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**A MODELING STUDY OF CONTAMINANT TRANSPORT RESULTING FROM
FLOODING OF PIT 9 AT THE RADIOACTIVE WASTE MANAGEMENT COMPLEX,
IDAHO NATIONAL ENGINEERING LABORATORY**

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ABSTRACT

A simulation study was conducted to determine if dissolved-phase transport due to flooding is a viable mechanism for explaining the presence of radionuclides in sedimentary interbeds below the Radioactive Waste Management Complex. In particular, the study focused on ^{241}Am migration due to flooding of Pit 9 in 1969. A kinetically-controlled source term model was used to estimate the mass of ^{241}Am that leached as a function of a variable surface infiltration rate. This mass release rate was then used in a numerical simulation of unsaturated flow and transport to estimate the advance due to flooding of the ^{241}Am front down towards the 110' interbed. The simulation included the effect of fractures by superimposing them onto elements that represented the basalt matrix.

For the base case, hydraulic and transport parameters were assigned using the best available data. The advance of the ^{241}Am front due to flooding for this case was minimal, on the order of a few meters. This was due to the strong tendency for ^{241}Am to sorb onto both basalts and sediments.

In addition to the base case simulation, a parametric sensitivity study was conducted which tested the effect of sorption in the fractures, in the kinetic source term, and in the basalt matrix. Of these, the only case which resulted in significant transport was when there was no sorption in the basalt matrix. The indication being that other processes such as transport by radiocolloids or organic complexation may have contributed. However, caution is advised in interpreting these results due to approximations in the numerical method that was used incorporate fractures into the simulation. The approximations are a result of fracture apertures being significantly smaller than the elements over which they are superimposed.

The sensitivity of the ^{241}Am advance to the assumed hydraulic conductivity for the fractures was also tested. Although the advance was greater when the conductivity of the fractures was increased, it did not approach that of the non-sorbing case. The overall conclusion was that it appears that dissolved phase transport by itself is not a feasible mechanism to explain the presence of radionuclides in the 110' interbed.

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1.0 Introduction

1.1 Background

The Idaho National Engineering Laboratory (INEL) has historically been a site of nuclear reactor and waste management research. More recently, technical programs have been focused on assessing the impact of past disposal practices and developing site clean-up plans. As part of these environmental restoration activities, technical evaluations are being conducted for the low-level radioactive, hazardous, and mixed-waste sites such as the Radioactive Waste Management Complex (RWMC). These evaluations contribute to: (1) the Remedial Investigation/Feasibility Study (RI/FS) process and (2) compliance studies required by the U. S. Department of Energy (DOE) Order 5820.2A (Performance Assessments).

Performance assessments require analyzing contaminant movement from waste sites through the atmospheric, biotic, and water pathways. At the RWMC, the water pathway is the most significant (Case et al., 1990). Because of the limited number of natural, surface-water features at the RWMC, the ongoing performance assessment for the Low Level Waste Facility at the RWMC has focused on the issue of subsurface contaminant movement in the vadose zone and in the Snake River Plain Aquifer. Specific goals of this performance assessment are:

- o understanding the role of physical and chemical transport mechanisms,
- o assessing the rate, direction and extent of contaminant movement, and
- o predicting contaminant concentrations and evaluating compliance with prescribed limits.

1.2 Purpose

The ongoing performance assessment for the Low Level Waste Facility at the RWMC has, so far, only addressed contaminant migration resulting from yearly-averaged infiltration of meteoric water. The basis for this approach was justified by results from simulation studies (Baca et al., 1992) that showed significant damping of infiltration events with depth. However, this approach resulted in low migration rates of contaminants that cannot, for instance, explain the presence of actinides in sedimentary interbeds at depth (Laney et al., 1988). One hypothesis previously proposed (Schmalz, 1972; Rawson et al. 1991) to explain the actinides in the interbeds was transport associated with past occurrences of flooding of open pits. The primary mechanism of contaminant transport during this flooding was subject to debate. Possibilities included dissolved-phase, colloidal phase, and/or organically-complexed transport. This study addresses only dissolved-phase transport as a possible mechanism. In particular, this simulation study focused on the 1969 flooding of Pit 9 and possible resulting transport of ^{241}Am in the subsurface.

The intent of this study was to numerically simulate the flooding of Pit 9 and the concomitant release of actinides into the subsurface. This study was

similar to work by Walton et al. (1989) except in the current study the fractures and the basalt were represented separately, rather than lumping their hydraulic properties together. By comparing the simulation results for transport to the sedimentary interbeds to measured radionuclide concentrations in the interbed, a determination could be made whether dissolved-phase transport during flooding was a feasible mechanism to explain the presence of actinides in the interbed.

This study will also "set the stage" for future flooding studies of the RWMC as may be required in the performance assessment process. The process by which fractures are incorporated in numerical models of subsurface flow and transport will be initiated and hopefully improved in future studies.

2.0 Conceptual Model

This section will discuss the conceptual model used in the simulation study in terms of the physical setting, the processes affecting flow and transport that were simulated, the mechanisms governing the release of contaminants from the buried waste, and the hydraulic and transport parameters used in the simulation.

2.1 Physical Setting

The RWMC is located in the south-eastern portion of the INEL (Figure 1). Pit 9, a shallow waste disposal pit, is located in the north-eastern corner of the SDA. The pit was open for waste emplacement from November, 1967 to June, 1969 (Vigil, 1989). The pit had approximate dimensions of 100 m by 40 m and a depth of approximately 2.5 m with 1.0 m of underburden soil. Figure 2 depicts a vertical cut-away view which represented the physical setting of Pit 9 in relation to the subsurface. The subsurface representation consisted of surface sediments, vesicular and massive basalts, and a sedimentary interbed at 110 ft below the surface. The thicknesses of each unit were the same as those used by Baca et al. (1992). For the simulations, each of these units were assumed to be horizontal and of uniform thickness. This is not entirely accurate but was justifiable for the purposes of this study.

2.2 Transport Processes

Several different mechanisms by which wastes have migrated from the pits are hypothesized (see Figure 3). The majority of these mechanisms are related to infiltrating water. The infiltrating water resulted from either precipitation or snowmelt at the surface or in rare instances, flooding. Both sources of infiltrating water were included in this study.

Once water infiltrated into the waste, contaminants either dissolved into the water, formed radio-colloids, and/or complexed with organics that then migrated with the water. Only dissolved-phase transport was considered in this conceptual model.

Once the solutes were released from the pit or source-term model, any number of processes could affect transport as depicted in Figure 3. The processes considered in this study were diffusion, dispersion, advection, sorption and radioactive decay.

2.3 Source Zone

The source zone consisted of the pit along with the intermixed and immediate surrounding surface sediments. The mechanism controlling nuclide release within the source zone was kinetic leaching which was allowed to respond to variable water infiltration rates, such as flooding. This release mechanism was based on a model developed by John Walton in Bargelt et al. (1992) and is explained fully in Section 3 below.

