

Published in the 1979 Transactions of the American
Nuclear Society, Vol. 33, pp. 386-387

UC-70
LBL-10345 c.2
Preprint

MIGRATION BEHAVIOR OF THE
 $^{234}\text{U} \rightarrow ^{230}\text{Th} \rightarrow ^{226}\text{Ra} \rightarrow$ DECAY CHAIN

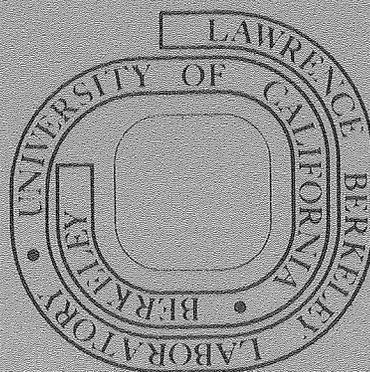
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November 1979

Prepared for the U.S. Department of Energy
under Contract W-7405-ENG-48

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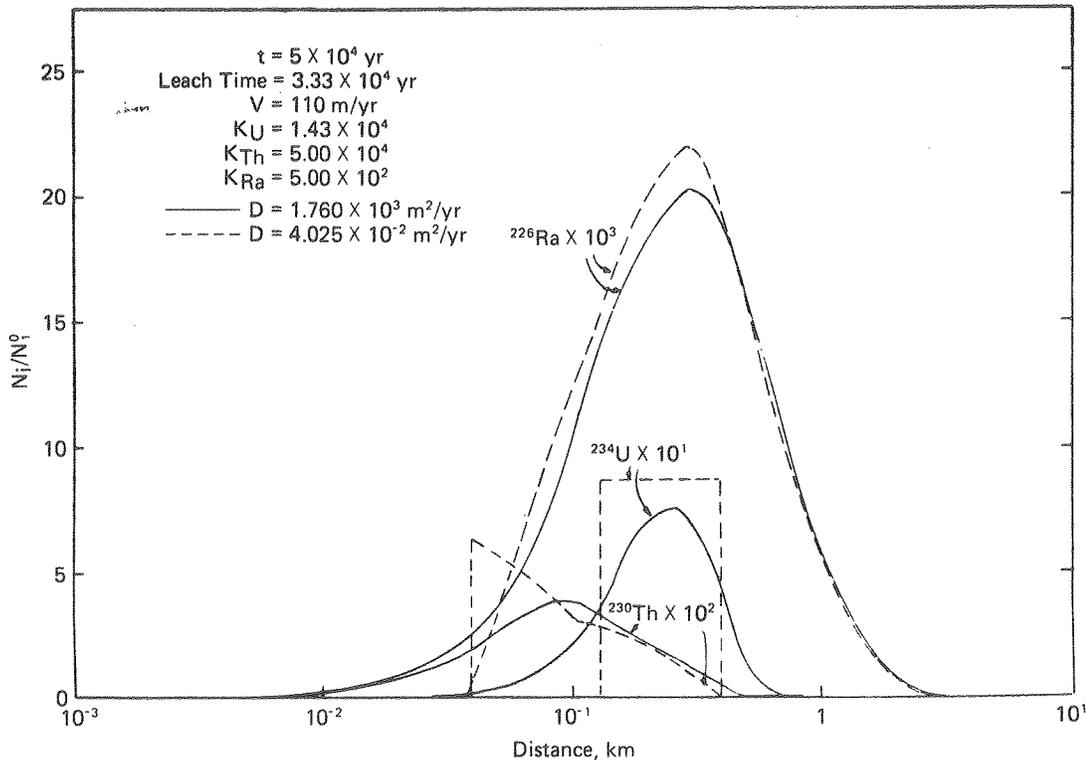
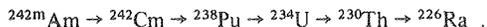


Fig. 1. Migration pattern of $^{234}\text{U} \rightarrow ^{230}\text{Th} \rightarrow ^{226}\text{Ra}$ decay chain (pure ^{234}U source at $t = 0$).

3. M. FOGLIA, T. H. PIGFORD, and P. L. CHAMBRÉ, "The Superposition Equation for the Band Release of Decaying Radionuclides Through Sorbing Media," UCB-NE-3935 (Mar. 1979).
4. M. HARADA, F. IWAMOTO, and T. H. PIGFORD, "Effect of Source Boundary Conditions in Predicting the Migration of Radionuclides Through Geologic Media," UCB-NE-3349 (June 1979).
5. K. HIGASHI and T. H. PIGFORD, "General Solutions for the Migration of Radionuclides Through Sorbing Media Without Dispersion," UCB-NE-3351 (Mar. 1979).

