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Interflow in Semiarid Environments: An Overlooked Process in Risk Assessment

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ABSTRACT

Risk assessment, both human and ecological, embodies fundamental assumptions about hydrological processes, especially how they affect the movement of contaminants in the environment. The lateral movement of water through the soil, or interflow, is frequently a component of risk assessments for humid environments, but not of those for semiarid environments. Our research has shown that, contrary to what was previously thought, interflow can be important in semiarid landscapes and is, therefore, an essential consideration for risk assessment in these regions. To illustrate and assess the effect of interflow on estimates of risk, we (1) developed a simple conceptual model to describe the role that interflow may have in the redistribution of surface and near-surface contamination, and (2) used RESRAD, an exposure model for assessing radionuclide doses to humans, to evaluate the effectiveness of landfill covers in mitigating doses of three contaminants (^{3}H, ^{238}U, and $^{239,240}Pu$) at a site in northern New Mexico at which interflow is known to be occurring. Only those calculations of the model that took interflow into account yielded the result that the radionuclides would contaminate groundwater — underscoring the potential importance of interflow as a mechanism for the transport of contaminants. We conclude that failure to take interflow into account can render risk assessments inaccurate and remediation ineffective. Further, our work demonstrates that a general understanding of hydrological processes is essential for accurate risk assessment, ecological as well as human.

Key Words: runoff, contaminant transport, interflow, New Mexico hydrology, risk assessment, RESRAD, tritium, uranium, plutonium, landfill

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INTRODUCTION

Assessing the impact of environmental contamination in semiarid areas is an enormous task. Within the DOE complex alone, large tracts are potentially contaminated — for example, Los Alamos, Hanford, the Nevada Test Site, Idaho National Engineering Laboratory, Rocky Flats, and Pantex (Riley and Zachara, 1992). Human and ecological risk assessments play a central role in the development of mitigation strategies (Harwell, 1989; Bartell, Gardner, and O'Neill, 1992; Suter, 1993). To be effective, however, risk assessment must be based on a sound understanding of environmental processes (Till, 1988). Without that understanding, even the most elaborate risk assessment exercise will be fatally flawed. In many cases, the most important of these environmental processes is the movement of water, a primary transporter of contaminants. Even in semiarid landscapes, which are by definition water-limited, water movement is often the principal mechanism by which contaminants are redistributed (Hakonson, Lane, and Springer, 1992).

The escalating costs of remediation are resulting in calls to ensure that remediation work not only reduces actual risk, but is cost-effective (McGuire, 1989; Abelson, 1990, 1992, 1993; Zeckhauser and Viscusi, 1990; Breshears, Whicker, and Hakonson, 1993). Risk assessment is intended to drive the evaluation of remediation options by helping to answer such questions as: Which options result in the most risk reduction? Is remediation "riskier" than no remediation?

One remediation option that is being evaluated for many drier locations, especially where contaminants are buried, is that of surface covers (Nyhan, Hakonson, and Drennon, 1990; Caldwell, 1992; Black and Latham, 1994). These covers are designed to prevent or reduce (1) the vertical movement of water through the contaminated material, and (2) the movement of contaminated sediments across the surface. They can be as simple as a layer of gravel or as complex as a multilayered sequence of earth materials and geotextiles. In either case, the primary purpose is to isolate the contaminated material from the environment by reducing movement of water. However, surface covers can be effective only as long as the main direction of water movement is from top to bottom; if interflow—water moving laterally through subsurface soils—is occurring, even the best-designed surface cover can be circumvented unless provisions are made to divert the water.

In this paper, we evaluate the relative importance of interflow as a mechanism of contaminant movement in water-limited landscapes, especially areas in which engineered surface covers have been put in place to contain contaminants. Specifically, our objective is to evaluate the implications of interflow for risk assessment in semiarid environments, with respect to both surface and subsurface contamination.

CONCEPTUAL MODELS OF THE WATER BUDGET

Fundamental to understanding how water moves in semiarid environments is knowledge of the *water budget*, that is, how water is partitioned in the environment (Dunne and Leopold, 1978). Major categories within the water budget are evapotranspiration, runoff, storage in the soil, and storage in groundwater.