2.4 Flow

Water movement in the vadose zone is in response to gravitational and capillary forces and can be described by Richard's equation (Hillel, 1971) which is given by

$$\frac{\partial \theta}{\partial t} = - \frac{\partial}{\partial x} (K(\psi) \frac{\partial \psi}{\partial x}) - \frac{\partial}{\partial z} (K(\psi) (\frac{\partial \psi}{\partial z} - 1))$$

where ψ = matric potential or suction head (m),
 θ = volumetric water content,
 K = hydraulic conductivity (m/s).

In considering subsurface movement of water, there were two important factors which governed this movement. The first was the source of water at the surface. This source served as the driving force for all subsequent movement of water in the subsurface and was therefore important. Two sources of this water, background infiltration and flooding, are discussed below. The second important factor was the method by which water moved through the fractured basalt. This will be discussed in Section 2.4.3.

2.4.1 Background Infiltration

The net average infiltration rate assumed in the study was 5 cm/yr. Although somewhat arbitrary, this value was in rough agreement with other estimates either inferred from radio-isotope measurements (Cecil et al., 1992) or previous simulation studies (Baca et al., 1992). Both these studies yielded estimates on the order of 1 to 2 cm/year net infiltration. Therefore, the assumed 5 cm/yr infiltration is conservative because it results in slightly wetter conditions which would result in both greater leaching of buried wastes and faster movement due to higher hydraulic conductivities.

The 5 cm of water is assumed to infiltrate at a constant rate throughout the year. This assumption is questionable since the average precipitation on the INEL is approximately 22 cm and the potential evaporation estimated from pan evaporation measurements is 110 cm (Clawson et al., 1989). Water that does infiltrate under these conditions would likely do so during times of wetter conditions, such as during spring snowmelt. At depth, however, Baca et al. (1992) showed significant damping of variable surface infiltration which supports the assumption of the constant infiltration rate. The implication is that for flow, the constant infiltration rate may be proper. This may not be the case for transport, in which periodic infiltration rates may affect leaching from wastes buried in the near surface. The amount of infiltrating water and its time distribution are areas of ongoing study.

2.4.2 Flooding

During the spring of 1969, a local snowmelt event resulted in flooding of Pit 9 (Department of Energy, 1983). This flood was incorporated into the model as an increased infiltration rate that lasted for a period of two weeks, following the example of John Walton as reported in Bargelt et al. (1992). Vigil (1988) estimated that the volume of flood water in Pit 9 was 653 m³.

Evaporation would have a minimal affect on this flood water due to the time of year so it is assumed to have all infiltrated through the bottom of the pit. Using the estimated surface dimensions of the entire pit, this implied that 0.16 m of water infiltrated over a two-week period. This defined the infiltration rate used during the two-week flooding.

2.4.3 Water Movement in the Fractured Basalt

In a previous study, Walton et al. (1989) made the bounding assumption that water movement in the fractured basalt was controlled strictly by the fractures; the effect of the matrix was not considered. More current information (Bishop, 1991) indicates that the matrix does have some permeability, albeit small. In this study, both the matrix and the fractures were simulated and allowed to contribute to flow. The fractures were considered to be partially infilled with overlying sediment and act as a highly-conductive porous media compared to the surrounding basalt matrix. Because of a lack of site-specific information, the hydraulic properties of the fractures were assumed as was their 4 m spacing. Ongoing assessment work (Knutson et al., 1992) indicates that the assumed spacing may have been slightly too wide.

2.5 Hydraulic and Transport Properties

The hydraulic properties necessary to describe flow in an unsaturated, porous media consist of the saturated conductivity, the effective porosity, and a moisture characteristic curve. The moisture characteristic curve describes the nonlinear relationship between the pressure head or energy state of the soil water, the volumetric water content, and the relative conductivity. The commonly used van Genuchten (1980) and Mualem (1976) equations were used to relate the relative conductivity to the volumetric water content and the pressure head. They are

$$\theta(\psi) = \theta_r + (\theta_s - \theta_r) [1 + (\alpha |\psi|)^n]^{-m}$$

and

$$K_r = S_n^{\frac{1}{2}} [1 - (1 - S_n^{\frac{1}{m}})^m]^2$$

where θ_r = residual water content,
 θ_s = saturated water content,
 α = fitting parameter (1/m),
 n = fitting parameter,
 $m = 1 - 1/n$,
 K_r = relative hydraulic conductivity, and
 $S_n = (\theta - \theta_r) / (\theta_s - \theta_r)$.

The values assigned for each material type for these fitting parameters were taken from Baca et al. (1992) and were based on nonlinear curves fitted to measurements on core samples. The exception to this was the fractures, for which were assumed. Baca et al. (1992) lumped the effect of fractures into

the vesicular basalt by increasing the hydraulic conductivity, similar to Walton et al (1989). In this study, the values determined by Bishop (1991) for the vesicular basalt matrix were used and the fractures were represented separately. The values used in this simulation study are given in Table I. Figures 4 and 5 show the moisture characteristic curves and relative conductivity curves for each material type.

The transport properties required for each material type were the effective diffusivity, partition coefficient, bulk density, and longitudinal and transverse dispersivities. The diffusivities in free water for most ions is approximately $1.0 \times 10^{-5} \text{ cm}^2/\text{s}$. Tortuosity, which reduces this diffusivity was neglected in this study. Rather, the $1.0 \times 10^{-5} \text{ cm}^2/\text{s}$ value was used as the effective diffusivity. This approximation was conservative because it maximized the diffusive movement of contaminant.

The partition coefficient describes the relationship between the sorbed soil concentration and the concentration in the soil water. This relationship is assumed to be always in equilibrium except for in the immediate area of the buried waste. The ^{241}Am partition coefficients were taken from Case et al. (1990) for consistency. The partition coefficient in the fractures was assumed. The bulk densities were taken from Knutson et al. (1990), Bishop (1991), and McElroy and Hubbell (1990) for the massive basalt, the vesicular basalt, and the interbed, respectively. The dispersivities were assumed with a general rule of thumb relationship that the transverse dispersivity was one-tenth the longitudinal, except for the fractures. The values used in the simulations are given in Table II.

Table I. Hydraulic Properties for Pit 9 Simulation

Material	K_s (m/day)	θ_s	θ_r	α (1/m)	n
Vesicular Basalt	8.64×10^{-2}	0.228	0.015	3.84	1.4741
Massive Basalt	1.02×10^{-2}	0.145	0.010	3.84	1.4741
110' Interbed	6.56×10^{-2}	0.480	0.100	1.22	1.3600
Fractures	8.50×10^1	0.500	0.005	3.00	3.0000

Table II. Transport Properties for Pit 9 Simulation

Material	K_d (ml/gm)	ρ_b (gm/cm ³)	α_l (m)	α_t (m)
Vesicular Basalt	70	1.9	1.0	0.1
Massive Basalt	70	1.9	1.0	0.1
110' Interbed	700	1.5	1.0	0.1
Fractures	2	1.5	5.0	0.1

3.0 Compartmental Radionuclide Source Term Model: Kinetic Solubility Controlled Leaching

A description of the mechanisms by which radionuclides are/were released from the pits and trenches in the RWMC is needed as part of an attempt to simulate the release and transport of radionuclides that have been disposed at the SDA. Previous attempts (Walton in Bargelt et al., 1992) to describe this nuclide release have relied on equilibrium sorption-desorption reactions to describe the mechanism by which the nuclides escape from a generalized waste container to the immediate near-field environment. This previous attempt included the transient effects of flooding which increased the leach rate from the pits. At background or low infiltration rates, the equilibrium assumption is reasonable. However, during flooding events, as the amount of water infiltrating through the waste rapidly increases, the equilibrium assumption results in unrealistically high amounts of contaminants instantaneously desorbing from the disposed waste. A more realistic description would include reaction kinetics by making use of reaction rate constants. Including the reaction rate constant is the primary additional contribution of this study to the source term in this report.