7. Migration Behavior of the $^{234}\text{U} \rightarrow ^{230}\text{Th} \rightarrow ^{226}\text{Ra}$ Decay Chain, K. Higashi,* M. Harada, F. Iwamoto, T. H. Pigford (UC-Berkeley)

One of the important nuclide chains to be considered in the predictive modeling of the hydrogeological transport of radionuclides in high-level wastes is:



The most important species is usually ^{226}Ra , because of its relatively high biological hazard and its relatively high mobility in geologic media. The important time scales for the appearance of ^{226}Ra are on the order of tens of thousands of years, as controlled by the long life of its precursor ^{230}Th . Therefore, because of the relatively short lives of the first three members of the above chain, the analysis of ^{226}Ra transport can be reduced to the analysis of the last three members of the above chain, with the assumption that the first three members have already decayed to form the

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long-lived ^{234}U . Analytical studies of the transport of this decay chain have been reported elsewhere.¹⁻³ In the present paper we analyze the loci of maximum concentrations of ^{226}Ra and we analyze the effect of dispersion on the concentration profile of ^{226}Ra .

The radionuclide transport equation to be solved is the one-dimensional (1-D) equation which assumes constant water velocity and local sorption equilibrium.¹⁻³ Figure 1 shows the isopleths of radium concentration, calculated for the dispersion-free case³ with assumed leaching of the waste matrix material at a constant rate over a leach time T . Other assumed parameters are indicated on the figure. The isopleths indicate that the peak time-dependent radium concentration at any given distance from the source increases with source distance until a maximum concentration is reached, after which it decreases monotonically with distance. This is a result of (a) the slow growth with time of ^{226}Ra in the waste material, and (b) the reconcentration effect due to the distribution of sorbed ^{230}Th through the media, as has been discussed elsewhere.^{3,4}

When thorium sorption (K_2) is greater than uranium sorption (K_1), the location of the maximum of the radium concentration at any time t is near the front of the uranium chromatographic band, as shown by the line $z = K_1 t/v$ in Fig. 1.

Figure 2 shows the effects of dispersion and of the overall radium sorption constant on the concentration profile of radium, at a given instant in time and as a function of distance from the source. The lower value of the dispersion coefficient gives results equivalent to a dispersion-free transport analysis. The curves indicate that dispersion has relatively little effect on the concentration profile of radium in this time period, unless the radium sorption is much greater

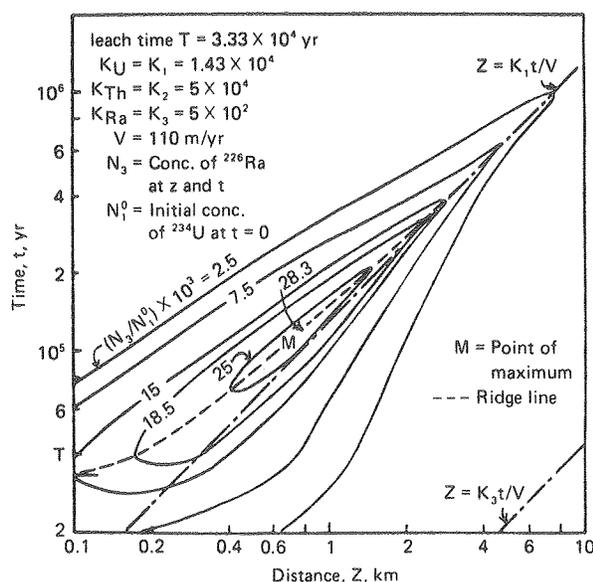


Fig. 1. Isopleths of ^{226}Ra (no dispersion, pure ^{234}U source at $t = 0$, K = overall sorption equilibrium constant, V = water velocity, 1 = ^{234}U , 2 = ^{230}Th , 3 = ^{226}Ra).