In drier environments, most of the water (generally, 85–100%) is lost through evaporation or transpiration (Dunne and Leopold, 1978). Of the water that enters the soil, very little percolates beyond the root zone; most is eventually lost via evapotranspiration. (The high evapotranspiration rate, combined with low amounts of precipitation in arid and semiarid landscapes, explains the suitability of these environments for long-term storage of wastes — Reith and Thompson, 1992). Surface runoff accounts for most of the remaining water and is a very important agent of transport for sediment, nutrients, and contaminants. How much water runs off and how much percolates beyond the root zone are critical factors in assessing risk.

Water that is retained in the soil may move either vertically or laterally through the soil. If the soil contains contaminants, these can be transported by the moving water. Water that moves vertically through the soil and into the underlying earth material will, given enough time, reach groundwater. Even though groundwater recharge is very small in most dry environments, this process has become a subject of increasing interest and study because of the long-term potential for groundwater contamination (Meyer, 1992).

Water that moves laterally through the soil, or interflow, has generally been overlooked in studies of semiarid landscapes. Although recognized as common in humid environments, such as the eastern United States (Anderson and Burt, 1985), interflow had been thought not to be important in semiarid environments. But our research has led us to conclude that interflow can be an important contributor to the water budget in areas in which annual precipitation exceeds 450 mm/year (Wilcox et al., 1996). In such cases, failure to take it into account can lead to incorrect estimates of contaminant transport, which leads to poor risk assessment and inappropriate application of environmental restoration technologies.

Generally, for interflow to occur, two conditions must be satisfied: (1) there must be an impermeable layer close to the surface (either in the soil or in the underlying parent material) that greatly reduces the vertical movement of water; and (2) there must be enough precipitation to saturate a portion of the soil above the impermeable layer, allowing the development of a perched saturated zone (interflow has been observed in unsaturated soils, but only in small quantities — Mulholland, Wilson, and Jardine, 1990). And, of course, some slope is required. Since water in this zone cannot move vertically, it will move laterally down-gradient.

Detailed hydrometric studies on a 900-m^2 hillslope, in a semiarid ponderosa pine forest within the boundaries of the Los Alamos National Laboratory, have demonstrated that interflow is not only a very important process, but is the major mechanism by which sustained streamflow is generated from these forests (Wilcox et al., 1996). During snowmelt or periods of prolonged rainfall, a saturated zone develops at a depth of about 1 m (which corresponds to the

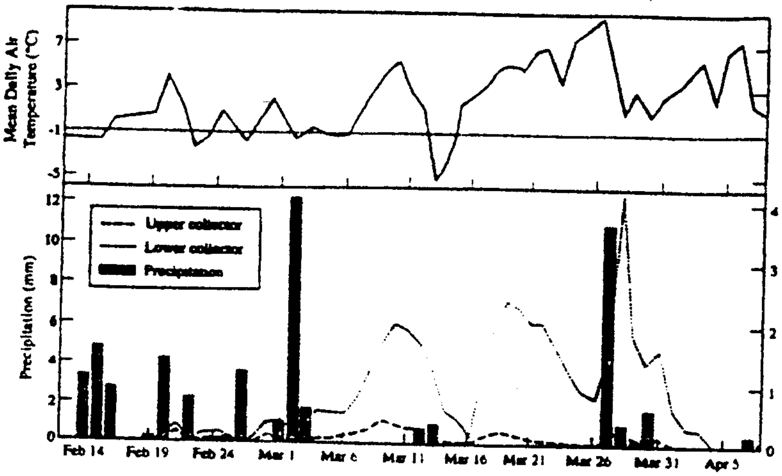

Figure 1. Daily interflow vs. temperature and precipitation, February 14 - April 5, 1993 (from Wilcox et al., 1996).

interface between the soil and the underlying tuff bedrock), and interflow can be sustained for weeks.

Two of the four winters of our study have seen significant amounts of interflow — the largest during the winter of 1992-1993, when the snowpack was above average. Nearly 50 mm of interflow (representing about 20% of the winter snow pack) was measured, most of it from the clay-rich Bt horizon. Daily interflow measurements during the late winter and spring of 1993, and their correlation with precipitation and average daily temperature, are shown in Figure 1. We recorded three major phases of interflow as the snow pack melted (which began in the latter half of February, when air temperatures began to rise). The first phase, in early March, showed a clear correspondence with rising temperatures; interflow dropped off sharply when a period of below-freezing temperatures ensued in mid March. The second major phase, which began around March 16, also corresponded with a rise in air temperatures that further reduced the snow pack. The third phase, in late March, resulted from a rain-on-snow event that melted much of the remaining snow pack. Interflow may occur in summer as well, but only in small amounts.

