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RESUSPENSION AND REDISTRIBUTION OF PLUTONIUM IN SOILS*

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ABSTRACT

Formulae are derived which parameterize dust and plutonium fluxes in terms of micrometeorological and soil erodibility parameters. Interim, simple models utilizing the time-dependent resuspension factor and mass-loading approaches are derived which may be used to predict the average concentration of resuspended plutonium. Intensive measurements of the resuspension of plutonium at the USAEC's Nevada Test Site are summarized in terms of a resuspension rate through the use of Healy's model of transport and diffusion. Values of the resuspension rate varied from 2.7×10^{-12} to 4.8×10^{-10} sec^{-1}. The relationship between the resuspension rate and the historical resuspension factor is develcped. The various models and formulations used are shown to be internally consistent for this experimental site.

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INTRODUCTION

It is appropriate that a discussion of resuspension be included in this Symposium because concern about the health implications for man of plutonium releases has been the most important motivation for the study of the resuspension of contaminants from native surfaces. It was recognized many years ago that the problems associated with the release of plutonium to the environment are rather different from those of other prominent radionuclides of the nuclear age such as ^{90}Sr, ^{131}I, and ^{137}Cs. The plutonium radionuclides of primary concern, ^{238}Pu, ^{239}Pu, and ^{240}Pu, decay with negligible emission of high-energy β's and γ's and therefore are not of concern from the standpoint of external exposure unless direct skin contamination occurs. Plutonium radionuclides are also characterized by very low transfer through food-chains to man, and have one of the lowest known absorption rates from the human GI tract to blood.

The probability of deposition in the pulmonary region of the lung, however, is governed solely by the aerodynamic particle size of the contaminant. It is generally accepted, therefore, that the most significant environmental pathway of plutonium to man is inhalation, either directly following an airborne release or from the resuspension of surface deposits. Following lung deposition, the high-energy α particles emitted and the long retention times combine to make the long-lived, alpha-emitting radionuclides of plutonium and other actinide elements orders of magnitude more toxic than other radionuclides.

Several authors have attempted to model the resuspension process with the goal of deriving standards for plutonium in soil (Langham, 1966; Kathren, 1968; Healy, 1974; Anspaugh, et al., 1974a). The models which have been used, however, are rather simplistic and do not consider many variables which undoubtedly affect the resuspension process. A desirable general model would quantitate the resuspension process itself as an upward flux of contaminant aerosol as a function of the soil contamination per unit area, the soil surface characteristics, vegetation cover, micrometeorological parameters, and time since deposition. The latter factor is necessary to include the processes which alter the physical and chemical state of the contaminant, attachment to host soil particles, downward migration through the soil profile by physical and chemical processes, and loss from the site. This parameterization of the flux should then be coupled with a model of atmospheric transport and diffusion which accounts for the areal distribution of the contaminant on the soil surface. Finally, some method of parameterizing the effects of artificial disturbances in greatly increasing the resuspension flux must be derived.

While no such general model presently exists, considerable progress has been made toward its achievement. The purpose of this paper is to review this progress, and also to present two of the simple, interim models which may be used to predict the average airborne concentration of resuspended contaminant until a more general model is available. A complete review of the resuspension literature is not presented; the topics chosen for inclusion were highly influenced by the authors' own

experiments and opinions regarding the applicability of the work of others toward achieving a general model of the resuspension process.

SOIL EROSION STUDIES

A great deal of effort has been devoted to the qualitative and quantitative study of soil erosion and sand movement; the early work has been summarized by Free (1911), Bagnold (1954), and Chepil and Woodruff (1963). These studies demonstrated that wind-driven soil movement occurs by three processes: surface creep, saltation, and suspension. The most important process in initiating and sustaining soil movement is saltation. Particles moving in saltation are observed to abruptly rise nearly vertically into the airstream where they obtain horizontal momentum from the wind and then impinge back upon the soil surface due to gravity. Particles moving in saltation have diameters of about 50 to 500 μm; they must be small enough to move by direct wind action but large enough to have settling velocities higher than the upward eddy velocity of the wind. Upon impingement, saltating particles may bounce upward again or may cause other particles to move in saltation, surface creep, or suspension. Particles moving in surface creep are relatively large, but < 2 mm in diameter and merely roll along the surface. Particles moving in suspension are small, certainly < 100 μm in diameter and have settling velocities less than the turbulent eddy velocities of the wind.

Particles moving in suspension are obviously of the greatest interest in terms of the resuspension of contaminants because only these smaller particles can be transported any significant distance or be deposited in the pulmonary region of the respiratory tract. However,

by far the greatest mass of eroding soil or sand moves by saltation or surface creep, and even particles moving by saltation rarely rise more than a few tenths of a meter above the soil surface. There is therefore no appreciable vertical flux compared to the horizontal flow, and quantitative studies on soil erosion have typically sought to parameterize the horizontal flow with emphasis on the larger particles.