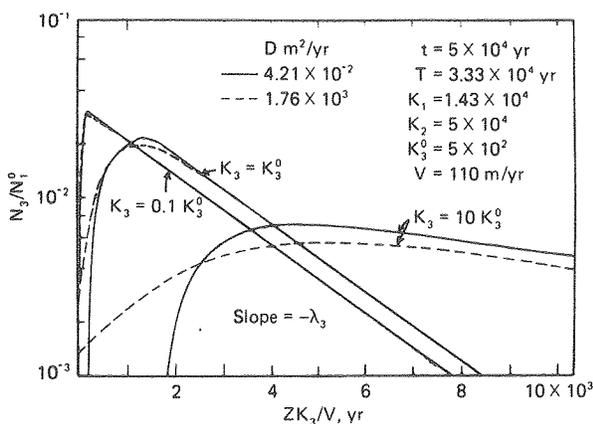


Fig. 2. Concentration profiles of ^{226}Ra at 5×10^4 yr (pure U source at $t = 0$, K = overall sorption equilibrium constant, V = water velocity, 1 = ^{234}U , 2 = ^{230}Th , 3 = ^{226}Ra).

than the values normally used for this calculation the radium profile is not strongly affected by uncertainties in this sorption coefficient. This is physically reasonable, since in the time scales of interest here the radium's half-life is too short for it to proceed in appreciable concentration much ahead of the chromatographic bands of its precursors. Therefore, the rate of transport of Ra and the magnitude of its concentration in the chromatographic band are controlled more by the transport properties of the radium precursors.

1. H. C. BURKHOLDER, M. O. CLONINGER, D. A. BAKER, and G. JANSEN, "Incentives for Partitioning High-Level Waste," *Nucl. Technol.*, **31**, 202 (1976).
2. M. HARADA, F. IWAMOTO, and T. H. PIGFORD, "Effect of Source Boundary Conditions in Predicting the Migration of Radionuclides Through Geologic Media," UCB-NE-3349 (June 1979).

3. K. HIGASHI and T. H. PIGFORD, "General Solutions of the Migration of Radionuclides Through Sorbing Media Without Dispersion," UCB-NE-3351 (Mar. 1979).

4. H. C. BURKHOLDER and M. O. CLONINGER, "The Reconcentration Phenomenon of Radionuclide Chain Migration," BNWL-SA-5786 (1978).

8. Polymeric Solidification of Low-Level Radioactive Waste, Harold E. Filter (Dow-Mich)

The Dow waste solidification process forms stable mixtures of the waste and binder which are then chemically cured to form hard, solid monoliths. Liquid or slurry waste is stirred with a commercially available modified vinyl ester resin until a stable mixture is formed. The mixture then is cured by the addition of a catalyst and a promoter. The final result is a homogeneous solid which contains no free liquid. The Dow definition of free liquid is liquid on the product or in the container on completion of the solidification, or liquid that exudes from the product after solidification.

The Dow process can be used to solidify the various low-level radioactive wastes normally resulting from the operation and/or decontamination of nuclear power plants. The process works equally well for radioactive wastes containing dispersed solids such as ion-exchange resins or filter-aid materials. The process uses readily available chemicals that are easily handled in a safe manner.

The process is essentially the same for small laboratory mixes or for full-scale 55-gal drum and 50-ft³ linear mixes. The mixing is done in a cylindrical container, which also serves as the burial container for the solidified waste.

The solidification process is relatively simple and when applied with proper equipment produces a uniform product that meets all existing regulations and standards.

Laboratory and field studies^{1,2} have shown that the Dow process produces a product with properties required for solidified low-level radioactive waste disposal. The major properties are as follows:

1. liquid-free product.
2. immobilizes radioisotopes homogeneously, resulting in no "hot spots"
3. minimizes leaching of radioisotopes to the environment
4. resistant to combustibility at high temperatures
5. resistant to radiation
6. high strength and impact resistance.

Previous data with nonradioactive simulated wastes¹ solidified with the Dow process have shown that the solid product passes DOT and International Atomic Energy Agency (IAEA) heat exposure test^{3,4} standards. Samples of solidified simulated wastes did not melt, sublime, or ignite when exposed at 1000°F for 10 min. The test has been expanded to include radioactive simulated wastes. The solidified specimens passed the test standards and exhibited weight losses consistent with previous data on nonradioactive samples. None of the samples lost any detectable amount of radioactivity during the test.

The effect of the heat exposure on leaching was also investigated. Deionized-water leach tests were run on the samples exposed in the muffle furnace and unexposed controls. The solidified specimens exhibited long-term leach rates consistent with previous data. No detrimental effects on the long-term leaching of radioisotopes from the solidified specimens were caused by the exposure to heat.