On average, interflow has accounted for a small portion of the total water budget for the site. At the same time, our data not only show that interflow can periodically be a very important runoff mechanism, but they reveal its dynamic nature: water can move through these soils at a faster rate than can be explained by the hydraulic properties of the soil matrix. In humid environments, where the phenomenon of interflow has been well studied, it has been shown that "macropores" (large pores or cracks in the soil) are capable of conducting large quantities of water at very rapid rates (Beven and German, 1982). And at our site as well, measurements demonstrate that the very dynamic and often quite rapid movement of interflow is explained by the fact that it travels primarily through macropores (Newman, 1996). Using stable isotopes and other natural tracers, Newman demonstrated the preferential movement of water through macropore networks in the soil. The conceptual model developed from these studies protrays the preferential flow process and macropore/matrix interactions.

Before having the results from these studies, we had developed a conceptual model of water movement in semiarid ponderosa pine forests that focused on surface runoff and groundwater recharge as transport mechanisms. Now, in light of these new findings, we have revised the model (Fig. 2). Our results confirm that surface runoff is very important in these areas, but they also show that recharge to groundwater is much less important than originally assumed because of the presence of a zone of low permeability (restrictive horizon) in the soil, very little water is able to infiltrate and recharge groundwater bodies. Even more important is that during the periods of high water availability (spring snowmelt, prolonged frontal storms), it is this restrictive horizon that causes water to build up, creating a perched zone of saturation that gives rise to interflow (see Fig. 2b). Such a layer of low permeability, which may be bedrock or a natural soil feature, is more often present than not.

In addition, water generated as interflow can reemerge at the surface and flow into a nearby stream channel. It is this mechanism that can sustain streamflow for a period of days or weeks. Interflow waters, then, can also transport contaminants from the soil to the surface and, now as surface runoff, carry them swiftly downslope.

These findings concerning the importance of interflow and its relevance to risk assessment are timely given that risk assessments are currently under way for numerous contaminated sites in and around Los Alamos (Dorries et al., 1993; Hartmann et al., 1993).

IMPLICATIONS OF INTERFLOW FOR CONTAMINANT MOVEMENT AND RISK ASSESSMENT

Potential Effect of Interflow on Contaminant Movement

The potential effect of even occasional occurrences of interflow on contaminant movement is illustrated by the following two examples. The first is built on a qualitative evaluation of contaminant movement, whereas the second involves a more quantitative analysis.

Example I: Movement of Near-Surface Contaminants

Many of the activities of the Los Alamos National Laboratory have involved testing of explosives devices or materials. Some of the explosions created shallow pits that contain contaminants at or near the surface (Los Alamos National Laboratory, 1995). Water collects in these pits and easily moves into the soil, potentially carrying contaminants with it. In other areas of such test

