Soil Testing Services, 1981, Geohydrological Program - Kerr-McGee Corporation, West Chicago Site 798, Volumes I and II, STS Project No. 18943-A.
- Law Engineering Testing Company, "Hydrologic Studies - West Chicago Thorium Plant", August 24, 1981.
- Hajek, B.F., "Characterization of Soils: Chemical, Mineralogical, Physical and Water Retention Properties - Kerr-McGee Soils Study", February 27, 1981.

2.0 OBJECTIVES OF THE PLAN

The objectives of the proposed stabilization plan are to provide conditions appropriate for the permanent disposal of the waste materials at the site in a manner which will minimize short- and long-term deleterious effects upon the environment, and to minimize the long-term commitment of human and natural resources necessary to assure continued protection of the environment.

These objectives are to be achieved through the following actions:

- Wastes will be disposed only in a portion of the present site. This allows the immediate return of a substantial portion of the site area to productive use.
- The disposal system design is structured so that the performance of its components can be confidently predicted or estimated in a conservative manner.
- The disposal system design is structured to provide a stable repository for the wastes which will require a minimum of monitoring and maintenance to assure its continued acceptable performance.
- The disposal system design is structured to prevent the future generation of secondary wastes (leachate) in quantities which would require treatment and disposal.
- The disposal system design includes facilities to allow direct monitoring of system performance for a period after stabilization. This monitoring period is intended to provide additional assurance of proper system performance. In addition, the monitoring system allows for the removal of leachate in the event of unanticipated cell cover failure. Thus, the protection of the environment in the event of system failure during the monitoring period is enhanced.

Shallow groundwater is not presently used as a public or private water supply in the site vicinity. However, it is the intent of the proposed disposal plan to limit contaminant concentrations from the disposal material at the site boundary to small fractions of the contaminant levels specified in the National Interim Primary Drinking Water Regulations or the National Secondary Drinking Water Regulations. Discharges of radioactive contaminants are intended to be limited to values which will result in calculated doses of less than 25 millirems per year to an individual at the site boundary, and concentrations less than drinking water standards at the nearest public drinking water supply.

Currently, the shallow ground water is not used as a public or private water source in the site vicinity (Law Engineering Testing Company, 1981), and the nearest location at which this ground water will enter the surface environment is at Kress Creek. Conservative calculations, presented in Section 4.4 of this report, of contaminant concentrations in the shallow ground water at the site boundary indicate that the above-stated intent will be achieved at the site boundary. Concentrations at Kress Creek or in the underlying Silurian Aquifer would be less than in the shallow ground water at the site boundary.

3.0 COMPONENTS AND DESIGN BASES OF THE PROPOSED PLAN

The following discussion describes the components of the disposal cell, and indicates how the design of the system is in accordance with the achievement of the objectives stated in Section 2. The discussions in this section are qualitative in nature and are intended to illustrate the bases for the selection of the proposed disposal cell design. Computations and analyses which demonstrate the efficiency of the design and justify the chosen design parameters are presented in Section 4.0. A plan view of the disposal cell is shown in Figure 1. Figure 2 shows a typical cross-section of the disposal cell. Figure 3 shows a schematic vertical section of the cell.

3.1 BOTTOM LINER

The lowermost component of the disposal cell is a liner of natural or compacted clayey material with a minimum thickness of two (2) feet. Tests on samples of the natural surficial clayey material indicate that the hydraulic conductivity of this material is approximately 10^{-8} cm/sec. This natural low-conductivity material is ideal for use as the disposal cell bottom liner. Additional compacted materials will be placed in areas where the surficial clayey stratum is thin or absent, and will be chosen from available local materials to provide a bottom liner with the smallest hydraulic conductivity reasonably achievable; however, the maximum hydraulic conductivity of the liner, either natural or compacted, will not exceed 10^{-7} cm/sec. The liner is depicted on Figure 3. Figure 4 shows proposed grades for the liner.

The disposal cell liner is intended to serve several purposes. During the implementation of the stabilization plan, the liner will provide a low-permeability barrier between the exposed waste materials and the underlying water-bearing strata. This barrier will restrict the downward movement of meteoric water which may become slightly contaminated upon contact with the waste materials. Thus, the liner will allow the collection and treatment of the majority of this water and prevent the infiltration of contaminants to the underlying water-bearing strata.

After the disposal cell has been completed, the liner will serve to protect the ground water environment by limiting the amount of seepage through the waste which can reach the underlying water-bearing strata.

The low-hydraulic conductivity of the liner will limit the amount of seepage through the liner. Therefore, in the event that unanticipated quantities of leachate accumulate in the cell, the liner will cause the majority of this leachate to be diverted to the leachate monitoring and collection system (to be discussed later) where it can be detected and removed from the cell.

The liner, as designed, will serve to minimize resources necessary for long-term maintenance of the facility by allowing normally expected leachate quantities to exit the cell through the bottom liner. This feature is achieved by a coordination of the design of the liner and the cell cover, and prevents the accumulation of leachate within the cell which would require continued removal, treatment and disposal. This controlled, safe release of small amounts of leachate would not have been possible in the earlier proposed design incorporating ten feet of compacted clay. Leachate would build up and eventually produce a definite potential environmental hazard.

3.2 LEACHATE MONITORING AND COLLECTION SYSTEM

The leachate monitoring and collection system is positioned between the cell liner and the waste, as shown in Figure 3. Figure 4 shows a plan view of the leachate monitoring system. This system consists of a graded coarse aggregate filter and drain, one-foot thick, within which is placed a network of perforated drain pipes. These pipes drain by gravity to sumps located within the disposal cell. Capped risers extend from the sumps to the ground surface to allow monitoring of fluid levels within the cell.

The aggregate drain will be constructed of materials of high hydraulic conductivity, approximately 10^{-3} cm/sec. The significant difference in hydraulic conductivity between the drain and the cell liner (a factor of 10^4 to 10^5) will cause leachate reaching the liner to be directed into the underdrain system. Thus, the system allows fluid levels within the cell to be monitored and fluids collected in the sump to be removed in the event that objectionable quantities accumulate.

3.3 WASTE CHARACTERISTICS AND PLACEMENT

Wastes to be disposed at the site consist of milling wastes, slightly contaminated earth, building rubble, and relatively small amounts of piping, tubing, machinery parts and drums. Wastes will be placed within the cell in a dense state, and will be mixed to the extent that large voids will not exist within the emplaced wastes. Waste materials, which by their nature and shape could create significant voids, will be crushed or filled to the extent possible with stable materials prior to disposal. This provision is intended to exclude the occurrence of large voids within the disposed waste.