Bagnold (1954) derived an expression for the horizontal flow of sand moving in saltation which was largely based upon the analysis of wind speed gradients. Normally, the increase in wind speed with height is given by

$$u = \frac{u_*}{k} \ln \frac{z}{z_o} \qquad (1)$$

where u = Horizontal wind speed at height z

u_* = Friction velocity = $\sqrt{\tau/\rho}$

τ = Momentum flux

ρ = Air density

z_o = Roughness length = Height at which u is zero.

Bagnold observed during wind tunnel studies over mobile surfaces of loose sand, that while sand is moving in saltation, the usual wind speed profile is altered to the form:

$$u = \frac{u_*'}{k} \ln \frac{z}{z'} + u_t \ , \qquad (2)$$

where u_*' = Friction velocity while saltation is occurring

u_t = Impact threshold velocity

z' = Height where $u = u_t$ ($z' \gg z_o$).

Bagnold then made two assumptions. The first was that, while saltation is occuring, the momentum flux τ' is entirely transmitted to the saltating particles and is equal to the momentum flux produced by the impact of the saltating particles when they impinge on the ground. This results in the expression:

$$q_s = \frac{\rho}{g} u_*'^2 w \ , \tag{3}$$

where q_s = Horizontal sand flow, mass per unit width per unit time

g = Gravitational acceleration

w = Initial vertical speed of saltating particles.

The second assumption was that w is proportional to u_*', which gives

$$q_s = B \frac{\rho}{g} u_*'^3 \tag{4}$$

where B is a constant* for a given sand size, or mixture of sizes. This functional dependence of q_s on u_*' was confirmed by measurements in a wind tunnel and in the Libyan Desert over dune sand. The sand flow can also be expressed in terms of the wind velocity at any height by combining Equations 2 and 4.

The threshold velocity in Equation 2 is not the threshold to initiate saltation movement, but the threshold velocity necessary to

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*Bagnold found a value for B of 0.8 for nearly uniform sand with an average particle diameter of 250 μm; the other parameters were expressed in cgs units.

sustain saltation. This impact threshold decreases as the square root of particle diameter. The threshold velocity necessary to initiate sand movement is greater than the impact threshold velocity and was found to have a minimum for particles 80 μm in diameter. For mixtures of sand, therefore, particles in the size range of about 50 to 150 μm play an important role in initiating the movement of both smaller and larger particles.

It is important to point out, however, that Equation 4 is only valid for the maximum value of q_s for a given u_*', as it is based upon the assumption that all of the momentum flux is transferred to particles moving in saltation. This assumption is probably rarely valid for any surfaces except loose sand, and actual values of q_s should generally be much lower than Equation 4 would predict.

W. S. Chepil greatly extended the quantitation of the factors influencing soil erosion on agricultural fields in studies spanning 30 years, and the results have been summarized by Chepil and Woodruff (1963). These studies confirmed the basic relationship of Equation 4 for agricultural soils, not only for saltation but also for the horizontal flow of mass moving by surface creep and saltation as well. A great many factors were found to influence the value of B, however.

The dependency of soil erosion on the most significant factors for unsheltered native fields with a long fetch is expressed by the equation (Chepil and Woodruff, 1959):

$$X = 400 \frac{I}{(RK)^{1.26}} \qquad (5)$$

where X = Wind tunnel erodibility, tons per acre

I = Soil erodibility index based on percentage of soil mass associated with particles of diameter = >840 μm

R = Amount of crop residue, pounds per acre

K = Ridge roughness equivalent, inches.

The term "wind tunnel erodibility" is used because the soil erodibility is normalized to a wind tunnel measurement of the erodibility of a soil with 60% of its mass associated with particles of diameter = >840 μm; this fraction of the soil is termed nonerodible. The soil erodibility index is a significant determinant of the erodibility; it varies from 0.02 to 1000 for 90% and 1% nonerodible soil compositions, respectively.

Although these expressions are useful in defining the total erodibility of the soil where a contaminant is deposited, they are not directly useful in defining the resuspension or vertical flux of respirable particles of the contaminant. Recently, however, significant progress has been attained in relating the vertical flux of aerosols to conventional measurements of soil erosion. Gillette, et al. (1972) and Gillette (1974) have reported simultaneous observations of vertical flux and horizontal flow in saltation. Vertical flux was calculated on the basis of aerosol (diameter = <20 μm) concentration measurements at two heights according to the equation:

$$F_A = -K_A \rho \frac{\partial n}{\partial z} \qquad (6)$$

where F_A = Aerosol flux, number of particles per unit area per unit time

K_A = Coefficient of exchange for aerosols

n = Number of particles per unit mass of air.

The assumption was made that K_A is equal to the eddy viscosity K which is given by*

$$K = k\ z\ u_*. \tag{7}$$

Shinn, et al. (1974) introduced a further simplification by expressing Equation 6 in a modified form:

$$F = -K \frac{\partial X}{\partial z} \tag{8}$$

where F = Aerosol flux, mass per unit area per unit time

X = Concentration of aerosol at height z, mass per unit volume of air.