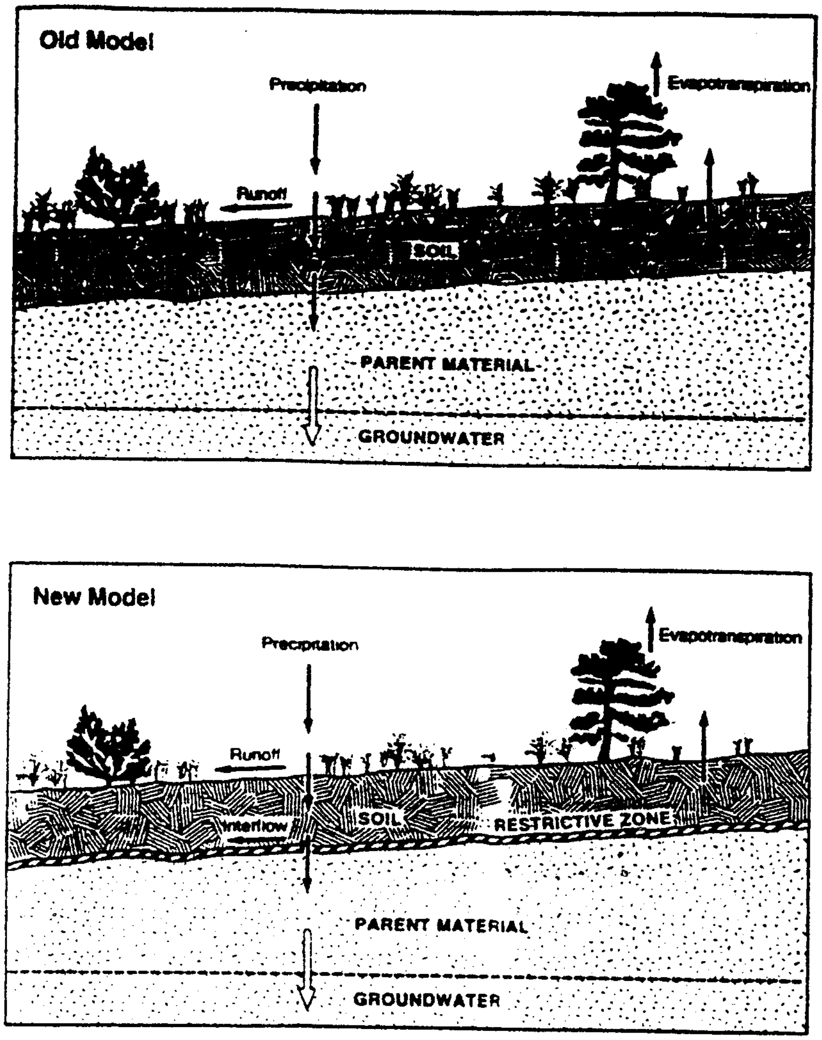

Figure 2. Conceptual models of water movement in semiarid environments: (a) old model; (b) new model.

sites, waste has been piled on the ground surface and covered with earth material; contaminants present in the buried waste could also be transported by water percolating through the overburden.

Using our original conceptual model (Fig. 2a), our focus would have been on the potential for vertical movement of contaminants towards the groundwater, and we would have structured the investigation to detect downward movement of water and contaminants. But for the shallow pits as well as the buried waste piles, the subsoil remained undisturbed and thus acts as a "restric-

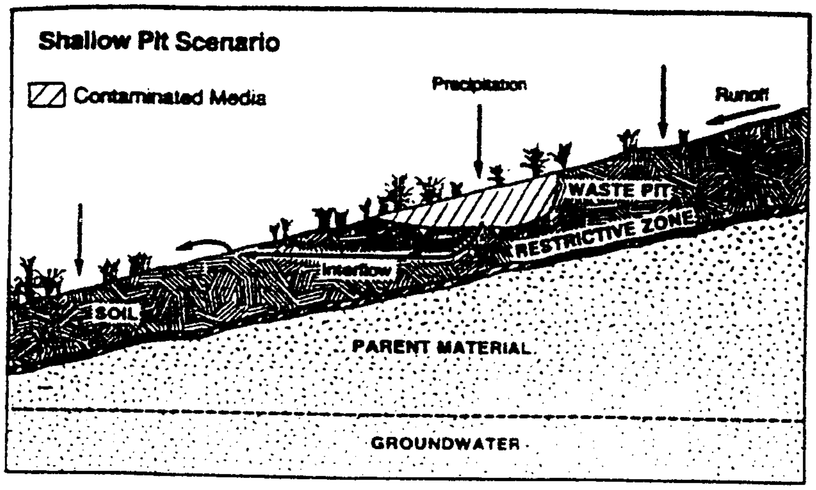

Figure 3. Potential effect of interflow on movement of surface contaminants.

tive horizon," impeding the vertical movement of water. Our new conceptual model (Fig. 2b) takes this factor into account; it supposes that if the pit does not extend below the restrictive horizon, it is much more likely that water and contaminants will move laterally — and therefore that contaminants may instead be redistributed within downslope soils or be carried back to the surface and thence transported elsewhere by surface water or wind (Fig. 3). This revised perspective not only improves our assessment of associated risks, but allows for more effective site characterization. Further, the model allows us to monitor future contaminant movement, whether or not the site is remediated.

Example II: Movement of Contaminants "Isolated" by a Surface Cover

Surface covers are an especially effective form of environmental restoration for sites in semiarid environments that contain buried contaminants (Nyhan, Hakonson, and Drennon, 1990; Caldwell, 1992). They are designed to prevent or minimize the vertical movement of water into contaminant-containing zones. If, however, interflow is an active process at the contaminated site, a surface cover will be ineffective: it can be completely bypassed by water moving laterally (Fig. 4). Moreover, because the pits dug for disposal of the wastes are deep enough to have penetrated the "restrictive horizon," once the water reaches the pit it is no longer forced to move laterally; it can now move vertically into and through the waste.