In the following sections the source term model is described, the inputs to the model are explained, and results from applying the source term model to simulate ^{241}Am release from Pit 9 are presented.

3.1 Source Model Description

The complexity of the RWMC site and the disposed waste prevented building a complete model of all the physical/chemical mechanisms that controlled the release of radionuclides. Thus the definition of the mathematical model required simplifying assumptions based on identification of the mechanisms which have greatest influence on release rate (Sullivan et al, 1988; Baca and Walton, 1988).

The model presented herein consisted of a compartmental model in which the mass of radionuclides in various compartments was calculated as a function of time. The radionuclides were conceptualized as being initially disposed of in drums that were buried and intermixed with backfill. Over time the drums failed exposing the contents to infiltrating soil moisture. The radionuclides leached out of the waste and either sorbed onto the backfill soil or moved further down into the subsurface. A flowchart of the compartmental model is given in Figure 6.

Migration of radionuclides out of the pits and trenches was assumed to occur from the percolation of water through the waste system. The water infiltration rate resulted from a combination of precipitation, evapotranspiration, runoff, snowmelt, and flooding. Past flooding events may have led to individual releases of radionuclides far in excess of release rates in the absence of floods. The contribution of flooding to existing releases was important both from a release and from a transport through fractured basalt perspective (Walton et al., 1989).

Several processes were expected to have a major influence upon radionuclide migration from the waste pits and trenches. The initial barrier to migration was the container holding the waste. At the SDA, the container could be a steel drum, plywood box, cardboard box, or nothing. A steel drum may provide containment of radionuclides for a significant period of time after disposal where cardboard and wood provide little containment. Estimation methods for drum lifetime were discussed in the work by Mutch and Sutherland (1990). The rate of drum corrosion at the SDA has previously been modeled as a first-order process with an estimated rate constant from historical INEL data of $3.2 \times 10^{-9} \text{ s}^{-1}$ (Walton et al., 1989). Once the drums failed, the waste contained in the drums was considered to be available for leaching.

The previous mathematical source model included sorption and/or precipitation of radionuclides on the materials contained in the waste as well as allowance for degradation of the container. The current mathematical model included a description of the rate at which this desorption from the waste and subsequent resorption occurs. This was accomplished by specifying the reaction rate constants so as to satisfy two conditions: 1) the time for the solid and liquid phases to reach equilibrium and 2) the equilibrium partitioning of radionuclides between the waste and the water in contact with the waste. For two rate constants K_1 (sorbing) and K_2 (desorbing), the time to reach effective equilibrium was given by

$$T_e = \frac{10 \cdot \ln 2}{K_1 + K_2}$$

and the equilibrium partitioning was given by

$$K_d \rho = \frac{K_1}{K_2}$$

Solving these two equations simultaneously for K_1 and K_2 yielded

$$K_1 = K_d \rho \cdot K_2$$

and

$$K_2 = \frac{10 \cdot \ln 2}{T_e (1 + K_d \rho)}$$

Additionally, solubility limits were enforced on the dissolved radionuclide concentrations to prevent unrealistically high concentrations. Once the radionuclides are in the liquid they were allowed to sorb onto the backfill through a similar rate-limited reaction.

The previous model included multiple waste streams while the present study concentrated on disposal of TRU waste-containing drums in Pit 9. Therefore, the remaining discussion will focus on drums as a single waste stream. The drum disposal rate and radionuclide concentration within the drums could be specified as an arbitrary function of time.

In summary, the processes modeled were:

- a. The waste was enclosed in drums and emplaced in the trenches and pits.
- b. Over time the drums failed and the enclosed material was available for leaching.
- c. Water passed through the waste pits and leached out radionuclides according to first-order kinetic reactions. This leaching process could be slowed by sorption on the waste materials or backfill and/or solubility limitations.

The volume-based governing equations for the drums and for the waste outside the drums in the pits were:

$$\frac{d\Omega}{dt} = \iota - \gamma\Omega$$

and

$$\frac{d\omega}{dt} = \gamma\Omega$$

where Ω = volume of waste contained in drums (m^3)
 ι = drum input rate (m^3/s)
 γ = drum failure rate ($1/\text{s}$)
 ω = volume of waste outside the drums.

The following equations are all mass-based equations. The governing equation for the radionuclides in the drums was

$$\frac{dM_D}{dt} = \chi\iota - \gamma M_D - \lambda M_D$$

where M_D = moles of radionuclides in drums
 χ = initial concentration of waste in the drums (moles/ m^3 of drum)
 λ = rate constant for half-life of radioactive nuclide ($1/\text{s}$).

The equation for radionuclides outside the drums and available for leaching was

$$\frac{dM_w}{dt} = \gamma M_D - \lambda M_w - K_{w,L} M_w + K_{L,w} M_L$$

where M_w = moles of radionuclides outside drums
 M_L = moles of radionuclide in liquid
 $K_{w,L}$ = rate constant for desorption from waste to liquid (1/s)
 $K_{L,w}$ = rate constant for sorption from liquid to waste (1/s).

The equation describing the mass of radionuclides in the pit water was

$$\frac{dM_L}{dt} = K_{w,L} M_w - K_{L,w} M_L - \lambda M_L - K_{L,BF} M_L + K_{BF,L} M_{BF} - q_t C_{pit}$$

where M_{BF} = moles of radionuclide sorbed in the backfill
 $K_{L,BF}$ = rate constant for sorption from the liquid to the backfill (1/s)
 $K_{BF,L}$ = rate constant for desorption from the backfill to the liquid (1/s)
 q_t = water infiltration rate (m/s)
 C_{pit} = concentration in pit liquid (moles/m³).

The last mass balance equation is for the radionuclides sorbed onto the backfill and was given by

$$\frac{dM_{BF}}{dt} = K_{L,BF} M_L - K_{BF,L} M_{BF} - \lambda M_{BF}$$

The equation describing the radionuclide concentration in the liquid in contact with the waste was

$$C_{pit} = \frac{M_L}{t A \theta_w}$$

where t = thickness of the disposed drums and intermixed backfill
 A = surface area of the pit
 θ_w = volumetric water content.

The volumetric water content, θ_w , in the backfill sediment was calculated as a function of the infiltration rate, q_t , using the same moisture characteristic curves as were used for the flooding simulation.