Further, it has been documented (Chepil and Woodruff, 1957; Shinn, et al., 1974) that during periods of wind speed sufficient to produce a vertical aerosol flux, the concentration of aerosol at height z normalized to the concentration at a height of 1 m appears universally to exhibit a power law distribution. Data from several such measurements are shown in Fig. 1. An expression for the concentration gradient is therefore easily derived, and is

$$\frac{\partial X}{\partial z} = p \frac{X}{z} \tag{9}$$

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*The notation u_* rather than u_*' is used from hereon because the wind speed profiles measured during our experiments have conformed to Equation 1 rather than Equation 2.

where p is the power of z; measured values of p vary from -0.35 to -0.25. By combining Equations 7, 8, and 9; a simplified expression is derived for the aerosol flux:

$$F = -p\ k\ u_* \chi. \tag{10}$$

The vertical flux is therefore a function of height, but because χ varies only about ±20% from a height of 0.5 to 2 m, it is convenient for our purposes to specify the flux at a height of 1 m.

The dependency of flux upon micrometeorological parameters therefore simplifies to the parameterization of aerosol concentration at a reference height. Shinn, et al. (1974) have also investigated the dependency of χ on u_* at two sites; one in Texas over an eroding agricultural field and the GMX site at the USAEC's Nevada Test Site. The soil surface at the latter location exhibits a typical desert pavement protected by native vegetation and is only slightly erodible. The results of these studies are shown in Fig. 2. Both sites are similar in that the dust concentration is well correlated with a power of u_*:

$$\chi = \chi_o \left(\frac{u_*}{u_o}\right)^{\gamma} \tag{11}$$

where u_o is a reference friction velocity of 1 m sec^{-1} and χ_o is a reference concentration derived from the fit.

The most striking feature of the data is the marked difference in the dependency of χ on u_* at the two sites. For the eroding agricultural field the power is 6.38 whereas for the desert pavement site the power is 2.09. The latter value is essentially what we would expect if the aerosol flux were to exhibit the same dependency on u_* as

does the horizontal flow (Equation 4). A similar dependency of F_A on a higher power of u_* was also observed by Gillette (1974) for loamy soils. He postulated that the effect is due to an additional dependency on u_* of the production of suspendible particles due to the breakage of soil aggregates by the sandblasting effect of saltating particles. The dependency of χ on a lower power of u_* at the GMX site is consistent with our experimental observation that saltation is negligible at this location (Anspaugh, et al., 1974c).

Equations 10 and 11 have been combined and reduced to the form (Shinn, et al., 1974):

$$F = F_o \left(\frac{u_*}{u_o}\right)^{\gamma+1}, \qquad (12)$$

where F_o is a reference flux. This formulation, referred to as the "Gillette and Shinn model", has been expanded further to relate both F_o and γ to Chepil's soil erodibility index, I. Because only a limited number of measurements are presently available, the dependencies of F_o and γ on I are necessarily tentative; they are shown in Fig. 3. If these tentative relationships are verified, they offer a valuable link between the resuspension problem and the large body of information available on the factors which influence soil erosion.

This is particularly true for aged deposits of contaminant where it may be reasonable to assume that the contaminant is intimately associated with the host material in the soil surface. Then the vertical flux of contaminant may be predicted by the dust flux calculated using Equation 12 multiplied by the amount of contaminant per unit mass of soil surface material. This information may then be combined with a

suitable model of atmospheric transport and diffusion which also considers the areal distribution of the contaminant to calculate airborne concentrations of resuspended contaminant both within and outside the contaminated area.

RADIONUCLIDE RESUSPENSION STUDIES

The above approcach, while very promising in its possible general application, needs further development to include the situation immediately following a contaminating event when it is not reasonable to assume that the contaminant is intimately mixed with the host soil particles. There are, however, several experimental measurements of the airborne concentration of resuspended radionuclides immediately or soon after the contaminating event; several of these direct measurements were undertaken for the specific purpose of estimating the potential hazards associated with the accidental release of plutonium from accidents involving nuclear weapons.

It is unfortunate that most of these experiments are poorly documented and a comprehensive analysis of the results has not been possible. Several general conclusions may, however, be drawn from these experiments: 1) The airborne concentration of radioactive aerosols produced by explosions declines rapidly immediately after the detonation and is lower by a factor of 100 to 1000 by 100 hr after the detonation (Shreve, 1958; Anspaugh, et al., 1973); 2) After the initial rapid decline, a further decrease with time is noted with half-times of about 5 weeks observed during the first 6 to 20 weeks following release (Langham, 1971; Larson, et al., 1966; Wilson, et al.; Anspaugh, et al., 1973), but a

