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^{137}Cs Fallout Depth Distributions in Forest Versus Field Sites: Implications for External Gamma Dose Rates

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ABSTRACT

The depth profile of ^{137}Cs fallout in soil from atmospheric nuclear weapons tests was measured at neighboring field and forest areas for seven sites in the northeastern US. The inferred dose rates in air at 1 m above the ground per unit inventory of ^{137}Cs averaged a factor of 1·8 higher in the forest as compared to the field areas. Calculations indicate that the dose rate in forest areas would be a factor of four higher than that over deeply plowed land. Based on a limited set of historical measurements made since 1972, it appears that the dose rate per unit inventory in both field and forest areas has more or less stabilized after a sharp decrease following deposition events in the early 1960s. Estimated dose commitments for various land types are compared to the value suggested by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) and the implications with respect to certain population groups are discussed. The findings may have application in estimating future external doses from deposited ^{137}Cs associated with Chernobyl fallout in Europe.

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INTRODUCTION

As a principal long-lived gamma-emitting fission product, ^{137}Cs is a major contributor to the total integrated whole-body external dose delivered by the deposition of fallout following a nuclear weapon detonation or a nuclear reactor accident. Although a number of shorter-lived fission products may be prominent in the spectrum of environmental γ-radiation initially, most decay away within the first few days or weeks after deposition. In contrast, ^{137}Cs will contribute to the dose rate for decades due to its 30-year half-life. For fresh weapons debris that is only hours old, the fraction of the total dose delivered by ^{137}Cs is relatively small. However, in the case of delayed fallout from the stratosphere, it becomes more significant. Sixty per cent of the collective effective dose equivalent commitment from external radiation associated with past atmospheric nuclear weapons testing can be attributed to ^{137}Cs (United Nations, 1988). In the case of an accidental release of fission products from a nuclear power plant, Cs isotopes are especially significant due to their volatility and the large inventory that builds up in the reactor over time. For fallout in Europe from the Chernobyl accident, estimates indicate that 86% of the time-integrated exposure to infinity for external radiation will come from ^{137}Cs (USDOE, 1987).

The dose rate delivered from the fallout of ^{137}Cs is generally computed for an infinite half-space source geometry at a reference height of 1 m. The source strength is in the form of an inventory, that is, activity per unit area (e.g. Bq m^{-2}). As a simple approximation, one can invoke a source distribution that is in the form of an infinite plane atop the ground. While the activity can be assumed to be at or very near the surface of the ground shortly after the deposition event, this approximation will, in time, result in an overestimation of the dose rate. Providing there is little or no erosion, the dose rate above ground from a given inventory would be expected to decrease slowly over time due not only to radioactive decay, but also to penetration into the soil. This latter effect is influenced by soil characteristics and climate.

For ^{137}Cs from nuclear weapons test fallout, most of which was deposited in the late 1950s and early 1960s, it has been found in the past that the activity distribution with depth in the soil profile at most undisturbed sites can be reasonably well modeled using a negative exponential function with depth (Beck & Krey, 1980; Miller & Helfer, 1985). Our own data from across the USA have shown the mass relaxation depth for this exponential profile to vary between 1 and 30 g cm^{-2}. With this range of depth distributions, the resultant above-ground dose rate in air would be expected to vary by a factor of five for a given ^{137}Cs inventory

in the soil. For purposes of estimating worldwide population dose from weapons test fallout, the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) has used a conversion factor based on an average linear relaxation depth of 3 cm (United Nations, 1982). This was based on the early work of Beck (1966). For a soil density of 1·6 g cm^{-3}, this linear relaxation depth corresponds to a mass relaxation depth of 4·8 g cm^{-2}, which leads to a dose factor of 1·02 pGy h^{-1} per Bq m^{-2}. For fallout from the Chernobyl accident, UNSCEAR has proposed using a relaxation depth of 0·1 cm for the first month, then 1·0 cm through the first year and finally 3·0 cm thereafter (United Nations, 1988). Unfortunately, these numbers are based on rather sparse data and there clearly exists a need for more baseline information. Numerous studies have been conducted over the years on the distribution and migration of fallout in various ecosystems, particularly in relationship to the internal pathway to man through resuspension and crop uptake and also for estimating soil erosion. However, little work has been done to examine the geographical and temporal variations of ^{137}Cs depth profiles in soil and the resultant external dose. Apart from better estimating the global average dose, information of this nature would be useful to predict dose rates for areas with particular land and climate characteristics, so that improved dose commitment estimates could be made for various population groups as well.

In the course of various projects undertaken in the past 10 years by the Environmental Measurements Laboratory (EML) that have involved soil sampling, it has been observed that the depth profile of ^{137}Cs has generally been deeper at open field sites as opposed to forested sites. Although these sites were generally unrelated geographically and climatically, the data were suggestive of a systematic difference. Observations of this nature were made early in the history of fallout investigations (Alexander, 1967; Krey *et al.*, 1973) and have more recently been reported by other groups (McCallan *et al.*, 1980; Adriano *et al.*, 1981; Kuhn *et al.*, 1984). This paper presents the results of sampling seven paired sites in the northeastern US to further substantiate and quantify the differences between neighboring field and forest sites. The measured depth profiles of ^{137}Cs for these sites will be presented here and the corresponding above-ground air dose rate conversion factors will be inferred, comparing them to other general findings for watered lawns and desert sites in the western US. Based on depth profiles previously measured at two of the sites that are in the present study and one additional site, the dose as a function of time for field and forest sites is also inferred and compared to values computed for plowed land.