The instantaneous mass flux, R , released from the source term model was given by

$$\frac{dR}{dt} = q_t A C_{pit}$$

This quantity was then input as a source into the unsaturated zone flow and transport simulation for the larger domain.

The instantaneous water released from the pits was given by

$$\frac{dQ}{dt} = q_t A$$

This quantity was effectively supplied to the source term model and was the same as the upper flow boundary condition for the vadose zone flow simulation. It was calculated in order to supply an internal mass balance to ensure correct input to the source term model.

3.2 Source Term Model Input Descriptions

Input parameters to the source term model include the geometry of the waste pit, rate constants for the kinetic reactions of the waste leaching process, the solubility limit for the dissolved radionuclide, the transient water infiltration history, and a moisture characteristic curve for the backfill/waste. Other information concerning the disposal history and concentration of the waste was necessary for determining inventory and emplacement rates

Pit 9 was assumed to have an approximate surface area of 4000 m² and a depth of 2.5 meters. Of this depth, 1.0 m was backfill which yielded 6500 m³ of emplaced waste. The pit was open from 11-8-67 to 6-9-69 (Vigil, 1989) or approximately 1.6 years. Averaging the waste emplacement over the entire time yielded a waste emplacement rate of 1.29 x 10⁻⁴ m³/s. As previously mentioned, the drums were estimated to fail at a rate of 3.2 x 10⁻⁹ /s. An estimate of the quantity of ²⁴¹Am contained in Pit 9 was 619 g or 2.57 moles (King, 1991). This quantity was averaged over the entire volume of disposed waste to yield an average ²⁴¹Am concentration of 3.95 x 10⁻⁴ moles/m³. The half-life of ²⁴¹Am is 433 years which resulted in a decay constant of 4.8 x 10⁻¹¹ /s.

The moisture characteristic curve for the waste/backfill was taken from the characteristic curve for the 110' interbed sediments described earlier. The assumption was that the backfill was mixed with the waste during or shortly after emplacement.

The background infiltration rate was assumed to be 5 cm/yr or 1.59 x 10⁻⁹ m/s. During the 1969 flooding of the SDA approximately 653 m³ of water went into Pit 9 (Vigil, 1988). Evaporation would be minimal during early spring, the

time of the flooding, so this entire quantity of water was assumed to have infiltrated. Using the estimated surface area of 4000 m² resulted in a net infiltration of 0.16 m of water over the entire pit. If this infiltration occurred over a two-week period, the infiltration rate during the flooding would have been 1.32×10^{-7} m/s.

The solubility limit for ²⁴¹Am of 6×10^{-9} molal (6×10^{-6} mole/m³) was originally estimated by Robert Smith in Bargelt et al. (1992) based on carbonate being the controlling chemical species. This estimate of solubility was also used in this study. As mentioned previously, rate constants were determined by simultaneously solving two equations involving time to equilibrium and a equilibrium partition coefficient. The partition coefficient for ²⁴¹Am in the source zone was 700 ml/g and the assumed time to equilibrium was 5 days. The resulting rate constants for the dissolution and reprecipitation of ²⁴¹Am from the waste were 1.43×10^{-8} s⁻¹ and 1.60×10^{-5} s⁻¹, respectively. These same rate constants were used to describe the precipitation and dissolution to the sediments.

3.3 Source Term Model Results

The mass release results from the near-field source term model are shown in Figure 7. Both the mass release from the whole pit as well as from a unit length cross section are shown. The latter was used as input to the flow and transport model in the vadose zone because only a unit length section of the pit was simulated. While waste emplacement in the source term model ended shortly after the flood, the mass release rate continued to grow slightly. An important point to remember was that while the mass release rate increased during the flooding, the concentration of the leachate decreased due to the larger volume of water passing through the waste.

4.0 Pit 9 Flooding Simulation: Base Case

This section describes both the methodology used to perform the simulation as well as the results.

4.1 Simulation Methodology

4.1.1 Computer Code

The PORFLOW computer code, Version 2.39, (Runchal and Sagar, 1991) was used to perform the flow and transport simulation. The PORFLOW code is capable of simulating multiphase fluid flow, heat and mass transport in a fractured porous media. Fractures can be simulated either discretely or by superimposing the fracture into a series of elements which represent the surrounding matrix.

4.1.2 Domain

A vertical half-plane extending outward from the centerline of Pit 9 was simulated (see Figure 8). A comparison to the physical domain pictured earlier in Figure 2 indicates how a symmetry plane at the center of the pit was used to reduce the computational burden. As seen in Figure 8, the simulation domain begins at the bottom of Pit 9. The majority of the water exiting the pit would have left via the bottom. The small portion of water, and related contaminants, that exited the sides of the pit is treated as if it left via the bottom of the pit in this simulation. The bottom of the simulation domain was arbitrarily extended 10 meters under the 110' sedimentary interbed. This truncation of the vadose zone was justified since the region of interest in the simulation study was down to the interbed.

The actual domain utilized in the simulation extended 80 m from the centerline of the pit. The choice of 80 m was arbitrary and was based on the lateral extend of perched water spreading in an earlier simulation study (Baca et al, 1992). A subset or portion of the total domain is shown in Figure 9. This same subset is used later to present simulation results. Fractures were assumed to exist every 4 m, which is approximately consistent with Knutson et al. (1992). For conservative purposes or lack of better description, these fractures were treated as if they were through-running from almost the top of the simulation domain to just above the sedimentary interbed.

These fractures were described in the simulation using the superposition option available in PORFLOW, rather than being discretely included in the grid. The fracture aperture was 0.005 m and they were superimposed in an element of 0.1 m width. The resulting ratio of the fracture width to the element width was 1:20. Implications of this ratio are discussed below in the results.

4.1.3 Boundary Conditions

For purposed of the simulation, the left boundary (see Figure 9) or center of the pit was treated as a zero-flux boundary for both flow and transport

because of symmetry. The right boundary was also a prescribed zero-flux for both flow and transport. The boundary was far enough away from the pit so that the flow remained vertical and the no-flux condition was appropriate. The right boundary was also far enough away so either a no-flux or a prescribed concentration of zero would have been appropriate.

Infiltration at the top boundary was incorporated with a combination of a no-flux condition for both flow and transport and a source zones located directly underneath the boundary to describe the infiltrating water and related mass of radionuclide. Two source zones were utilized. One that corresponded to the half-pit, from 0 to 19 m. The other extended from 19 m to 80 m, the outer edge of the simulation domain. This combination of a no-flux boundary at the top and source zones was necessary to prevent contaminants from diffusing back out the top of the simulation domain due the no-flux boundary for transport and then convecting back into the domain.

The background infiltration rate over the entire domain was assigned a constant value of 5 cm/year. This is the same value of infiltration that was used previously by Walton et al. (1989) and Baca et al. (1992). Although it is merely an estimate it was thought to be conservative. The infiltration was treated as if it were constant over the entire year. While this appears to be appropriate outside the RWMC (Baca et al, 1992) it may not be the case inside the RWMC where the surficial sediments are disturbed (McElroy, 1990). Flooding in the simulation was achieved by increasing the infiltration rate to 1.32×10^{-7} m/s for that portion beneath of the upper boundary that represented the bottom of the pit.