longer half-time of decrease of about 10 weeks was observed by Anspaugh, et al. (1973) during one experiment conducted 12 to 40 weeks post release and Sehmel and Orgill (1974) reported a half-time of about 9 months after a 10 year old source had been artificially disturbed; 3) Such long-term decreases of the concentration of resuspended radionuclides with time are not due to an appreciable net loss of radionuclide from the area, but to a "weathering" process whereby the contaminant becomes less erodible (Shreve, 1958; Olafson and Larson, 1961; Stewart, 1964; Larson, et al., 1966); 4) Areas which were contaminated 10 to 20 years previously are still significant sources of resuspended radioactive particulates (Anspaugh, et al., 1974b; 1974c; Sehmel and Orgill, 1974); 5) For explosion-produced sources, the quotient of air concentration of resuspended contaminant divided by ground deposition increases with distance from the source (Wilson, et al., 1960; Larson, et al., 1966; Anspaugh, et al., 1971) which may be partly due to the decrease in particle size of the contaminant which is deposited at further distances (Shreve, 1958); 6) Short-term, order of magnitude fluctuations of the airborne concentration of resuspended radionuclides are frequently observed presumably due to changes in meteorological conditions although functional relationships are not well-defined, especially for freshly deposited sources (Shreve, 1958; Wilson, et al., 1960; Olafson and Larson, 1961; Larson, et al., 1966; Anspaugh, et al., 1973); 7) Artificial disturbances in a contaminated environment can also produce orders of magnitude increases in the airborne concentration of contaminant (Mork, 1970; Sehmel, 1973; Sehmel and Orgill, 1974); and 8) Measurements

of the particle size distributions of resuspended, plutonium-containing aerosols indicate that the fraction which would undergo pulmonary deposition is about 0.15 to 0.25 (Wilson, et al., 1960; Volchok and Knuth, 1972; Anspaugh, et al., 1974c).

INTERIM RESUSPENSION MODELS

Time-dependent resuspension factor approach

The earliest efforts to define the magnitude of the resuspension pathway used the empirical approach of calculating a resuspension factor, S_f, which Langham (1971) defined as

$$S_f = \frac{\text{Resuspended air concentration (activity / m}^3)}{\text{Surface deposition (activity / m}^2)}. \qquad (13)$$

This simple approach neglects many factors which undoubtedly are important in determining resuspension, but the available direct measurements of the resuspension of radioactive aerosols immediately and soon after deposition have no common basis except the measurement of the two parameters defining S_f.

Stewart (1964) and Mishima (1964) have tabulated values of S_f from a variety of experiments. For native environments soon after the contaminating event, calculated values of S_f range from 10^{-7} to 10^{-3} m^{-1}, with only a few values greater than 10^{-4} m^{-1} which were associated with artificial disturbances. Many of the lower values were derived from measurements made in close proximity to explosive detonation points (Wilson, et al., 1960; Anspaugh, et al., 1971) where very large particles could have been deposited.

A convenient way to model the airborne concentration of resuspended contaminant over long periods of time is to make the resuspension factor

a function of time to account for the observed decrease in air concentration which has been noted to occur in the absence of a significant net loss of the deposited contaminant. Conceptionally, it would be preferable to define a time-dependent fraction of the total deposition which is available for resuspension. However, there is no realistic way in which such a fraction can be experimentally determined, so this approach will be avoided for the present purpose. With the time dependency inherent in the resuspension factor, it follows that the average airborne concentration, $\overline{X}$, of resuspended contaminant will be given by

$$\overline{X}(t) = S_f(t)\, S_A \qquad (14)$$

where S_A is the total amount of contaminant deposited per unit area. S_A is therefore considered a constant although the actual distribution of the contaminant with soil depth will change with time.

Langham (1966; 1971) and Kathren (1968) have each formulated predictive resuspension models which, when expressed in the above format, give the following time dependency:

$$S_f(t) = S_f(0)\, \exp(-\lambda t) \qquad (15)$$

with values of λ corresponding to half-times of 35 and 45 days, respectively. Such a formulation appears to simulate reasonably well the available observations for time periods up to several weeks post deposition. After a few years, however, such a formulation underestimates by many orders of magnitude the airborne concentration of resuspended plutonium measured over aged sources (Olafson and Larson, 1961; Volchok, 1971; Anspaugh, et al., 1974b). For example, the Langham and Kathren models

would predict values for $S_f(t)$ of 10^{-38} and 10^{-29} m^{-1} respectively 10 years after a contaminating event whereas an average value determined from 236 individual air concentration measurements at a location contaminated with plutonium 17 years previously was found to be 10^{-9} m^{-1} (Anspaugh, et al., 1974b).

We have derived a different formulation of the time dependency of the resuspension factor which more accurately reflects the resuspension process as it is observed in the proximity of aged sources. This model was empirically derived to conform to the following constraints: 1) The apparent half-time of decrease during the first 10 weeks should approximate a value of 5 weeks and should approximately double over the next 30 weeks; 2) The initial resuspension factor should be 10^{-4} m^{-1}; and 3) The resuspension factor 17 years after the contaminating event should approximate 10^{-9} m^{-1}.

A simple model which closely approximates these constraints is

$$S_f = 10^{-4} \exp(-\lambda\sqrt{t})\ m^{-1} + 10^{-9}\ m^{-1} \qquad (16)$$

where λ is 0.15 / $\sqrt{\text{day}}$. The second term was added based upon the assumption that there may be no further measurable decrease in the resuspension process after 17 years which is the longest period post deposition for which measurements have been reported. This was deemed appropriate because such a "model" was derived empirically to simulate experimental measurements, and contains no fundamental understanding of the resuspension process. A graphical representation of this model, both with and without the second term is shown in Fig. 4; the equations used by Langham and Kathren are also shown for comparison.