Using RESRAD to Assess the Effect of Interflow on Risk Assessment and Remediation

What impact might interflow — if not taken into account — have on risk to humans? To answer this question, we applied an exposure model, RESRAD,

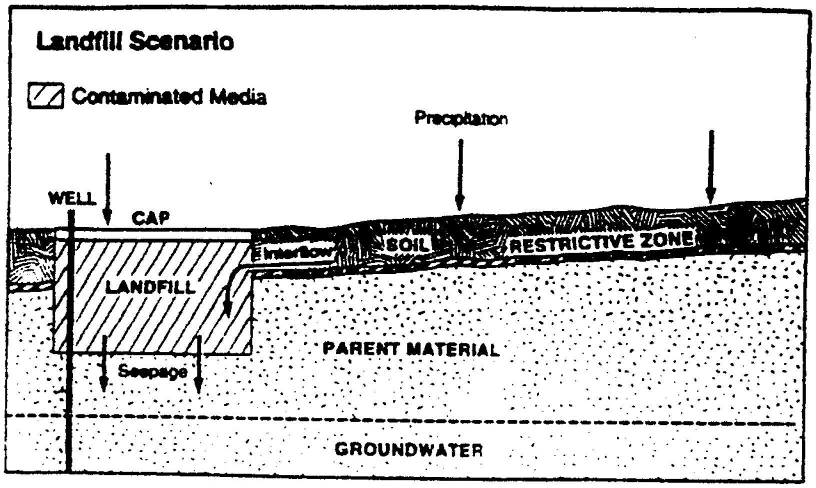

Figure 4. Potential effect of interflow on movement of subsurface contaminants (in a covered landfill).

to a hypothetical site fitting the description in Example II—a covered landfill—under three scenarios: (1) no interflow, (2) moderate amounts of interflow, and (3) high amounts of interflow. Because we are most familiar with the Los Alamos area and have documented the importance of interflow in that area, we have parameterized the model for Los Alamos conditions.

Description and Parameterization of the Model

RESRAD is well suited to our purpose, which is to evaluate the relative risk posed by interflow. The model is currently being used at many contaminated DOE sites (Cheng, Yu, and Zielen, 1991; Cheng and Yu, 1993; Wang, Biwer, and Yu, 1993; Yu et al., 1993a, 1993b). Because many environmental regulations are based on estimated radionuclide doses, RESRAD is designed to predict these doses, and has been used extensively to assess risk to humans posed by radionuclides in the environment (Dorries et al., 1993; Rutz and Green, 1993; Yu et al., 1993b; Espegren, Pierce, and Halford, 1996). As such, it has been both extensively verified (Halliburton NUS Corporation, 1994) and validated against other models (Faillace, Cheng, and Yu, 1994). The hydrologic component, which is quite simple, is not designed to simulate complex hydrologic processes. But none of the currently existing risk assessment models directly simulate interflow; and, moreover, it has never been demonstrated that complex hydrologic models provide more accurate simulations than do simple models — quite the opposite, in fact (Beven, 1989; Blaylock, 1990; Grayson, Moore, and McMahon, 1992). Our strategy for taking into account the additional water contributed by interflow was to modify the precipitation and evapotranspiration terms in RESRAD. Having direct measurements of interflow on the site for which the risk estimates are being made, we are confident that these manipulations are appropriate.

The model assumes that a family is living on the site in question and obtaining drinking water from local groundwater. Because ingestion of drinking water is the exposure route most likely to be affected by interflow (the assumption being that interflow waters, upon entering a waste pit, will move downward to groundwater), we did not consider other potential exposure routes in this modeling exercise. Our specific question was, under the moderate and high interflow scenarios (as contrasted with the no-interflow scenario) to what extent will the rate of groundwater contamination be increased and, thereby, the dose to humans drinking the contaminated water?

RESRAD employs some simple relationships to derive the rate of contaminant movement from the source area to groundwater. The rate at which radionuclides will be leached from the contaminated zone is estimated with a sorption-desorption ion-exchange leaching equation:

$$R_i(t) = L_i \rho_b^{(cz)} A\, T(t)\, S_i(t), \tag{1}$$

where

$R_i(t)$ = release rate for radionuclides (pCi/yr);

L_i = leach rate for radionuclides (yr^{-1});

$\rho_b^{(cz)}$ = bulk density of the contaminated zone (kg/m^3);

A = area of the contaminated zone (m^2);

$T(t)$ = thickness of the contaminated zone at time t (m); and

$S_i(t)$ = average concentration of the ith principal radionuclide in the contaminated zone available for leaching at time t (pCi/kg).