The bottom boundary was assigned a prescribed head of -0.3 m for flow which corresponds to a unit gradient or free drainage boundary condition. A zero concentration boundary was assigned at the bottom for transport. These boundary conditions were valid because the simulations were truncated before the wetting front from the flooding reached the bottom boundary.

4.1.4 Initial Conditions

The initial conditions for the simulation were achieved by simulating in transient fashion the movement of water in the subsurface with the background infiltration rate of 5 cm/yr over the entire top of the simulation domain until a steady state condition was reached. The flux through various depths in the domain was monitored as a function of time. When the flux was the same everywhere to two significant figures, the simulation was said to be at steady state.

4.1.5 Simulation Procedure

The resulting steady state simulation was assigned to be time $t=0$ days. The simulation was initiated in a transient mode with background infiltration over the whole top of the domain and the source for ^{241}Am turned on. This source represented the leaching of the nuclide by background infiltration as it was emplaced in the pit. After 547 days (~June, 1969), flooding began and lasted for 14 days. Then background infiltration resumed for the zone representing the bottom of the pit. The simulation continued for a minimum of 100 days

after the flooding or until the wetting front encroached on the bottom boundary.

4.2 Performance Measures

From the advantage of retrospect, it could be seen that the movement of the ^{241}Am in response to flooding had essentially stopped by 100 days after the flooding. The advancement of the contaminant front down the fractures at this time point was chosen as a common comparison point between the different simulations. The complete PORFLOW input data deck for this simulation is contained in the Appendix.

4.3 Results

4.3.1 Flow

The steady-state saturation conditions just before the flooding began are shown in Figure 10. The effect of the fractures can be seen at several different elevations. Near the top of the domain, the fractures draw water from the surrounding vesicular basalt. At the interbed, this additional water caused the region immediately surrounding the fractures to become wetter. The interbed redistributes this water laterally. No fractures were present below the interbed so the saturation profile in that region was uniform horizontally.

The advance of the wetting front in the fractures can be seen in Figures 11 and 12. Saturations at the end of the flooding and one month after the flooding are shown. The fractures played a dominant role in the quick advance of water. A residual wetting front can be seen near the top of the simulation domain where water was forced into the matrix as a result of using a source term description for the flooding. This approximation to the physical system could underestimate the contribution of the fractures in contributing to flow during flooding because water must move through the basalt to reach the fracture.

Figure 13 shows the instantaneous water flux through several horizontal planes in the simulation domain and shows the effect of fractures on the downward movement of water from the flood. The net water flux passing through a hypothetical horizontal plane 0.3 m below the pit shows a single peak or pulse of water due to the flood. Another plane located 3 m meters below the pit and in the vesicular basalt shows two peaks occurring approximately 1 month apart. The peak occurring first is due to the flood water moving through the fractures while the second peak is due to the wetting front moving through the basalt matrix. It takes only a few days after the start of the flood for the water to reach this depth via the fractures but several days for the water to arrive via the matrix. The first peak is more abrupt indicating the bulk of water passes through the fractures relatively rapidly. This is expected given the higher conductivity of the fractures. Flux through another plane located further beneath the pit (5 m) but still in the vesicular basalt exhibits the same behavior but the peaks are lower and occur later in time. The peak due

to flow in the matrix is especially drawn out indicating a lower conductivity and more dispersion. The final line on the graph shows the water flux through a plane nearly 20 m below the pit. This depth is immediately above the vesicular basalt/massive basalt interface. The increased water due to flooding has not yet peaked at this depth even one year after the flood. This is because at this depth the pulse of flood water in the fractures has dissipated into the surrounding basalt matrix.

4.3.2 Transport

The main objective in this simulation study was to evaluate the feasibility of dissolved-phase transport of ^{241}Am during flooding to explain the presence of radionuclides in the 110' interbed beneath the RWMC. As previously mentioned, the performance measure was the advance of the contaminant front with time. A flux plane analysis for the advance of the ^{241}Am , similar to the analysis for flow, showed that the rate of advance of the contaminant front had slowed down to the background rate by approximately 100 days after the flooding occurred (see Figure 14).

For the input parameters described in the conceptual model, the resulting concentration profile at 100 days is shown in Figure 15. As can be seen, the contaminant advance was approximately 2.2 m, based on a concentration of 1×10^{-12} moles/m³. There is admittedly a large component of numerical diffusion laterally away from the fractures. Since the performance measure considered transport vertically, this lateral diffusion does not significantly affect the results.

This result contrasts significantly with that of Walton et al. (1989) for several reasons. First, Walton used a conservative or non-sorbing contaminant. Second, Walton assumed the fractures completely controlled flow in the basalt, even for the two-dimensional flooding analysis.

There are several implications of these results. First, dissolved phase flow that undergoes significant sorption, such as ^{241}Am and other actinides undergo, is not likely to move significant amounts of contaminant during flooding. Second, the method of superimposing the fracture into an discretized element appears to allow undue sorption, even when the partition coefficient in the fracture is quite low. This may indicate that the 1:20 ratio for the fracture width to the element width was too high.

5.0 Parametric Sensitivity Study

Based on the results obtained for the base case presented in Section 4, a parametric sensitivity study was conducted to determine which were the controlling parameters. This study consisted of decreasing the partition coefficient in the fractures, the source term model, and the basalt matrix. Additionally, a simulation was performed with the aperture and conductivity of the fractures increased. Each of these will be discussed in turn.

5.1 Partition Coefficient in Fractures: Run 1

The partition coefficient in the fractures for the base case was 2.0 ml/g. In this simulation, the partition coefficient was decreased to 0 ml/g. The resulting concentration profile almost exactly matched that shown for the base case in Figure 15 and is not shown. It seems likely that the 1:20 ratio between the fracture aperture and the width of the element in which it was embedded damped out the transport contribution of the fracture. Normally, a significant change would be expected. In all subsequent runs the partition coefficient of the fracture was left at 0 ml/g.

5.2 Partition Coefficient in Source Term: Run 2

The effect of the partition coefficient in the source term was tested in this sensitivity simulation. The partition coefficient was reduced from 700 ml/g to 70 ml/g. The source term code was rerun and the new mass release rate from the bottom of Pit 9 was input to the PORFLOW code. The resulting concentration profile at 100 days after the end of flooding is shown in Figure 16. By comparison to the base case, it can be seen that the contaminant front advanced slightly further.

5.3 Partition Coefficient in Basalt Matrix: Runs 3 and 4

The effect of sorption of the ^{241}Am onto the basalt matrix was tested in this sensitivity simulation by reducing the partition coefficient for the vesicular and massive basalt matrix from 70 ml/g to 7 ml/g. This resulted in the contaminant front advancing further but still not to the interbed. The concentration profile is shown in Figure 17.

The last simulation to examine the sensitivity of contaminant advance to partition coefficients used a conservative or non-sorbing K_d for the basalt. The concentration profile for this run is shown in Figure 18. Significant levels of ^{241}Am are shown to have reached the interbed after 100 days of flooding. These results compare favorably with the two-dimensional flooding results in Figures 8 and 9 in Walton et al. (1989).