It is emphasized that Equation 16 was derived from a composite of results from numerous experiments; there have not been measurements at any individual source over such long time-periods. The tentative relationships shown in Fig. 3 indicate that the use of composite results from differing sites may be misleading; however, many of the measurements of resuspended radionuclides have been made over grossly similar desert terrain.

There is an obvious need to model the "weathering" process on a fundamental basis. Several authors have presented data (Beck, 1966; Hogowski and Tamura, 1970; Romney, et al., 1970) which indicate that radionuclides deposited on the earth surface in either solution or particulate form penetrate within a few months to depths of more than 1 cm. Beck (1966) has noted that the distribution with depth is frequently well described by an exponential function:

$$S_m(z) = S_m(0) \exp(-\alpha z) , \tag{17}$$

where $S_m(z)$ is the activity per unit mass of soil at depth z. The total amount of contaminant per unit area, S_A, is then given by

$$\begin{aligned} S_A &= \rho \int_0^\infty S_m(z) \exp(-\alpha z)\, dz \\ &= \frac{\rho}{\alpha} S_m(0). \end{aligned} \tag{18}$$

If there is no net loss of contaminant from the area, S_A is a constant and the value of α / ρ may provide a rough index of the fraction of the total surface deposit which is available for resuspension. Values of the parameter α will obviously vary with time, and are

strongly influenced by other factors as well. As an illustration, the distribution of plutonium with soil depth is shown in Fig. 5 for three locations with differing sources of plutonium contamination. At the USAEC's Nevada Test Site (NTS), the plutonium source was a non-nuclear detonation; at Rocky Flats the probable source was leakage from storage drums (Krey and Hardy, 1970); at Livermore the source was world-wide fallout (Gudiksen, et al., 1972; 1973). The local climatic conditions and the physicochemical features of the soil as well as those of the plutonium source contribute to this substantial difference in the distribution with depth. It is possible that the parameters a and soil erodibility index, I, are also correlated, as purely mechanical mixing may be a significant factor in moving surface deposited radioactivity to lower depths.

Mass-loading approach

The previously cited observation, that radioactivity is commonly found at depths of several cm a few months after its deposition on the soil surface, implies that an intimate association with the host soil is achieved rapidly. As discussed above, Equation 12 and the activity per unit mass of surface soil may be used to predict the vertical flux of the contaminant. However, a much simpler method of predicting the average concentration of resuspended contaminant is simply to multiply the surface soil concentration by the average mass loading of the atmosphere. In the absence of data for a specific site, Anspaugh, et al. (1974b) have suggested the use of 100 $\mu g/m^3$ for predictive purposes. The choice of this value is partly based upon measurements of particulate concentration in 1966 reported by NAPCA (1968) for 30 nonurban locations.

Annual arithmetic averages* varied from 9 to 79 $\mu g/m^3$ with a mean for all 30 locations of 38 $\mu g/m^3$.

Several measurements are available to check the accuracy of this predictive approach, and are tabulated in Table 1. The agreement between the predicted and measured values is generally excellent.

AN INTEGRATED METHOD OF ASSESSING RESUSPENSION

Intensive studies on the resuspension of $^{239,240}Pu$ are currently being conducted at the GMX site of the USAEC's Nevada Test Site, and recent results are summarized by Anspaugh, et al. (1974c). The results of Shinn, et al. (1974) at this site indicated a parameterization of dust concentration with $u_*^{2.09}$ and, with the assumption of a power law distribution of concentration with height, a dust flux proportional to $u_*^{3.09}$. For a sampling location close to the center of the contaminated area, a correlation of resuspended Pu concentration with u_*^{γ} was also found with $\gamma = 2.2 \pm 0.6$. However, at a sampling location on the edge of the contaminated area, the results were more widely scattered and a somewhat better fit to the concentration data was achieved with $\gamma = 4$.

An effort was therefore made to analyze the data using a method whereby the effect of the geometrical configuration of the source could be eliminated. Healy (1974) has provided such a method by deriving a model of transport and diffusion which is based upon the specific source

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*Short-term fluctuations due to duststorms may be extreme. Hagen and Woodruff (1973) indicate that there were about 14 days per year averaged over 10 Great Plains States during the 1950's when the mass loading was ~ 10 mg/m^3.

geometry of the GMX site. The output of Healy's calculations is a set of values of the form:

$$\frac{\chi(r)\ \bar{u}}{K_h\ \Omega_p} = H(r, \Delta\theta, Ri) \qquad (19)$$

where $\chi(r)$ = Pu concentration at distance r from the center of the source

$\bar{u}$ = Average wind speed

K_h = Resuspension rate, sec^{-1}

Ω_p = Peak activity per unit area

H = Calculated result which varies as a function of distance, r; deviation of wind direction from pattern (and sampling) center-line, $\Delta\theta$; and atmospheric stability as determined by the calculated value of the Richardson number, Ri.