As a consequence of the milling procedures, the materials in the residue pile and the pond sediments are somewhat acidic. Prior to placement in the disposal cell, these materials will be neutralized by mixing with lime or other suitable basic material. The purposes of this treatment are to prevent the development of acidic leachate which may have deleterious effects upon the cell liner, and to reduce the solubility of contaminants within the waste materials.

The waste treatment and placement procedures discussed above serve both to protect the ground water environment and minimize long-term maintenance requirements. Environmental protection is enhanced by reducing the solubility of potential contaminants and rendering the leachate less likely to degrade the cell liner. Maintenance requirements are minimized by the placement of the wastes in a stable form which will not be subject to future deterioration, compaction or collapse. This contributes to the overall stability of the disposal cell.

3.4 CELL COVER

The cell cover serves as the primary barrier between the environment and the stabilized wastes at the site. As such, the cover acts to control emanations of gases from the wastes, to limit infiltration of rainfall into the cell, to control and direct runoff from rainfall falling on the disposal area, and to minimize the likelihood of accidental intrusion into the disposed wastes.

The cover consists of several sub-components described as follows.

3.4.1 Compacted Low-Permeability Soil Cap

The lowermost component of the cover is a cap constructed of clayey soil of low hydraulic conductivity. The soils used for the construction of the cap will be obtained from locally available sources. The character of the soils utilized and the methods of placement and compaction will be controlled to yield a hydraulic conductivity not greater than 10^{-8} cm/sec. The thickness of the cap will be at least two (2) feet. A bentonite additive will be added to the soil if necessary to achieve the specified hydraulic conductivity. The cap will be directly connected to the cell liner to produce a totally encapsulated disposal volume as shown in Figure 2.

The cap will serve to limit the infiltration of percolating water into the disposal cell, and to limit the diffusion of gases from the cell into the environment.

The upper surface of the cap will be sloped at grades of at least one percent to allow water which infiltrates to the upper surface of the cap to drain away from the cell. The configuration of the upper surface of the cap will closely approximate the configuration of the cell cover as shown on Figure 1.

3.4.2 Drainage Layer

Overlying the cell cap is a one-foot thick layer of graded coarse aggregate. The hydraulic conductivity of this layer will be approximately 10^{-3} cm/sec. The contrast between the hydraulic conductivities of the drainage layer and the cap (approximately 10^{5}) will encourage the movement of excess water through the drainage layer and away from the disposal cell. This design, therefore, will divert excessive infiltration away from the cell, eliminating greater than normal recharge volumes incapable of moving through the compacted clayey cap.

The drainage layer will be extended around the sides of the disposal cell and connected to the shallow sand and gravel zone beneath the disposal area. This connection will allow the drainage of diverted water directly into the shallow water-bearing stratum, avoiding the necessity for drainage at the ground surface.

Because of the nature of the materials which will be utilized to construct the drainage layer, the layer will be difficult to excavate and thus will serve the secondary purpose of discouraging accidental intrusion into the wastes.

3.4.3 Soil Cover

The final component of the cell cover is a four-foot thick layer of soil placed above the drainage layer. This soil layer is intended to protect the compacted soil cap from effects of freezing, thawing and erosion. In addition, the layer will provide a growth medium for vegetation, and will serve as an initial barrier to infiltration of precipitation.

The lower three feet of the soil layer will be constructed of the same material and by the same methods of placement and compaction as is the compacted soil cap. However, it is anticipated that subsequent frost penetration, and wetting and drying cycles will prevent the maintenance of the low permeability of this layer. Therefore, the limits upon hydraulic conductivity placed upon this layer will be less severe than that of the compacted soil cap. The hydraulic conductivity of the lower three feet of the soil cover will not exceed 10^{-6} cm/sec. Uncontaminated durable building rubble (brick and concrete) may be placed in the lower two feet of this layer. The size of this material will be restricted to a maximum of one cubic foot and the pieces of rubble will be embedded within the soil matrix. The purpose of the rubble within this zone is to discourage future inadvertent intrusion into the waste by creating a zone through which digging will be difficult.

The upper one foot of the soil cover will be constructed of material suitable for establishment of vegetation. This soil will be selected for resistance to erosion and infiltration and will be compacted to the extent consistent with the eventual establishment of an erosion-resistant grass cover. No permeability limits will be placed upon the compacted topsoil layer. The grass utilized for the vegetative cover will be a type of grass compatible with the local climate, requiring no irrigation and having a shallow, fibrous root system.

3.5 SURFACE WATER DRAINAGE

Surface water from offsite areas will be diverted from the disposal cell by a system of perimeter runoff channels surrounding the disposal area. During construction, these channels direct water to a sediment pond constructed in the southwest corner of the disposal area. The design of the runoff channels and the sediment pond will conform to the drainage code of the City of West Chicago. The sediment pond will be removed as construction is completed.

4.0 TECHNICAL BASES OF DISPOSAL CELL DESIGN

This section presents the results of computations and analyses utilized to evaluate the design of the disposal cell.

4.1 PERCOLATION ESTIMATES

Percolation estimates were made for conditions typical of the site in a natural condition and for the cell cap. These estimates were made using a computer code developed by the U. S. Army Corps of Engineers for the EPA. A description of this code is presented in EPA Publication SW-868, Hydrologic Simulation on Solid Waste Disposal Sites. Climatological data for the West Chicago vicinity for the years 1974 through 1978 inclusive were used in the analyses. Surficial soils were assumed to be silty clay for both cases. The natural condition was modeled as a 36-inch thick layer of this silty clay. The cell cover was modeled as a 48-inch layer of silty clay overlying a 24-inch layer of compacted clay. No account was taken of potential drainage away from the cell through the drainage layer. A "fair" grass cover was assumed for both analyses.

Annual values of computed percolation averaged 4.50 inches for the natural condition and 0.72 inches for the cell cover. Figure 5 shows the computed average monthly percolation for the natural condition. Percolation occurs during the winter and spring when evapotranspiration demands are least. Little percolation occurs during the months of June through November, largely because of high evapotranspiration during this period.

The percolation model used in this analysis does not account for limits upon infiltration resulting from frozen soils. Therefore, the estimates of percolation during the winter months likely are high, and the estimate of annual percolation as a consequence also may be somewhat high. Walton (1970) indicates that ground water runoff during years of near-average precipitation in terrains similar to the West Chicago site range from about 3.3 to 5.2 inches, with a median value of about 3.8 inches. Ground water runoff is precipitation that infiltrates into the soil and percolates into a stream channel. This estimate of ground water runoff, a rough measure of percolation as the term is used above, is in agreement with predicted natural percolation rates.