The leach rate (L_i) is estimated as

$$L_i = I / (\theta^{(cz)} T_0 R_{di}), \tag{2}$$

where

I = infiltration rate (m/yr) [water entering unsaturated zone below landfill];

$\theta^{(cz)}$ = volumetric water content of the contaminated zone;

T_0 = initial thickness of the contaminated zone (m); and

R_{di} = retardation factor in the contaminated zone for radionuclide i.

The retardation factor (R_{di}) is the ratio of the average pore water velocity to the radionuclide transport velocity. It is calculated as

$$R_{di} = 1 + [(\rho_b K_{di})/\theta], \tag{3}$$

where

ρ_b = bulk soil density (g/cm^3);

K_{di} = distribution coefficient for the radionuclide i; and

θ = volumetric water content.

The infiltration rate is given by

$$I = (1-C_e)[(1 - C_r)\ P_r],\tag{4}$$

where
C_e = evapotranspiration coefficient.
C_r = runoff coefficient; and
P_r = annual precipitation (m/yr).

Further details on the computational methodology may be found in Yu et al. (1993b).

We parameterized RESRAD to simulate contaminant movement from a small (24 m^2), covered landfill in a ponderosa pine (*Pinus ponderosa*) community within Los Alamos National Laboratory in northern New Mexico. Average annual precipitation at the site is 500 mm. The landfill lies perpendicular to the slope of a hill, creating an upslope contributing area for interflow of about 600 m^2. Depth to the main aquifer may be as great as 300 m, but perched groundwater bodies exist that are much closer to the surface. We have therefore assumed an average depth to groundwater of 50 m.

For the no-interflow scenario, we used the following parameters: annual precipitation (P_r) = 500 mm; runoff coefficient (C_r) = 0.2; and evapotranspiration coefficient (C_e) = 0.99. Per eq. 4, the infiltration rate is 4 mm/yr.

For the two interflow scenarios, we selected amounts of 10 and 20 mm/yr to represent moderate interflow and high interflow, respectively (on the basis of our finding that interflow generally makes up 2 - 5% of the annual water budget—Wilcox et al., 1996). For a contributing upslope area of 600 m^2, these amounts translate to volumes of water entering the landfill of 6 m^3 and 12 m^3, respectively. We then calculated the infiltration rates that would result in these two annual volumes: for the moderate interflow scenario, the infiltration rate is 250 mm/yr (6 m^3/24 m^2); and for the high interflow scenario, the infiltration rate is 500 mm/yr (12 m^3/24 m^2).

Because interflow cannot be simulated directly in RESRAD, we then substituted these infiltration rates into eq. 4 and manipulated other parameters such that the equation yielded those rates. For the moderate interflow scenario, with the runoff coefficient (C_r) maintained at 0.2 and the annual precipitation rate (P_r) at 500 mm, setting the evapotranspiration coefficient (C_e) at 0.375 yielded the desired infiltration rate of 250 mm/yr. For the high interflow scenario, with C_r maintained at 0.2 and C_e at 0.375, we had to increase P_r to 1000 mm/yr to obtain the desired infiltration rate of 500 mm/yr. For the groundwater ingestion route, infiltration rate is the only factor affected by these parameter manipulations. In other words, other calculations in RESRAD are not affected by these changes.

A listing of pertinent RESRAD parameters for each simulation is given in Table 1. We selected the nondispersive flow option in RESRAD for these calculations.

Table 1. RESRAD parameters pertinent to the drinking water exposure route.

RESRAD PARAMETER	No Interflow	Moderate Interflow	High Interflow
	Contaminated Zone		
Area	24 m^2	24 m^2	24 m^2
Thickness	2 m	2 m	2 m
Depth of surface cover	0.5 m	0.5 m	0.5 m
Density of cover material	1.5 g/cm^3	1.5 g/cm^3	1.5 g/cm^3
Porosity	0.4	0.4	0.4
Effective porosity	0.35	0.35	0.35
Hydraulic conductivity	100 m/yr	100 m/yr	100 m/yr
Evapotranspiration coefficient	**0.99**	**0.375**	**0.375**
Runoff coefficient	0.2	0.2	0.2
Precipitation	**0.5 m/yr**	**0.5 m/yr**	**1.0 m/yr**
	Unsaturated Zone		
Thickness	50 m	50 m	50 m
Density	1.6 g/cm^3	1.6 g/cm^3	1.6 g/cm^3
Porosity	0.5	0.5	0.5
Effective porosity	0.4	0.4	0.4
Hydraulic conductivity	470 m/yr	470 m/yr	470 m/yr
	Saturated Zone		
Density	1.5 g/cm^3	1.5 g/cm^3	1.5 g/cm^3
Porosity	0.4	0.4	0.4
Effective porosity	0.35	0.35	0.35
Hydraulic conductivity	100 m/yr	100 m/yr	100 m/yr
Well pump intake depth	10 m	10 m	10 m
Water table drop	0.001 m/yr	0.001 m/yr	0.001 m/yr
Well pumping rate	250 m^3/yr	250 m^3/yr	250 m^3/yr