This formulation introduces the concept of a resuspension rate, K_h, to parameterize the resuspension process, and also eliminates the effect of source geometry in calculating values of K_h. The product $K_h \cdot \Omega_p$ is a flux with units of Ci m^{-2} sec^{-1} if Ω_p is expressed in Ci m^{-2}. It is therefore analogous to the dust flux parameterization derived by Shinn, *et al.* (1974); and, for the GMX site, we may anticipate that K_h should be proportional to u_*^3. Further, if we assume an infinite source and equate the two fluxes ($K_h \cdot \Omega_p$ and Equation 10), K_h may be related to the resuspension factor:

$$K_h = \frac{-\rho\ k\ u_*\ \chi}{\Omega} = -\rho\ k\ u_*\ S_f \qquad (20)$$

The meteorological measurements associated with the sampling periods at the location on the edge of the contaminated area were used to calculate values of H from Healy's model. These values were then combined

with the measurements of X and a measured value for Ω_p of 630 μCi/m^2 to derive values of the resuspension rate. The calculated values vary from 2.7×10^{-12} sec^{-1} to 4.8×10^{-10} sec^{-1}; the top half of Fig. 6 is a histogram of these values. They are considerably lower than those tabulated by Healy for other experiments which vary from 10^{-10} to 10^{-6} sec^{-1}. This is, however, consistent with the long time since deposition of about 20 years when the measurements were made. As anticipated, there is a strong correlation between K_h and u_*^3 with a confidence level of >99.95%. The lower half of Fig. 6 is a histogram of the ratio K_h/u_*^3. The variance in this ratio is greatly reduced compared to the variance in K_h, and the distribution of the ratio approaches that of a lognormal distribution. The net result is a promising parameterization of the resuspension process at the GMX site which is site-geometry independent.

It is also of interest to calculate values of the resuspension factor, S_f, using the measured values of u_*, the previously calculated values of K_h, and values for p and k of - 0.25 and 0.4, respectively. The calculated values of S_f vary from 9.1×10^{-11} to 5.4×10^{-9} m^{-1}; their geometric and arithmetic averages are 2.9×10^{-10} and 6.8×10^{-10} m^{-1}, respectively. We have previously published values of the resuspension factor for this site based upon calculations using Equation 13 (Anspaugh, et al., 1974b). For measurements near the center of the contaminated area, the result was 3×10^{-10} m^{-1}; results at the edge of the contaminated area were 2×10^{-9} m^{-1}.

CONCLUSIONS

The internal consistency of the dust flux measurements, resuspension factor measurements, and resuspension rate calculations using Healy's model of transport and diffusion provides strong support for the formulations developed above. Additional effort, however, will be required to document more adequately the "weathering" process and the relationship of the flux parameters to the soil surface characteristics which have used by soil scientists to parameterize soil erosion.

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Table 1. A comparison of observed and predicted air concentrations based upon a simple mass loading model. The predicted values were calculated by multiplying average values of soil concentration by an assumed mass loading of 100 $\mu g/m^3$. Measured values of air concentration are averages of many individual measurements.

Location, etc.	Nuclide	Air Concentration	
		Predicted	Measured
G.X site, USAEC Nevada Test Site[a]			
NE, 1971-1972	^{239}Pu	7200 aCi/m^3	6600 aCi/m^3
GZ, 1972, 2 weeks	^{239}Pu	120 fCi/m^3	23 fCi/m^3
Lawrence Livermore Laboratory			
1971[b]	^{236}U	150 pg/m^3	52 pg/m^3
1972[c]	^{238}U	150 pg/m^3	100 pg/m^3
1973[d]	^{238}U	150 pg/m^3	86 pg/m^3
1973[d]	^{40}K	1000 aCi/m^3	980 aCi/m^3
Argonne National Laboratory[e]			
1972	^{232}Th	320 pg/m^3	240 pg/m^3
1972	^{nat}U	215 pg/m^3	170 pg/m^3
Sutton, England[f]			
1967-1968	^{nat}U	110 pg/m^3	62 pg/m^3

[a] Anspaugh, et al. (1974b).

[b] Gudiksen, et al. (1972).

[c] Gudiksen, et al. (1973).

[d] Silver, et al. (1974).

[e] Sedlet, et al. (1973).

[f] Hamilton (1970).

FIGURE CAPTIONS

Fig. 1. Normalized profiles of suspended dust concentration with height. Data from Kansas and Colorado (Chepil and Woodruff, 1957) and from Texas and Nevada (Shinn, et al., 1974) exhibit an universal decrease with a power of the height under strong wind speed conditions. The solid line describes a power of -0.25; the dashed line a power of -0.35.

Fig. 2. The empirical dependency of suspended dust concentration on u_*. Results for an eroding agricultural field in Texas are contrasted with measurements over a desert pavement in Nevada.

Fig. 3. Tentative parameterization of the variables that determine vertical plutonium and dust fluxes as a function of the soil erodibility index. F_o is a reference flux with $u_* = u_o = 1$ m sec^{-1} and $\gamma+1$ is the power of the friction velocity, u_*.