One conclusion that may be drawn from this latter sensitivity run is that mechanisms that would make the ^{241}Am , or other nuclides, non-sorbing could result in transport down to the interbed. Mechanisms such as sorption onto colloidal particles or organic complexation have been hypothesized to account

for the increased mobility that seems to be observed at the RWMC (Rawson et al, 1991). These mechanisms are deserving of further study.

5.4 Conductivity of Fractures: Run 5

The fractures in the base case were assumed to be 50% infilled with overlying sediment which reduced their conductivity below that for the equivalent open fracture. The effect of the assumed hydraulic conductivity was tested in the final sensitivity run by increasing the fracture conductivity from 85 m/day to 300 m/day and the aperture from 0.005 to 0.01 m. This reduced the ratio of the aperture to the element in which it was embedded to 1:10. The resulting concentration profile is shown in Figure 19 and shows a slight increase in the advance of the contaminant front compared to the base case.

5.5 Sensitivity Result Analysis

Table III presents a tabulation of the key input parameters for each run and the depth to which the simulated contaminant front traveled 100 days after the end of the flood. The basalt K_d can be seen to dominate the results. This is partly a function of the numerical method used to superimpose the fractures.

It is believed that in the physical problem, sorption would likely be higher in the fractures due to infilling of the fractures by overlying sedimentary material. This would support the conclusion from Run 4, in which a conservative solute reaches the interbed as a result of flooding. In this light, the effect of the partition coefficient in the fractures is inconclusive and deserving of further study.

The effect of the partition coefficient in the source term model appears to be somewhere in between that of the partition coefficient for the fracture and that of the basalt matrix. Likewise, increasing the fracture conductivity only resulted in a slight increase in the depth of contaminant movement as a result of the flooding.

Table III. Advance of ^{241}Am 100 Days After Flooding

Sensitivity Run	Fracture	K_d (ml/g) Source Basalt	Fracture Conductivity (m/day)	Advance of 1×10^{-12} Contour (m)
Base Case	2	700 70	85	2.2
Run 1	0	700 70	85	2.2
Run 2	0	70 70	85	2.6
Run 3	0	70 7	85	10.5
Run 4	0	70 0	85	24.4
Run 5	0	700 70	300	3.2

6.0 Conclusions

A simulation study was conducted to determine whether dissolved-phase transport during historical flooding events at the RWMC could explain the presence of radionuclides in sedimentary interbeds at depth. Sensitivity of the depth of radionuclide movement in response to the flooding to key input parameters was also tested. It appears likely that dissolved phase transport during flooding by itself is not a feasible mechanism to explain the presence of radionuclides in the 110' interbed. The sensitivity study results indicate that other mechanisms such as colloidal transport or organic complexation may be contributing to rapid transport during flooding.

Caution in interpreting the simulation results is advised due to the approximations involved in embedding the fracture within numerical elements which represent the basalt matrix. Additionally, the assumption of equilibrium partitioning in the vadose zone simulation likely affects the results.

Consistent with work by Walton et al (1989), it was observed that if nuclides are transported to the 110' interbed during flooding, the interbed serves as a barrier to further transport. The implication is that, for performance assessment purposes, the interbeds would serve to lessen the affect of future flooding.

7.0 Future Directions

The results of this report serve as a starting point for additional studies. Several interesting approaches available to try and explain the presence of radionuclides in the interbed are:

- o Reactive transport simulations with discrete fracture-matrix interaction during flooding
- o Simulation of radiocolloid transport during flooding
- o Simulation of reactive-transport and/or radiocolloid transport during seasonal infiltration events.

A reactive transport study would test whether the equilibrium partitioning in the vadose simulation was appropriate. To gain any useful information, a study of this type would have to be coupled with a site-specific geochemical study on the likely behavior of the nuclides in the subsurface.

Transport via radiocolloids is an area of ongoing research at a number of locations. In the opinion of the authors, this mechanism seems quite likely to have occurred during flooding and is deserving of further study.

The last suggested approach may have the greatest impact on the performance assessment process. Seasonal infiltration likely occurs each year within the RWMC. Evidence has been collected to support this (McElroy, 1990). Unfortunately, the collection of data to ascertain the amount of this seasonal infiltration was discontinued in FY-1990. A sensitivity study on the effect of seasonal infiltration coupled with reactive transport could determine the value of understanding the amount and timing of possible seasonal infiltration.

In conclusion, each of these suggested approaches to determining whether past (or future) flooding could serve as transport mechanism rely on a understanding of the movement of water in the subsurface beneath the RWMC. This understanding would rely on field measurements to validate the hypotheses used in this simulation study and the suggested studies. Until such data is collected, analyzed, and made available, our ability to model the movement of water and contaminants in the subsurface will only be as good as our ability to substantiate the hypotheses without supporting field measurements.

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Figure 1. Location of INEL and the RWMC

Figure 2. Conceptual Sketch of Pit 9 Physical Setting

Figure 3. Hypothesized Subsurface Transport Processes

Figure 4. Pit 9 Simulation Moisture Characteristic Curves

Figure 5. Pit 9 Simulation Hydraulic Conductivity Curves

Figure 6. Flowchart of Compartmental Source Term Model

Figure 7. Source Term Model Results for ^{241}Am

Figure 8. Sketch of Pit 9 Simulation Model Domain

Figure 9. Pit 9 Simulation Modeling Domain

Figure 10. Steady-State Saturation Profile Before Flooding

Figure 11. Saturation Profile at End of Flooding

Figure 12. Saturation Profile One Month After Flooding

Figure 13. Instantaneous Water Flux Through Selected Planes

Figure 14. Instantaneous ^{241}Am Flux Through Selected Planes

Figure 15. Base Case ^{241}Am Profile

Figure 16. Run 2 ^{241}Am Profile: Source K_d reduced to 70 ml/g

Figure 17. Run 3 ^{241}Am Profile: Basalt K_d reduced to 7 ml/g

Figure 18. Run 4 ^{241}Am Profile: Basalt K_d reduced to 0 ml/g

Figure 19. Run 5 ^{241}Am Profile: Increased Fracture Conductivity

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APPENDIX

The following is a complete PORFLOW input file for the base case run presented in Section 4.