Figure 6 shows computed average monthly percolation through the proposed cell cover. The total amount of percolation through the cell cover is approximately 16 percent of the amount computed for the natural condition, and occurs during the months of March through August. Percolation amounts are smaller for the cell cover than for the natural condition because of the greater storage capacity of the cell cover and the presence of the low-permeability clay cap. These factors allow more of the water which infiltrates the cover to be returned to the atmosphere by evapotranspiration. Peak percolation rates through the cell cover occur later in the year than peak rates through the natural cover because of the greater thickness of soil through which the water must move, and because of the retarding effect of the low permeability clay cap.

4.2 EROSION RATE ESTIMATES

The calculations of erosion potential on the cell cover were based on the universal soil loss equation (USLE). The equation is (Lutton, 1980):

$$A = R\ K\ L\ S\ C\ P$$

where:
A = average annual soil loss in tons/acre
R = rainfall erosivity index
K = soil erodibility factor, tons/acre
L = slope-length factor
S = slope-steepness factor
C = cover/management factor
P = practice factor

The value of R was determined from an index map (Lutton, 1980). Values of L were determined for each sub-area of the landfill. The area was subdivided on the basis of orientation of the slope and the slope length. L is determined from the percent slope and slope length for each sub-area. The C parameter takes into account the effects of vegetation, crop sequence, management, and agricultural erosion-control practices. On sites which have been freshly covered, without vegetation or erosion control practices, C is approximately equal to one. In this case, a moderate grass cover was assumed; and the value of C is 0.01. The P parameter accounts for erosion-reducing land management practices such as contouring and terracing. In the case of the West Chicago site, no support practice was assumed. The K parameter is the average soil loss for a given soil and is based upon soil texture. For the West Chicago site, the soil texture class silty clay and a soil organic content of 4 percent were used to determine the value of K.

Using these values to calculate A for each sub-area results in an average erosion rate of .12 tons/acre/year, or about 6.2 x 10^{-4} inches per year.

4.3 DEPTH OF FROST PENETRATION

Maximum frost penetration is an important parameter in the design of the cell cover, since a sufficient thickness of soil must be provided above the compacted cap to prevent cap damage. Maximum expected frost depths utilized in this study were determined on the basis of information supplied by Professor Barry Dempsey (personal communication, February 16, 1981) and by Professor George Sowers (Sowers, 1979, and personal communication, September 3, 1981). Professor Dempsey indicated the maximum expected frost depth in the West Chicago area to be about 42 inches under sod. Professor Sowers indicated a conservative design value of 60 inches to be appropriate. A value of 60 inches was utilized in the design of the cell cover.

4.4 PREDICTED CONTAMINANT CONCENTRATIONS IN SHALLOW GROUND WATER

The concentrations of constituents in waste leachate were determined by leachate tests performed by Kerr-McGee. These tests were performed according to RCRA methods. A summary of the results of these tests is presented in Table 1. Concentrations of heavy metals also were determined by tests performed according to procedures specified by the Illinois EPA. The results of these tests are presented in Table 2.

TABLE 1

ANALYSES OF LEACHATE
(RCRA EP Test Methods)

Parameter	Pile 1 (Sludge) avg.	Pile 1 (Sludge) min.	Pile 1 (Sludge) max.	Pile 2 (Tailings) avg.	Pile 2 (Tailings) min.	Pile 2 (Tailings) max.	Sump Residue Composite
Th 232 (pCi/l)	435	0.9	3680	71	0.2	291	0.6
Th 230 (pCi/l)	71	6.0	568	12	3.8	27	7.1
Th 228 (pCi/l)	2996	3.2	28,360	284	3.4	1330	2.4
U (ug/l)	46	12	154	27	10	79	.019
Ra 226 (pCi/l)	7.3	0.5	23	6.7	0.6	27.7	.8
Ra 224 (pCi/l)	40.60	n.d.	247	263	0	1066	1.5
Ag (mg/l)	0.35	<.001	. 160	0.011	<.001	.064	<.001
As (mg/l)	.003	<.001	.008	.004	<.001	.008	.001
Ba (mg/l)	.071	.021	.130	.075	.027	.140	.26
Cd (mg/l)	.021	.004	.062	.053	.004	.320	.028
Cu (mg/l)	.126	<.001	.730	.078	<.001	.23	.18
Cr (mg/l)	.009	<.001	.071	.008	<.001	.027	.001
Fe (mg/l)	14.4	<.001	150	.488	<.001	2.1	.006
Hg (mg/l)	<.001	<.001	<.001	<.001	<.001	<.001	.001
Ni (mg/l)	.483	.004	3.4	.024	.001	.070	.048
Pb (mg/l)	0.96	<.001	4.2	1.02	<.001	3.8	.002
Se (mg/l)	.007	<.001	.017	.007	<.001	.017	.014
Zn (mg/l)	.242	.053	1.5	1.7	.067	19.0	2.6
Ca (mg/l)	235	5.2	450	291	22	530	-
K (mg/l)	10.4	1.3	23	12.3	1.7	33	-
Mg (mg/l)	33.0	0.2	130	18.9	0.2	95	-
Na (mg/l)	140	54	230	136	17	310	-
SO_4 (mg/l)	1512	50	3900	1504	90	3200	675
Cl (mg/l)	6.2	<2	31	<2	<2	10	55
F (mg/l)	13.3	0.68	38.8	8.7	0.83	22.2	15.8
NO_3 (mg/l)	0.23	<0.1	0.8	0.2	<0.1	0.3	9.2

TABLE 2

ANALYSES OF HEAVY METALS IN LEACHATE
(IEPA Analysis Procedures)

Method Parameter	Pile 1 (Sludge) Concentrations			Pile 2 (Tailings) Concentrations		
	avg.	min.	max.	avg.	min.	max.
As	.079	.046	.097	.030	.011	.051
Cd	.002	<.001	.004	.001	<.001	.002
Cu	.063	.044	.083	.026	.014	.036
Cr	.043	.021	.082	.024	.008	.047
Fe	0.22	.19	.25	.16	.077	.23
Ni	.037	.023	.059	.024	.009	.041
Pb	.020	.004	.046	.12	.052	.23
Se	.060	.043	.074	.035	.017	.086
Zn	2.49	.067	16.0	.056	.016	.091

In general, the results of the analyses by the two procedures are comparable. Tests by the RCRA procedure generally yield slightly higher concentrations of heavy metals, although concentrations for arsenic, chromium and selenium are higher in the leachate developed by Illinois EPA procedures.

In order to appraise the effects of the proposed disposal plan upon the shallow ground water at the site, simple transport models developed by the Nuclear Regulatory Commission (Codell and Schreiber, 1979) were used to predict flux and concentrations of selected radionuclides in the shallow water bearing stratum at a point down-gradient of the site near the site boundary.

One of the models calculates the flux of radioactive liquid effluent passing a plane perpendicular to the direction of ground water flow. The other model calculates radionuclide concentrations at points in a uniform aquifer down-gradient from the source.