Fig. 4. A graphical representation of several time-dependent resuspension factor models. The two curves on the far left represent the models formulated by Langham (1966) and Kathren (1968). The upper two curves represent models described in this paper, both with and without a constant term of 10^{-9} m^{-1}. The hatched area indicates values of S_f recently measured at the GMX site at NTS (Anspaugh, et al., 1974b).

Fig. 5. A graphical representation of normalized plutonium profiles with depth in soil at three sites with differing sources of plutonium, climatic factors, and physicochemical characteristics of soil.

Fig. 6. Histograms of values of the resuspension rate, K_h, and K_h/u_*^3 derived from measured concentrations of plutonium resuspended from the GMX site at NTS and Healy's (1974) model of transport and diffusion. Both K_h and u_*^3 should be directly related to the vertical plutonium flux, and their quotient has substantially less variance than do the values of K_h themselves.

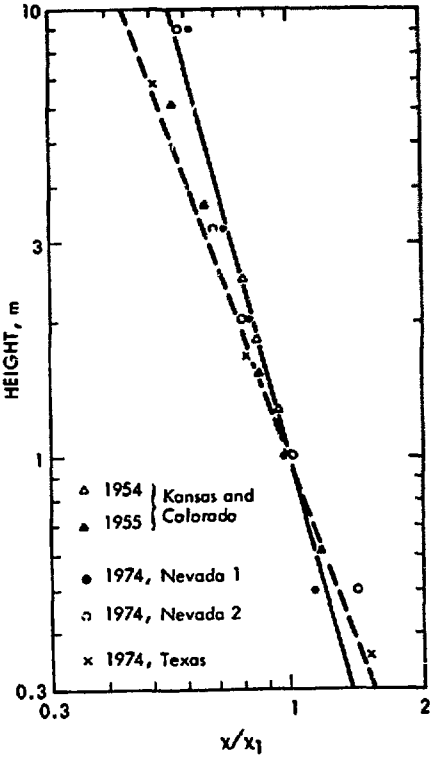

Fig. 1
Anspaugh, et al.

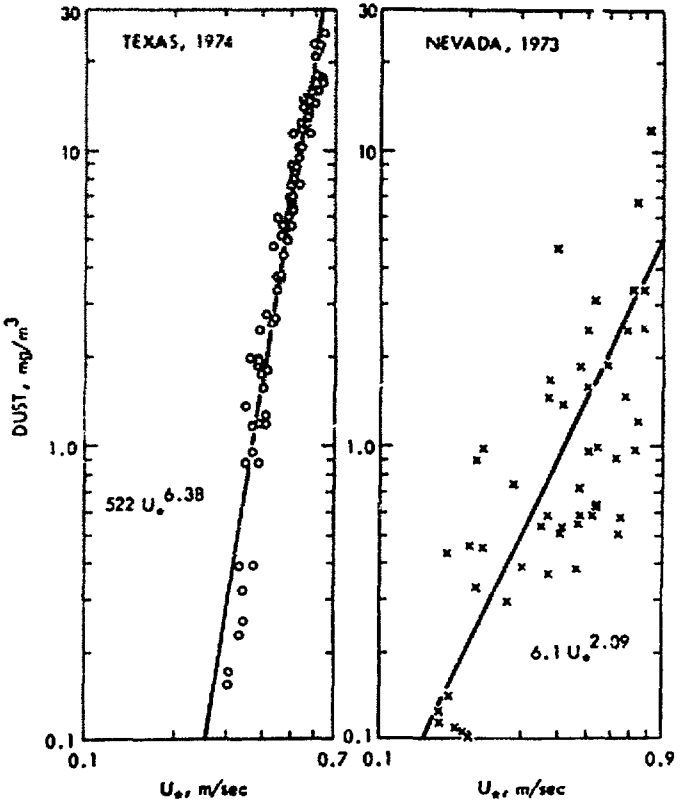

Fig. 2
Anspaugh, et al.

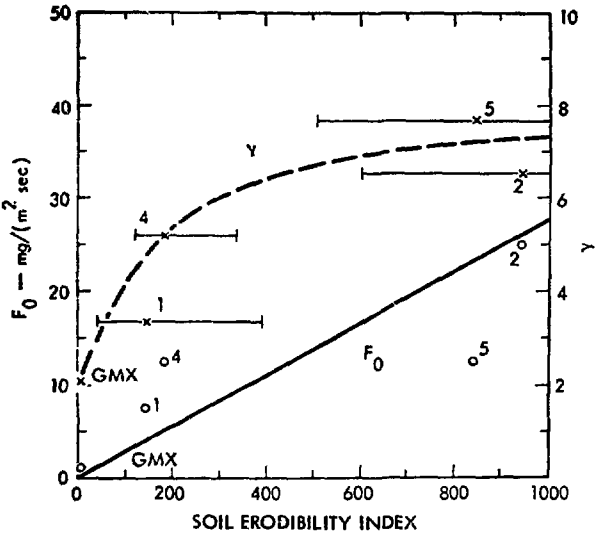

Fig. 3
Anspaugh, et al.