TITLE two-d Simulation for RWMC Pit NINE (version two)

/ GRID 151 by 223

/ SCREEn echo

/ COOR X:

-0.5	0.5	1.0	1.5	2.0	2.4	2.7	2.9	3.0	3.1
3.3	3.6	4.0	4.5	5.0	5.5	6.0	6.4	6.7	6.9
7.0	7.1	7.3	7.6	8.0	8.5	9.0	9.5	10.0	10.4
10.7	10.9	11.0	11.1	11.3	11.6	12.0	12.5	13.0	13.5
14.0	14.4	14.7	14.9	15.0	15.1	15.3	15.6	16.0	16.5
17.0	17.5	18.0	18.4	18.7	18.9	19.0	19.1	19.3	19.6
20.0	20.5	21.0	21.5	22.0	22.4	22.7	22.9	23.0	23.1
23.3	23.6	24.0	24.5	25.0	25.5	26.0	26.4	26.7	26.9
27.0	27.1	27.3	27.6	28.0	28.5	29.0	29.5	30.0	30.4
30.7	30.9	31.0	31.1	31.3	31.6	32.0	32.5	33.0	33.5
34.0	34.4	34.7	34.9	35.0	35.1	35.3	35.6	36.0	36.5
37.0	37.5	38.0	38.4	38.7	38.9	39.0	39.1	39.3	39.6
40.0	40.5	41.0	41.5	42.0	42.75	43.5	44.25	45.0	46.0
47.0	48.0	49.0	50.0	51.0	52.5	54.0	55.5	57.0	58.5
60.0	62.0	64.0	66.0	68.0	70.0	72.0	74.0	76.0	78.0
80.0									

/ COOR Y:

// Nodes for vesicular basalt under the interbed (1-56) (56 nodes)

-0.1	0.1	0.3	0.5	0.7	0.9	1.1	1.3	1.5	1.7
1.9	2.1	2.3	2.5	2.7	2.9	3.1	3.3	3.5	3.7
3.9	4.1	4.3	4.5	4.7	4.9	5.1	5.3	5.5	5.7
5.9	6.1	6.3	6.5	6.7	6.9	7.1	7.3	7.7	7.9
8.1	8.3	8.5	8.7	8.9	9.1	9.25	9.4	9.5	9.6
9.7	9.77	9.83	9.88	9.93	9.98				

// Nodes for 110' Sediment interbed (57-75) (19 nodes)

10.02	10.06	10.1	10.15	10.2	10.27	10.35	10.45	10.57	10.7
10.85	11.05	11.17	11.26	11.3	11.35	11.4	11.44	11.48	

// Nodes for Massive Basalt (76-106) (31 nodes)

11.52	11.56	11.60	11.65	11.70	11.77	11.85	11.95	12.1	12.3
12.5	12.7	12.9	13.1	13.3	13.5	13.7	13.9	14.1	14.3
14.5	14.7	14.9	15.1	15.3	15.45	15.5	15.6	15.68	15.74

15.78

//
// Nodes for vesicular basalt (107-223) (117 nodes)
//
15.82 15.86 15.9 15.95 16.0 16.05 16.1 16.17 16.25 16.35
16.47 16.6 16.75 16.9 17.05 17.2 17.35 17.5 17.65 17.8
17.95 18.1 18.25 18.45 18.65 18.85 19.05 19.25 19.45 19.65
19.85 20.05 20.25 20.45 20.65 20.85 21.05 21.25 21.45 21.65
21.85 22.05 22.25 22.45 22.65 22.85 23.05 23.25 23.45 23.65
23.85 24.05 24.25 24.45 24.65 24.85 25.05 25.25 25.45 25.65
25.85 26.05 26.25 26.45 26.65 26.85 27.05 27.25 27.45 27.65
27.85 28.05 28.25 28.45 28.65 28.85 29.05 29.25 29.45 29.65
29.85 30.05 30.25 30.45 30.65 30.85 31.05 31.25 31.45 31.65
31.85 32.05 32.25 32.45 32.65 32.85 33.05 33.25 33.45 33.65
33.85 34.05 34.25 34.45 34.65 34.85 35.05 35.25 35.4 35.5
35.58 35.65 35.70 35.74 35.77 35.79 35.81

/

ZONE 1 from (1, 1) to (151, 56) \$ vesicular basalt
ZONE 3 from (1, 57) to (151, 75) \$ interbed same as surface sediments
ZONE 2 from (1, 76) to (151,106) \$ massive basalt
ZONE 1 from (1,107) to (151,223) \$ vesicular basalt

/

ZONE 4 from (9,76) to (9,219) FRAC width = 0.005 m \$ fracture
ZONE 5 from (21,76) to (21,219) FRAC width = 0.005 m \$ fracture
ZONE 6 from (33,76) to (33,219) FRAC width = 0.005 m \$ fracture
ZONE 7 from (45,76) to (45,219) FRAC width = 0.005 m \$ fracture
ZONE 8 from (57,76) to (57,219) FRAC width = 0.005 m \$ fracture
ZONE 9 from (69,76) to (69,219) FRAC width = 0.005 m \$ fracture
ZONE 10 from (81,76) to (81,219) FRAC width = 0.005 m \$ fracture
ZONE 11 from (93,76) to (93,219) FRAC width = 0.005 m \$ fracture
ZONE 12 from (105,76) to (105,219) FRAC width = 0.005 m \$ fracture
ZONE 13 from (117,76) to (117,219) FRAC width = 0.005 m \$ fracture

/

LOCAtE SOURce 1 from (1,222) to (57,222) \$ water through pit
LOCAtE SOURce 2 from (58,222) to (151,222) \$ water background
LOCAtE SOURce 3 from (1,222) to (57,222) \$ mass in pit

/

FOR ZONE 1 \$ vesicular basalt k based on Carolyn Bishops thesis data
ROCK density=1900 kg/m*m*m, poros: 0.228 0.228 0.228 \$C. Bishop value
HYDR ss = 1.E-3, kx= 8.64e-2 ky= 8.64e-2 m/day
MULTiphase flow: VAN, n = 1.4741 alpha = 3.84 Swr=0.0658 \$C. Bishop's values
TRANsport props: Kd=.07 mm/kg, Dm=1.E-4, al=1.0, at=0.1

/

FOR ZONE 2 \$ massive basalt k based on 1 to 2 meters of ponding in this layer
ROCK dsty=1900 kg/m*m*m, poros=0.145, 0.145 0.145 \$from C. Knutson 85-110
basalt
HYDR ss = 1.E-3, kx= 1.0e-2 ky= 1.0e-2 m/day
MULTiphase flow: VAN, n = 1.4741 alpha = 3.84 Swr=0.0658 \$C. Bishop's values
TRANsport props: Kd=.07 mm/kg, Dm=1.E-4, al=1.0, at=0.1