The flux model is used to calculate the discharge rate of radioactive material crossing a plane perpendicular to the direction of ground water flow. Output from this model can be used to determine concentrations in a surface water body, such as a river or lake. The point concentration model is used to calculate radionuclide concentrations in the aquifer at some point down-gradient of a release. The theoretical development of these models is presented by Codell and Schreiber (1979).

These models were developed for uniform, unidirectional flow undisturbed by sources and sinks. Ground water flow is assumed to be under either water table or confined conditions in a saturated media of constant thickness, with no infiltration. The radioactive source is assumed to be uniformly distributed over an area whose center is the origin of the coordinate system. No consideration is given to processes which occur within the unsaturated zone between the bottom of the cell and the top of the saturated media, and so the model, in essence, assumes that the water is disposed within or at the top of the saturated media. Leaching immediately following disposal is assumed to occur in a uniform fashion over the entire waste disposal area. The models were formulated as analytical solutions to the three-dimensional equation for conservation of mass in porous media. (Radioactive decay is treated separately from the transport computations).

These models are idealizations of the true nature of contaminant transport in ground water. As such, real situations do not fit easily into simplified analytical models. On the other hand, there are not sufficient data to warrant the use of more complex finite difference or finite element transport models of the site, especially for the long time periods of migration of some of the radionuclides.

The selection of coefficients and parameters for the models is probably the most important task for any application. Necessary input parameters include the unidirectional ground water velocity, the aquifer thickness (concentration model), the half-life of each radionuclide, longitudinal and lateral dispersivities, the radionuclide retardation factor for each radionuclide and the down-gradient distance where flux or concentrations are desired. Other parameters are the source concentration and the rate of leaching of the various isotopes.

Ground water velocity is a function of hydraulic conductivity, effective porosity, and hydraulic gradient. Dispersivity is a characteristic property of the porous medium, reflecting in part its nonhomogeneity. The retardation factor is a function of the distribution coefficient, the aquifer bulk density, and porosity. The distribution coefficient is treated as a physical parameter, but is also strongly dependent upon the chemistry of the ground water system. For each radionuclide, the distribution coefficient will vary with the composition and pH of the effluent and the ground water, as well as the physical and chemical properties of the aquifer material. The major effects of a distribution coefficient are to retard the movement of the associated radionuclide and to reduce its concentration in the liquid phase.

Both models are based on calculations of unidirectional convection with three-dimensional dispersion, and correct for radiological decay in a separate calculation. This procedure allows for simpler computations in the case of long decay chains. A modified form of the Bateman equation is used to calculate the concentrations of all important daughter products in a decay chain.

In the models, the wastes are assumed to be homogeneous and uniformly distributed over the disposal area. It is assumed that all wastes were emplaced at a single point in time, and that leaching and migration begin immediately after placement. Radionuclides are assumed to have been introduced directly into and dispersed uniformly vertically within the saturated porous slab. The models use an exponential leach rate of the form

$$Lr = \lambda_L Q_i e^{-(\lambda L + \lambda d)t}$$

where Lr = the instantaneous leach rate (curies/yr^1)

λ_L = leach constant (yr^{-1})

Q = initial quantity of isotope "i" in disposal area (curies)

The parameter λ_L is equal to the rate of leaching of the isotope, and λ_d is the radioactive decay constant.

The output from both the flux and concentration models require conversion before they can be interpreted as predicted concentrations in any application. The concentration model describes the effects of hydrodynamic dispersion in a uniform aquifer with no vertical infiltration, and accretion along the flow path is not considered. Hydrodynamic dispersion includes the combined effects of hydraulic mixing and molecular diffusion. Hydraulic mixing results from a variable velocity distribution of fluid flowing in a porous medium as a consequence of boundary effects and inhomogeneities of the solid matrix. Molecular diffusion which occurs simultaneously results from chemical concentration gradients within the fluid. Hydrodynamic dispersion includes both phenomena in an inseparate form; however, molecular diffusion is generally significant only at very low velocities.

Assuming complete vertical mixing, the dilution which results from infiltration can be estimated by dividing the concentrations computed by the model by the volume of actual recharge to the contaminated zone which occurs along the flow path.

If one assumes complete mixing, (not always a valid assumption) the output from the flux model can also be expressed in terms of concentrations by dividing the annual contaminant flux by the annual water flux in the receiving body of surface water or the actual volume of water flowing through the aquifer. The result of this computation is the average concentration over the flow cross-section.

The two models were utilized to compute radionuclide flux and concentrations down-gradient of the site at the location shown in Figure 7. Model input utilized in these analyses is presented in Tables 3 and 4. Table 3 presents isotope data, and Table 4 presents hydrologic data utilized for the baseline analysis.

The values of the hydrologic parameters were obtained from various tests and assumptions. These are described as follows:

- The ground water velocity was determined by borehole dilution tests performed on site in the "E" stratum. An average velocity from two tests of 5.5 feet per day was used.
- An aquifer thickness of 12.9 feet was determined from averaging the thickness of the "E" stratum in the site borings.

TABLE 3

ISOTOPE DATA FOR CONTAMINANT TRANSPORT ANALYSIS

Isotope	t (years)	activity of isotope in waste, Ci	activity/m^3 of waste, Ci/m^3	Distribution Coefficient* ml/g	Leach Rates[1]
Th 232	1.41×10^{10}	356	1.90×10^{-3}	3082	1.0×10^{-5}
Th 228	1.910	364	1.95×10^{-3}	3082	1.0×10^{-5}
U 238	4.51×10^{9}	8.4	4.49×10^{-5}	6.6	1.0×10^{-5}
U 234	2.47×10^{5}	8.2	4.39×10^{-5}	6.6	1.0×10^{-5}
Th 230	8.0×10^{4}	37.7	2.02×10^{-4}	3082	1.0×10^{-5}
Ra 226	1602	12.4	6.63×10^{-5}	249	1.0×10^{-5}

*determined from laboratory measurements

[1]from NUREG/CR-0680, July, 1979

TABLE 4

HYDROLOGIC PARAMETERS USED IN FLUX AND CONCENTRATION COMPUTER PROGRAMS

Parameter	Value
ground water velocity	5.5 ft/day
aquifer thickness	12.9 ft
longitudinal dispersion coefficient	20 cm
transverse dispersion coefficient	20 cm
aquifer bulk density	1.65 gm/cm^3
total aquifer porosity	40%
aquifer effective porosity	35%
length of cell along x-axis	1400 ft
width of cell (⊥ to x-axis)	1160 ft
distance along x-axis from center of cell to property boundary	900 ft.
distance from x-axis of down gradient point	0 ft.

- The value of bulk density for the aquifer was assumed as 1.65 gm/cm^3, which is a representative value for sand.

- Total aquifer porosity was assumed to be 40 percent; effective porosity was assumed to be 35 percent. These are typical values for sand.