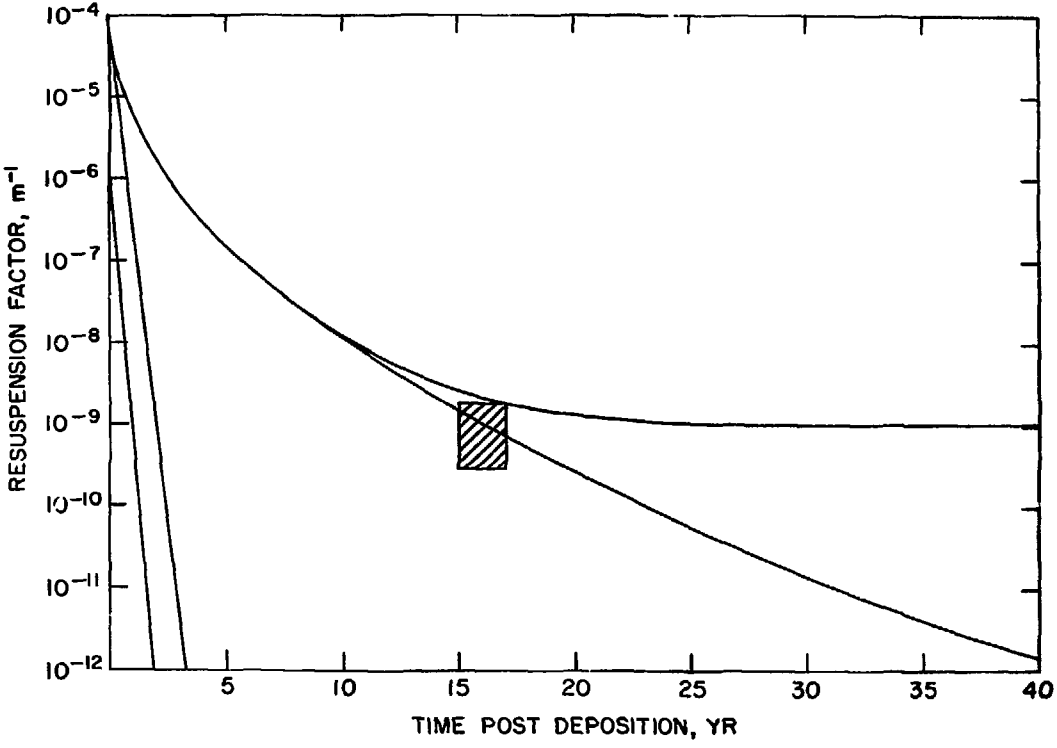

Fig. 1

Anspaugh, et al.

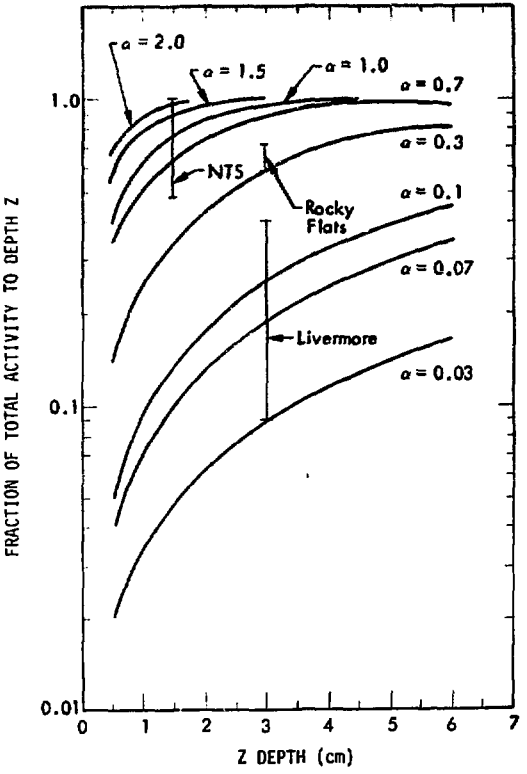

Fig. 5
Anspaugh, et al.

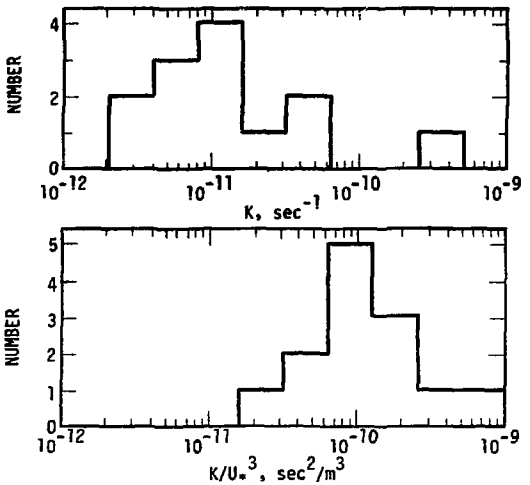

Fig. 6

Anspaugh, et al.