We estimated the movement of, and subsequent dose from, radionuclides of three elements that behave quite differently in the environment: tritium (^3H), uranium-238 (^{238}U), and plutonium-239/240 ($^{239/240}Pu$). The only radionuclide-specific parameter in the model that directly influences groundwater contamination is K_d (eq. 3) in the contaminated zone. For this parameter, we assumed values of 0 for ^3H, 5 for ^{238}U (site-specific data: LANL, 1995), and 2000 for $^{239/240}Pu$, reflecting their different behaviors: tritium is not adsorbed to soil particles and essentially moves at the same rate as water, whereas ^{238}U and $^{239/240}Pu$ are adsorbed to soil and thus will be transported much more slowly. These radionuclides also have different dose-equivalent conversion factors, with $^3H < {}^{238}U < {}^{239/240}Pu$. We assumed initial total concentrations of 1 pCi/g for each element, with a ^{239}Pu to ^{240}Pu ratio of 99.5% to 0.5%. Subsequent model runs confirmed that dose was linearly related to initial concentrations for these radionuclides: our results, then — provided as (mrem/yr)/(pCi/g) — can be multiplied by site-specific concentrations (pCi/g) to obtain estimates of site-specific dose rates (mrem/yr).

Model Results

Under the no-interflow scenario, dose from ingestion of groundwater was zero for all the radionuclides for the first 100,000 years; there simply was not enough water to move contaminants to groundwater. Under the two interflow scenarios, dose from ingestion of groundwater varied among radionuclides in magnitude and timing according to the amount of simulated interflow (6 m^3 or 12 m^3) (Fig. 5). Simulated dose for all the radionuclides was higher and more quickly delivered under the high-interflow (12-m^3/yr) conditions.

The differences among the simulations are all related to differences in infiltration rates, the radionuclide-dependent sorption/desorption process (the magnitude of which was set with the K_d), and the dose-conversion factors. The transport time to groundwater differed among radionuclides, reflecting the differences in K_d. Tritium had the lowest K_d and the most rapid transport time. Small concentrations of ^{235}U, produced as ^{239}Pu progeny, reached the groundwater at the same time as ^{238}U (RESRAD assumes all radionuclides are in equilibrium with their progeny initially). However, $^{239/240}Pu$ radionuclides did not actually reach the groundwater—because of its very large K_d, expected transport time exceeds 250,000 years.

For each of the radionuclides, the dose resulting from the moderate-interflow scenario is lower than that from the high-interflow scenario because the activity was added to the groundwater in lower concentrations over a greater period of time. For ^3H, the dose from the moderate-interflow scenario is further reduced by the additional physical decay that occurs during the longer transport time.

The highest dose resulted by far from ^{238}U. The ^{238}U-dose exceeded that from ^3H by nearly three orders of magnitude, due to the differences in the dose conversion factors between the radionuclides and the physical decay of the ^3H that occurred during transport. Further, the dose from ^{238}U exceeded