- The x-axis is defined in both computer programs as along the direction of ground water flow. Figure 7 shows the x-axis in relation to the cell. The length of the cell along the x-axis is approximately 1400 feet. The width of the cell perpendicular to the x-axis is approximately 1160 feet.

- The longitudinal and transverse dispersivity coefficients were calculated to be 2 cm using procedures described in Lenda and Zuber (1970).

The results of the computer calculations for the flux model are presented in Figure 8; the results of the calculations performed by the concentration program are presented in Figure 9. Computed concentrations of Ra 224 were essentially zero for this case. The results indicate that the concentrations of the radionuclides in the ground water will not exceed USPHS-USEPA maximum permissible concentrations shown in Table 5 (Nuclear Regulatory Commission, NUREG-0511, April 1979).

In order to test the sensitivity of the models, various input parameters were changed and additional analyses were made. The values of the longitudinal and transverse dispersivity coefficients were changed from 2 cm to 20 cm. These changes result in very slightly lower values of isotope concentrations. The distribution coefficients for each isotope then were set equal to zero. The peak concentration values for this case were essentially the same as for the baseline case for each isotope, however, peak concentrations occur sooner than with non-zero distribution coefficients.

TABLE 5

MAXIMUM PERMISSIBLE CONCENTRATIONS
OF SELECTED RADIOISOTOPES*

Parameters	USPHS - USEPA Maximum Permissible Concentrations
U-nat	550 pCi/l
Ra-226	5 pCi/l
Th-230	2,000 pCi/l

*from Generic Environmental Impact Statement in Uranium Milling, NUREG-0511, Volume II, Appendices, April, 1979

The maximum computed concentrations of Ra 224 for this case, considering the isotope as a parent nuclide, was calculated as 1.3×10^{-14} pCi/l at 0.5 year. The maximum computed concentrations, considering the isotope as a daughter in the Th 232 decay chain, was 8.81 pCi/l and occurred at 1 year.

The results of these sensitivity analyses indicate that maximum radionuclide concentrations in the shallow water-bearing stratum at the site boundary are not sensitive to reasonably-expected variations in dispersion coefficients, ground water velocities, or distribution coefficients. However, the time at which maximum concentrations first occur at the site boundary is strongly influenced by the distribution coefficients, and is somewhat influenced by ground water velocities.

The predicted concentrations in the ground water of the radionuclides at this site are sensitive to the assumed quantities and rate of leaching of the radionuclides; predicted concentrations vary approximately linearly with both the leach rate parameter and the quantity of the radionuclide in the disposal cell. The values of these parameters used in the above-described analyses are believed to be conservative in the sense that they yield computed concentrations larger than should be expected actually to occur.

The degree of conservatism in the leach rates may be assessed by a comparison of the calculated concentrations in leachate using the leach parameters with concentrations determined by the RCRA EP test procedures. The RCRA procedures utilize an acidic leaching solution, and therefore contaminant concentrations should be somewhat higher in this test than in leachate from the disposal cell where the wastes will be neutralized prior to disposal.

Average leachate concentrations for radioactive contaminants computed using a value of 10^{-5} per year and a percolation rate of 0.72 inches per year are presented in Table 6. Also shown in this table are concentrations of the same contaminants computed as a weighted average leachate concentration from Table 1 describing the sludge and tailings piles. In all cases, the computed concentrations exceed the concentrations measured in the EP test, showing the conservative nature of the model analysis.

The results of the radionuclide concentration analyses may also be applied to an assessment of the concentrations of non-radioactive substances in the shallow ground water by assuming that the concentrations of these substances measured in the leachate tests are equivalent to the concentrations in the actual cell leachate. The concentrations obtained from the analyses for the case where distribution coefficients were assumed equal to zero then represent the mixing capacity of the ground water in the shallow sand for an assumed constant rate of contaminant introduction. Applying the computed average value of percolation through the cell of 0.72 inches per year, the rate of contaminant introduction can be interpreted as the introduction of that volume of leachate at the concentrations given in the third column

TABLE 6

COMPARISON OF COMPUTED AND MEASURED CONTAMINANT CONCENTRATIONS

ISOTOPE	ACTIVITY IN WASTE (Ci)	COMPUTED CONCENTRATION IN LEACHATE (pCi/l)	MEASURED CONCENTRATION IN LEACHATE (pCi/l)	Column A/B
		A	B Avg.	
Th 232	356	2.8×10^3	281	10
Th 228	364	2.9×10^3	1852	1.6
Th 230	37.7	3.0×10^2	46	6.5
Ra 226	12.4	9.8×10^1	7	14

of Table 6. If these concentrations are divided into the computed ground water concentrations at the site boundary for those isotopes with long half-lives, the result is the amount of dilution from the cell to the site boundary at the assumed rate of leaching, which is predicted by the model for a continuous injection of contaminant. Because of the small values of dispersion coefficients used in this analysis, and because no account is taken of chemical reactions which might reduce the concentration of the contaminant, the mixing factor computed by the above method is essentially the ratio of the annual volume of percolation through the cell to the annual volume of water which flows beneath the cell through the shallow sand.

The mixing factor obtained by this method is 227; that is, the concentration at the site boundary of a non-reactive substance derived from the waste would be predicted by this model to equal the concentration of the substance in the cell leachate divided by 227.

Table 7 shows the concentrations of selected ionic species in the ground water at the site boundary as determined by the analysis described above. The average leachate concentrations were computed as the weighted average of the leachate results presented in Table 1. Also shown in Table 7 are USPHS-USEPA recommended maximum permissible concentrations for public drinking water supplies for species for which limits have been established. The drinking water standards in every case are greater than the model predicted concentrations, in many cases by more

TABLE 7

ESTIMATED WATER QUALITY AT SITE BOUNDARY

(all values in mg/l)

Predicted Concentration

Parameter	Weighted Average Concentration Piles 1 + 2	Average Concentration x Mixing Factor	USPHS-USEPA MAXIMUM PERMISSIBLE CONCENTRATION	RATIO MPC/Predicted Concentration
Ag	0.21	9.24×10^{-4}	0.05[2]	54
As	0.0034	1.50×10^{-5}	0.05[2]	3333
Ba	0.073	3.21×10^{-4}	1[2]	3115
Cd	0.034	1.50×10^{-4}	0.01[2]	66.7
Cu	0.106	4.66×10^{-4}	1[1]	2146
Cr	0.0086	3.78×10^{-5}	0.05[2]	1323
Fe	8.5	3.74×10^{-2}	0.3[1]	8.0
Hg	<.001	$<4.4 \times 10^{-6}$	0.002[2]	454
Ni	0.289	1.27×10^{-3}	-	-
Pb	0.98	4.31×10^{-3}	0.05[1]	11.6
Se	.007	3.08×10^{-5}	0.01[2]	325
Zn	0.86	3.78×10^{-3}	5[1]	1323
Ca	258	1.14	-	-
K	11.2	4.93×10^{-2}	-	-
Mg	27.0	1.19×10^{-1}	-	-
Na	138	6.07×10^{-1}	-	-
SO_4	1507	6.63	250[1]	37.7
Cl	<4.4	$<1.94 \times 10^{-2}$	250[1]	>12,887
F	11.3	4.97×10^{-2}	1.4-2.4[2]	28-48
NO_3	0.22	9.68×10^{-4}	10[2]	10,142

[1]from National Secondary Drinking Water Regulations, 1979

[2]from National Interim Primary Drinking Water Regulations, 1975

than a factor of 10. Continued contaminant migration with ground water flow to the first point of discharge at Kress Creek would show insignificant impact on water quality at this location.