/

FOR ZONE 3 \$ 110' interbed
ROCK density=1500 kg/m*m*m, poros = 0.480 , 0.480 , 0.480 \$ D. McElroy's data

```

HYDR ss = 1.E-3 kx = 6.560e-2 ky = 6.560e-2 m/day
MULTiphase flow: VAN n= 1.36, alpha= 1.216 Swr=0.21 $McElroy's data using RETC
TRANsport props: Kd=0.7 mm/kg, Dm=1.E-4, al=1.0, at=0.1
/
FOR ZONEs 4 to 13 by 1 $fracture, drainable soil from literature
ROCK density=1500 kg/m*m*m, poros = 0.500 , 0.500 , 0.500
HYDR ss = 1.E-4 kx = 85.0 ky = 85.0 m/day (~one hundred Darcies)
MULTiphase flow: VAN n= 2.43, alpha= 3.00 Swr=0.01 $Wosten and vg (1988)
TRANsport props: Kd=.002 mm/kg, Dm=1.E-4, al=5., at=0.1
/
DATUm = 0. 0.
GRAVITY COMPONENTS: 0., -1.
PROPERty options: GEOMetric mean
CONVergence for C: LOCAL mode; value=1.e-4 max iterations = 7
CONVergence for P: LOCAL mode; value=1.e-4 max iterations = 7
/
READ 1st record from 'pit9.ici'
INIT C = 0 everywhere
/
SOURce 1 for P VOLU, 6 sets, flooding region, matches MSOURCE data
      0.0, 6.850e-3 $ pit 9 open 11-8-67
      547.0, 6.850e-3
      547.5, 0.5714 $ flood begin 5-8-69
      561.0, 0.5714 $ flood end 5-22-69
      561.3, 6.850e-3
      1.e+30, 6.850e-3
SOURce 2 for P VOLU, 2 sets, background infiltration of five cm/yr
      0.0, 6.850e-3
      1.e+30, 6.850e-3
SOURce 3 for C, 53 sets,
      0.000000E+00, 0.0000E+00
      2.893519E+01, 2.2244E-13
      5.787037E+01, 8.9845E-13
      1.067824E+02, 3.0623E-12
      1.556944E+02, 6.4940E-12
      2.046065E+02, 1.1176E-11
      2.535185E+02, 1.7091E-11
      3.024306E+02, 2.4222E-11
      3.513426E+02, 3.2552E-11
      4.002546E+02, 4.2064E-11
      4.491667E+02, 5.2742E-11
      4.980787E+02, 6.4571E-11
      5.469907E+02, 7.7533E-11
      5.470833E+02, 9.8186E-10
      5.471759E+02, 1.7840E-09
      5.472685E+02, 2.5548E-09
      5.473611E+02, 3.3089E-09
      5.474537E+02, 4.0529E-09
      5.501356E+02, 4.0837E-09
      5.528307E+02, 4.1227E-09
      5.555390E+02, 4.1622E-09
      5.582605E+02, 4.2020E-09

```

5.609954E+02,	4.2421E-09
5.610648E+02,	3.4639E-09
5.611343E+02,	2.6753E-09
5.612037E+02,	1.8691E-09
5.612731E+02,	1.0294E-09
5.613426E+02,	8.1355E-11
6.040867E+02,	9.3902E-11
6.500856E+02,	1.0713E-10
6.995871E+02,	1.2116E-10
7.528579E+02,	1.3605E-10
8.101852E+02,	1.5182E-10
9.228689E+02,	1.8208E-10
1.051225E+03,	2.1538E-10
1.197434E+03,	2.5185E-10
1.363977E+03,	2.9157E-10
1.553685E+03,	3.3457E-10
1.769777E+03,	3.8079E-10
2.015925E+03,	4.3006E-10
2.296307E+03,	4.8209E-10
2.615687E+03,	5.3646E-10
2.979487E+03,	5.9257E-10
3.393885E+03,	6.4969E-10
3.865920E+03,	7.0688E-10
4.403608E+03,	7.6311E-10
5.016079E+03,	8.1720E-10
5.713734E+03,	8.6794E-10
6.508423E+03,	9.1414E-10
7.413640E+03,	9.5473E-10
8.444758E+03,	9.8886E-10
9.619288E+03,	1.0160E-09
1.095718E+04,	1.0359E-09

/

BOUN	P	-1	GRAD = 0.
BOUN	P	+1	GRAD = 0.
BOUN	P	-2	NODE = -0.3
BOUN	P	+2	GRAD = 0.

/

BOUN	C	-1	GRAD = 0.
BOUN	C	+1	GRAD = 0.
BOUN	C	-2	NODE = 0.
BOUN	C	+2	GRAD = 0.

/

FLUX	BALA	for P	from (1,221)	to (151,221)	every 100 steps
FLUX	BALA	for P	from (1,215)	to (151,215)	every 100 steps
FLUX	BALA	for P	from (1,200)	to (151,200)	every 100 steps
FLUX	BALA	for P	from (1,190)	to (151,190)	every 100 steps
FLUX	BALA	for P	from (1,110)	to (151,110)	every 100 steps
FLUX	BALA	for P	from (1,105)	to (151,105)	every 100 steps
FLUX	BALA	for P	from (1, 90)	to (151, 90)	every 100 steps
FLUX	BALA	for P	from (1, 50)	to (151, 50)	every 100 steps
FLUX	BALA	for P	from (1, 3)	to (151, 3)	every 100 steps

/

```

FLUX BALA for C from ( 1,221) to (151,221) every 100 steps
FLUX BALA for C from ( 1,214) to (151,214) every 100 steps
FLUX BALA for C from ( 1,189) to (151,189) every 100 steps
FLUX BALA for C from ( 1,164) to (151,164) every 100 steps
FLUX BALA for C from ( 1,139) to (151,139) every 100 steps
FLUX BALA for C from ( 1,107) to (151,107) every 100 steps
FLUX BALA for C from ( 1, 90) to (151, 90) every 100 steps
FLUX BALA for C from ( 1, 76) to (151, 76) every 100 steps
FLUX BALA for C from ( 1, 57) to (151, 57) every 100 steps
/
DIAGnostic node at 2,96 print every 1 steps
WINDow from 2,2 to 2,2
/
MATRix sweeps X and Y directions for P and C
DEBUg USER for C every 10 steps
/
SAVE C every 50 steps in 'cbefore'
SOLVe for 547.0 day in steps 1.00 day increase by 1.00 with a max 1000.
SAVE U,V,P,C,S in 'cbegin'
/
SAVE P C every 200 steps in 'pcflood'
SOLVe for 16.0 day in steps 0.02 day increase by 1.00 with a max 1000.
SAVE U,V,P,C,S NOW in 'cstop'
/
SOLVe for 1.0 day in steps 0.02 day increase by 1.00 with a max 1000.
SOLVe for 2.0 day in steps 0.04 day increase by 1.00 with a max 1000.
SAVE P C NOW in 'pcafter' $ 5 days after flood
SOLVe for 5.0 day in steps 0.05 day increase by 1.00 with a max 1000.
SAVE P C NOW in 'pcafter' $ 10 days after flood
SOLVe for 10.0 day in steps 0.06 day increase by 1.00 with a max 1000.
SAVE P C NOW in 'pcafter' $ 20 days after flood
SOLVe for 10.0 day in steps 0.10 day increase by 1.00 with a max 1000.
SAVE P C NOW in 'pcafter' $ 30 days after flood
SOLVe for 10.0 day in steps 0.2 day increase by 1.00 with a max 1000.
SOLVe for 20.0 day in steps 0.4 day increase by 1.00 with a max 1000.
SAVE P C NOW in 'pcafter' $ 60 days after flood
SOLVe for 20.0 day in steps 0.6 day increase by 1.00 with a max 1000.
SOLVe for 20.0 day in steps 0.8 day increase by 1.00 with a max 1000.
SAVE P C NOW in 'pcafter' $ 100 days after flood
SOLVe for 265.0 day in steps 1.0 day increase by 1.00 with a max 1000.
SAVE U,V,P,C,S NOW in 'cafter' $ 365 days after flood
/
END

```

**DATE
FILMED**

8 / 16 / 93

END