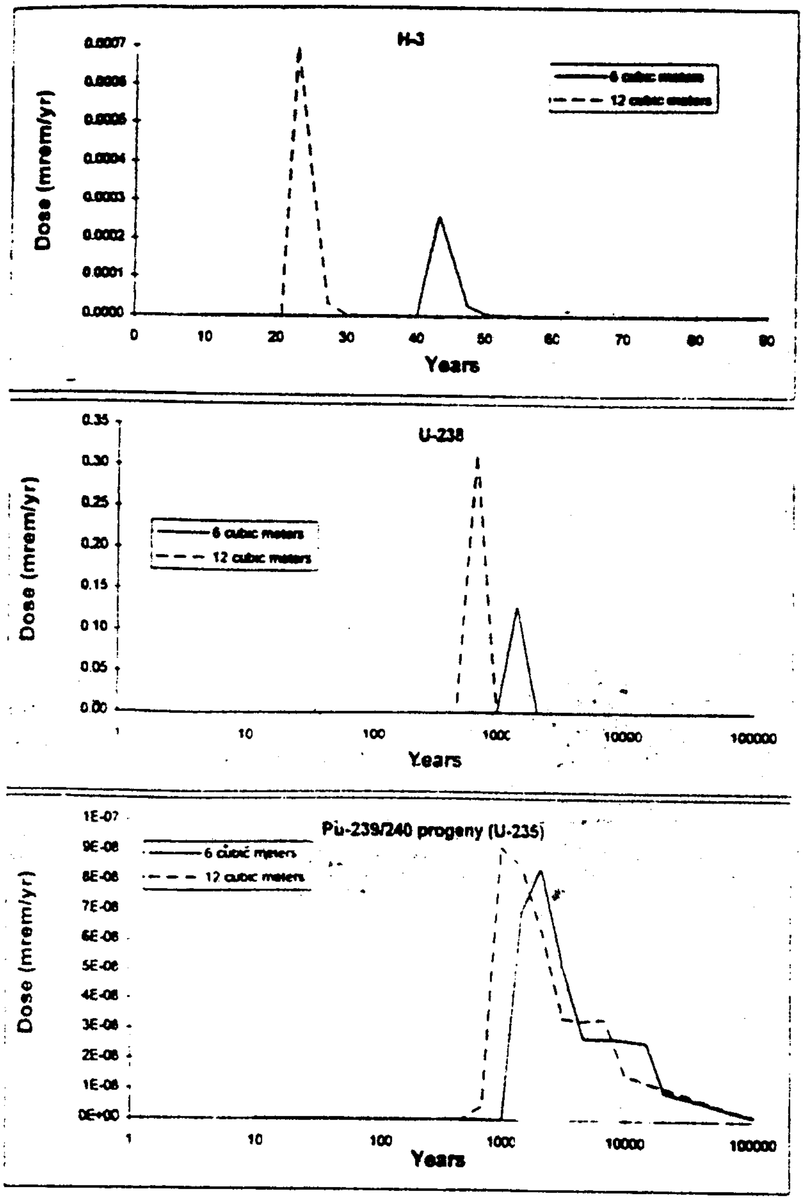

Figure 5. Simulated doses, from ingestion of groundwater, of (a) ^{3}H, (b) ^{238}U, and (c) $^{239/240}Pu$ under conditions of moderate and high interflow.

that from the ^{235}U, produced as ^{239}Pu progeny, by more than seven orders of magnitude because the ^{235}U was present only in extremely small concentrations. These results are in agreement with other modeling assessments of the groundwater contamination from $^{239/240}Pu$ in the Los Alamos area (Hansen and Rogers, 1983).

IMPLICATIONS FOR ENVIRONMENTAL MANAGEMENT

We have shown that interflow, if not recognized, may have unexpected implications for contaminant movement in semiarid regions, both on the surface and in the subsurface environment. Our simulations, although obviously a simplification of the actual hydrological processes involved, do demonstrate the potential impact of interflow on risk assessment and, thereby, on site characterization and the selection of a landfill remediation strategy. Many factors, including interflow amounts, depth to groundwater, and permeability of underlying media, will determine the extent to which interflow will influence the movement of contaminants at a specific site. But it is clear that consideration of interflow improves risk assessments and thereby enables more cost-effective remediation at Los Alamos. Moreover, although the examples we have presented deal with the movement of radionuclides, hazardous chemicals would be similarly affected by interflow (indeed, the RESRAD model is being modified to address risk assessment for hazardous chemicals — Cheng and Yu, 1993; Cheng et al., 1993). In like manner, although we have focused on human risk assessment, the principles demonstrated apply equally to ecological risk assessment.

We argue, then, that interflow is a potentially very important hydrologic process that currently is not being considered in risk assessments. We recommend the following actions for incorporating interflow into risk assessments and into remediation decisions:

1. Sites receiving more than 450 mm/yr of rainfall should be investigated specifically for the occurrence of interflow.

2. Risk-assessment models, such as RESRAD, should be modified to incorporate interflow. Simple mathematical representations of interflow have been developed (e.g., Flanagan and Nearing, 1995) that could readily be incorporated into RESRAD.

3. For sites at which interflow has been shown to be important, remediation designs, such as landfill covers, should include systems for routing water away from contaminated zones.

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