The results of the above calculations represent estimates of concentrations of the various species which would be derived from the disposal site. These species concentrations are additive to the same species naturally occurring in the ground water. The significance of the computed concentrations, both for the radioactive and non-radioactive constituents lies not in the actual value of the numerical estimates, but rather in the smallness of these values relative to the natural concentrations and published standards used to judge the quality of drinking water supplies. The shallow sand beneath the site is not known to be a source of drinking water in the site area. The drinking water standards were utilized only to illustrate the relative significance of the computed concentrations from the model.

The computations described above were deliberately made in what is believed to be a conservative manner; that is, the results of the computations should overestimate the proposed disposal plan effects on future water quality. On the basis of the concentrations in the shallow ground water at the site boundary predicted to result from the proposed disposal, and the conservative nature of the calculations used to predict these concentrations, it is concluded that the proposed disposal plan will have no significant adverse impact upon the shallow ground water system flowing beneath the site.

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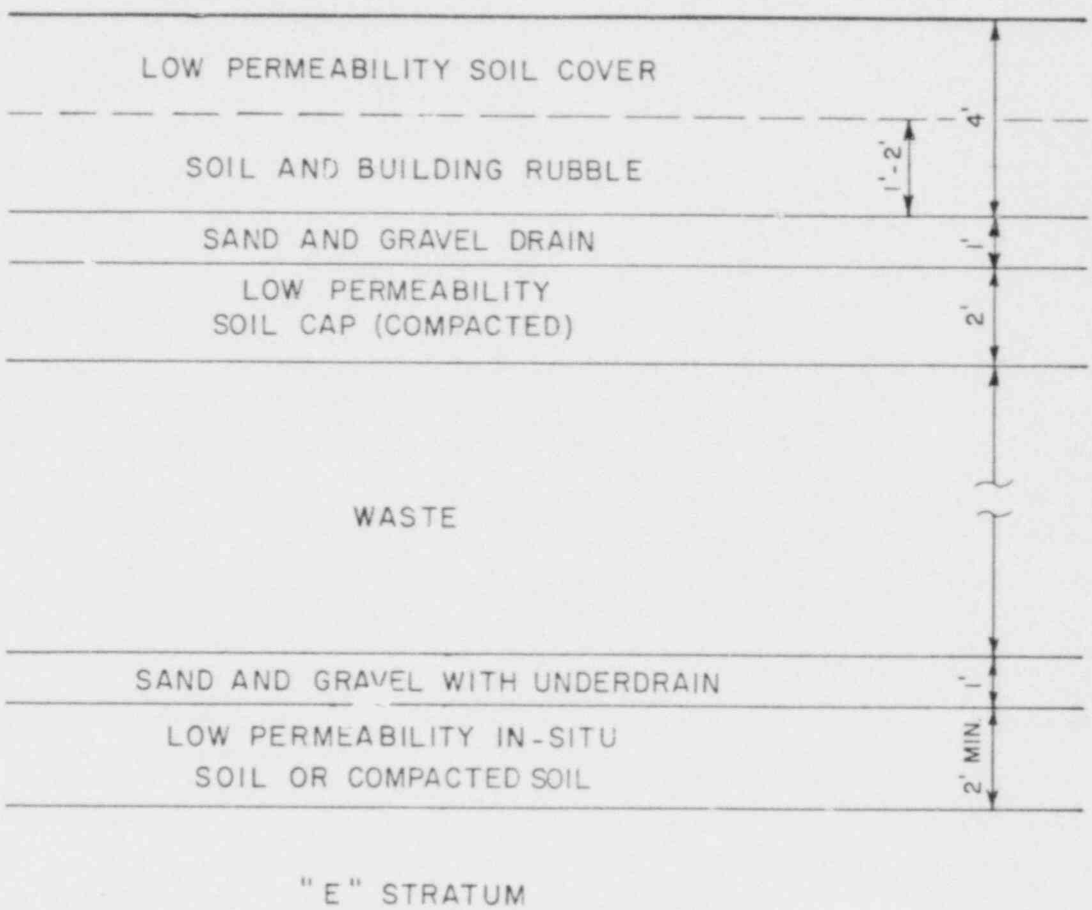

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FIGURE 3

SCHEMATIC SECTION OF WASTE DISPOSAL CELL

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FIGURE 5
COMPUTED AVERAGE NATURAL PERCOLATION

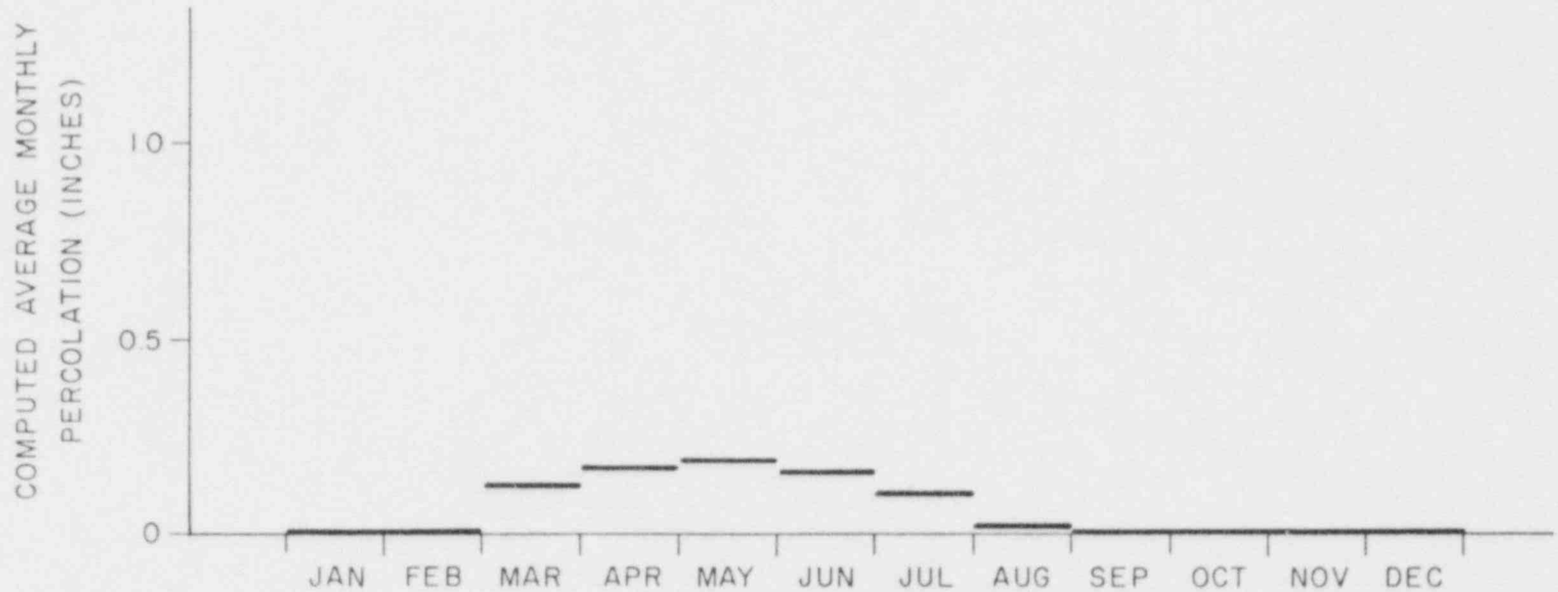

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FIGURE 6
COMPUTED AVERAGE CELL COVER
PERCOLATION

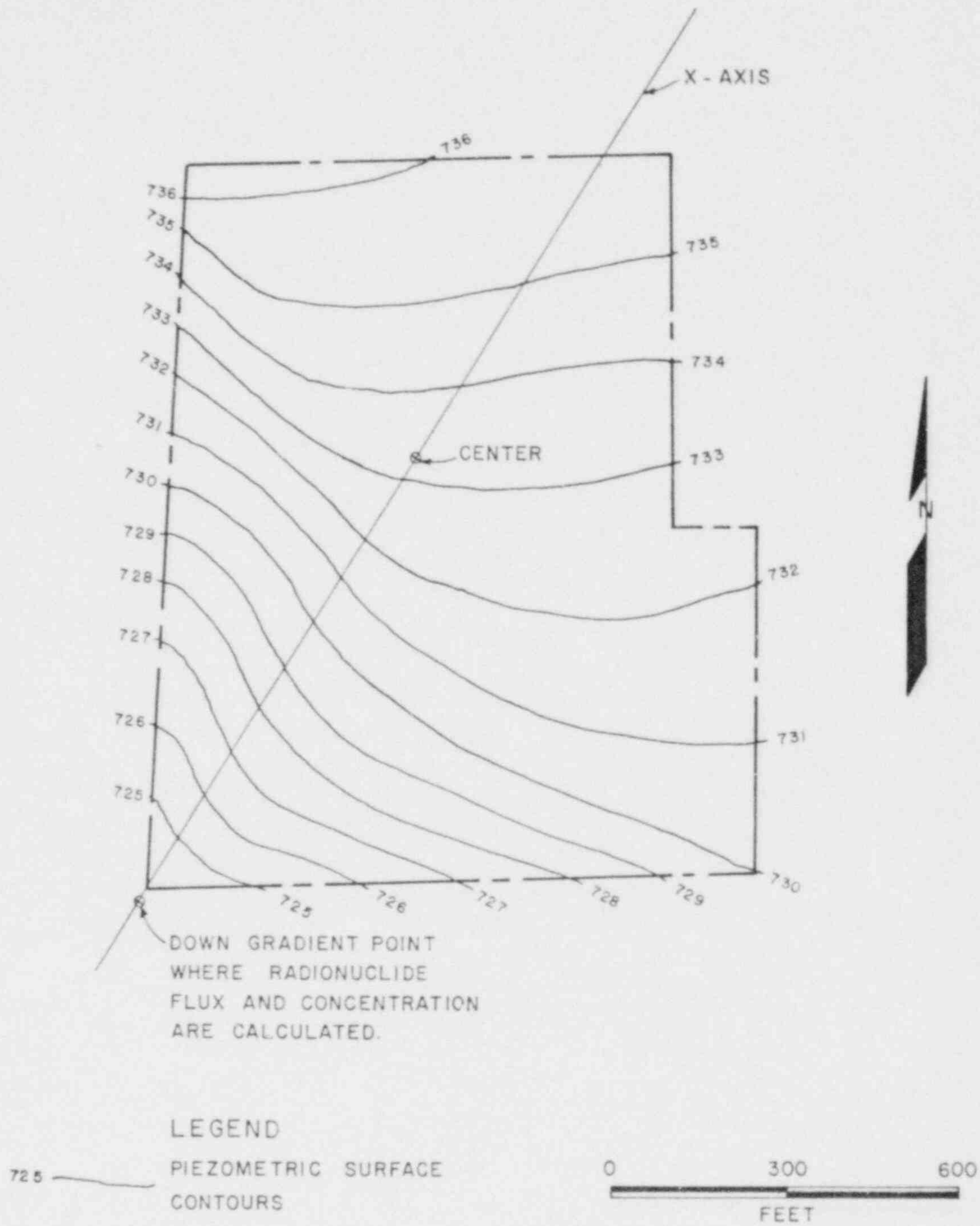

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FIGURE 7
LOCATION OF X-AXIS FOR COMPUTER PROGRAM

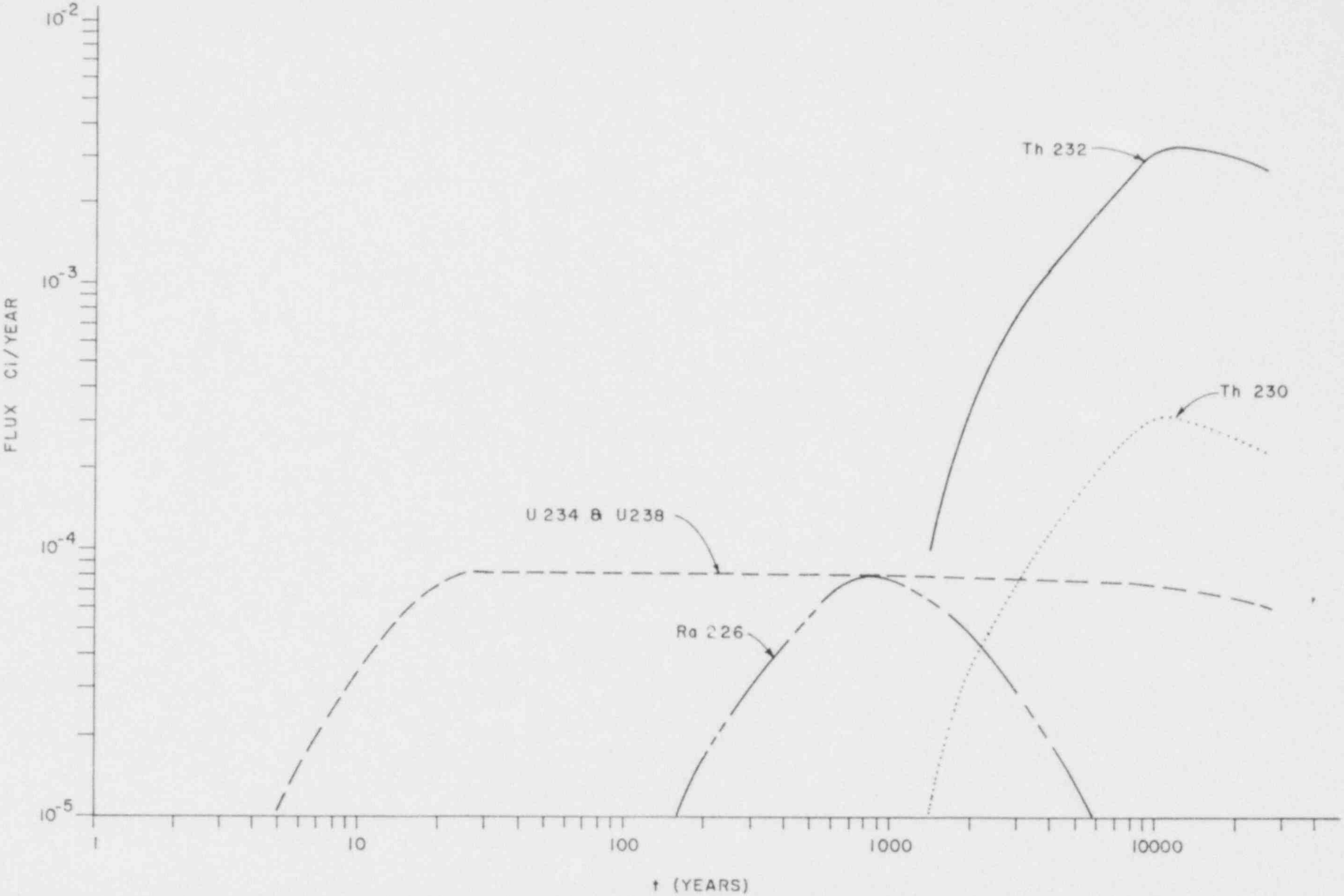

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FIGURE 8
FLUX OF RADIOISOTOPES DOWN GRADIENT FROM DISPOSAL CELL

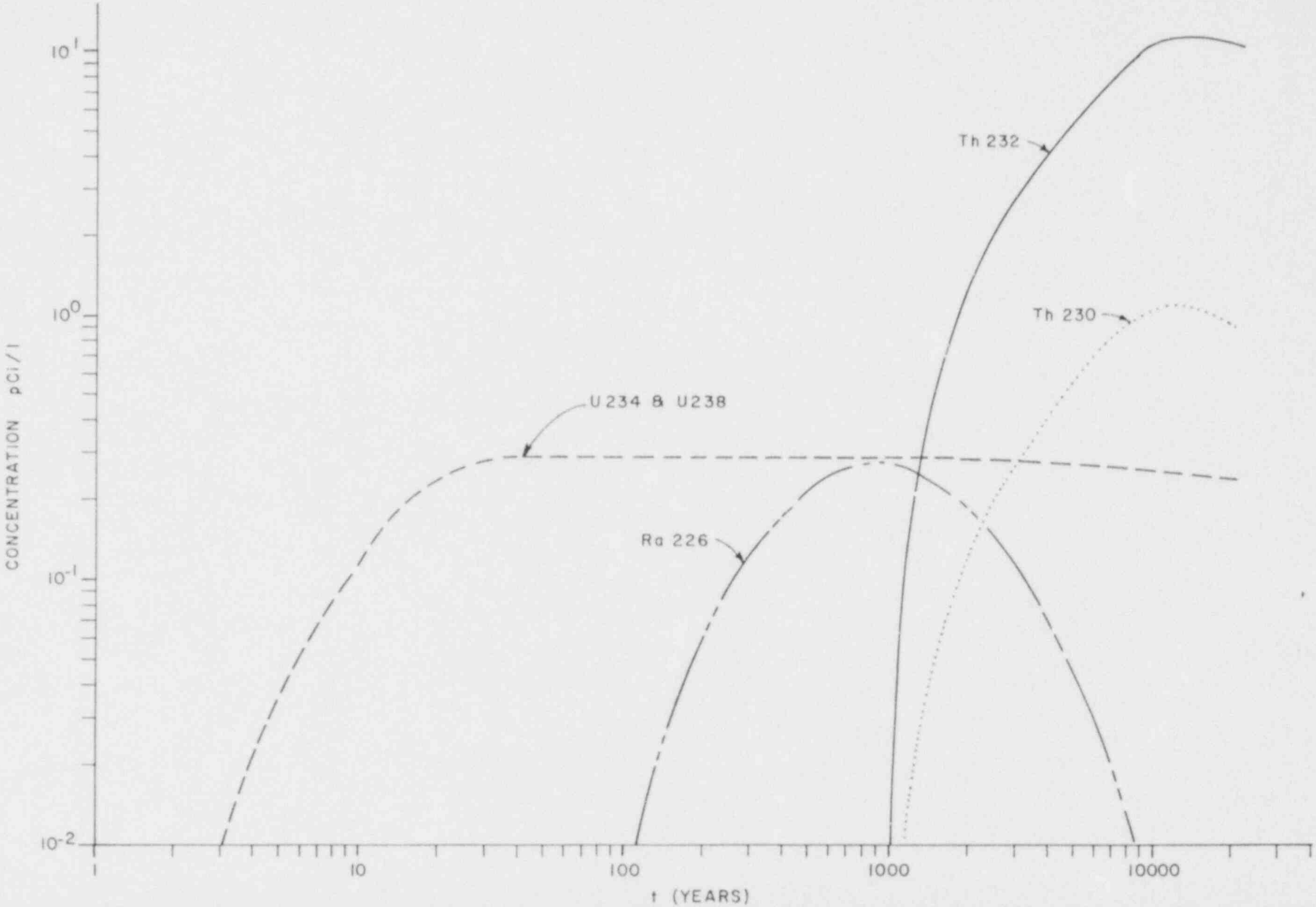

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FIGURE 9
CONCENTRATION OF RADIOISOTOPES DOWN GRADIENT FROM DISPOSAL CELL