METHODS

Figure 1 shows the location of the seven sites chosen for this study. These sites include parkland and private property that contain both field and forest areas where the soil has not been disturbed by any significant human activity since at least 1950. The sampling was performed from late 1986 to mid-1988. Six of the seven field areas are open grass-covered lawns or meadows that have not been plowed or harrowed, while all seven forest sites are stands of mature deciduous or coniferous trees with dead leaf or needle litter ground cover. One field site (Sandy Hook, New Jersey) is a back-dune area with relatively sparse vegetative cover. The soils in these areas are not subject to any obvious erosional processes and can be considered reasonably stable, with the exception of the Sandy Hook field site where there is evidence of some localized shifting of surface material. The soil types range between sandy-loam and sandy. At each site, soil samples were taken from representative field and forest areas that are within 500 m of each other. Almost all of the ^{137}Cs at these sites would

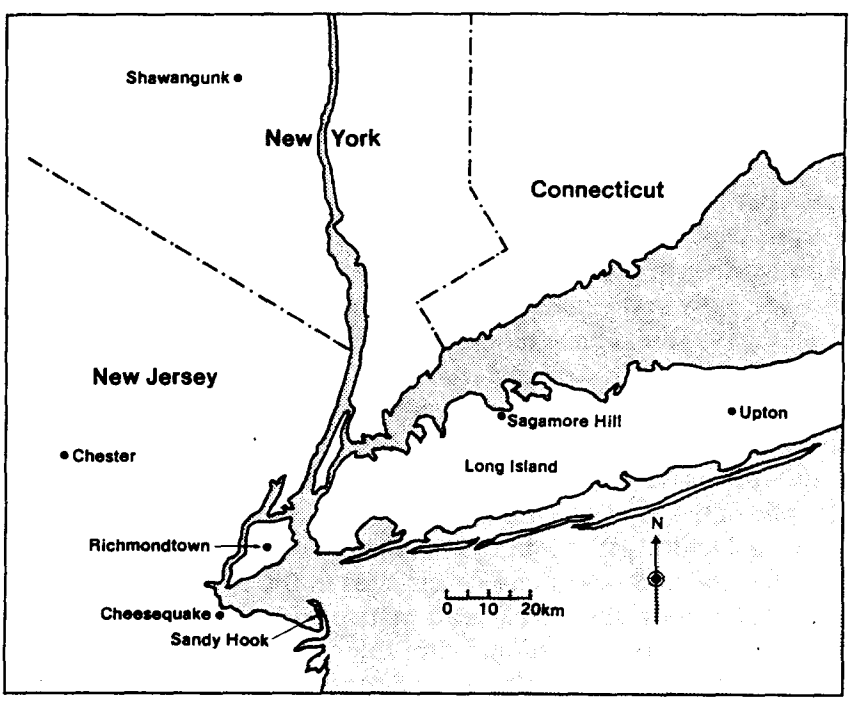

Fig. 1. Map of New York metropolitan area indicating locations of sampling sites.

have come from atmospheric weapons test fallout, the bulk of which was deposited from 1957 to 1965. Fallout from the accident at the Chernobyl nuclear power plant was measured in the New York area (Juzdan *et al.*, 1986) and the deposit of ^{137}Cs was <1% of the total inventory already present.

The collection and analysis of the soil samples were performed using standard EML techniques (EML Procedures Manual, in press). Soil samples were taken using an 8·9 cm diameter corer in depth increments of 0–5 cm, 5–10 cm and 10–15 cm. An auger was then used to extract a 15–30 cm increment. For the open field sites, the 0–5 cm cut included any dead or living grass, while, for the forest sites, this same cut included decayed organic matter but not fresh leaf litter. Care was exercised in the extraction of the soil sections to avoid the contamination of low-activity deep soil by high-activity surface soil. In most cases, duplicate cores were taken in each field and forest area. The sectioned samples were sealed in individual plastic bags and returned to EML where they were weighed, dried in an oven at <100°C and reweighed to determine water content. After mixing, an aliquot of each sample was packed into a 93 cm^3 aluminum can and hermetically sealed. Each can was analyzed using a high-resolution germanium γ-ray spectrometry system. ^{137}Cs concentrations were inferred using the full absorption peak count rate at 662 keV, the γ transition associated with the metastable state of the progeny, ^{137}Ba. Self-absorption corrections were applied based on the mass of the can contents. Detector calibration was performed with reference cans made up with solutions traceable to the National Institute of Standards and Technology. Organic content was inferred by loss on ignition at 550°C in a muffle furnace, after which a series of sieves and a shaker were used to determine soil particle sizes.

For most cases, the profiles of the measured ^{137}Cs fit reasonably well to an exponential function of the form

$$I = I_0[1 - \exp(-\alpha/\rho)\rho z] \tag{1}$$

where

I = the cumulative inventory (Bq m^{-2}) down to depth z,
I_0 = the total inventory to infinite depth,
α = the reciprocal of the relaxation length (cm^{-1}),
ρ = the soil density (g cm^{-3}), and
z = the linear depth (cm).

The above equation fits the profile as a function of mass depth, that is, ρz. Although the linear depth may be used instead and might under certain circumstances provide a better fit, the mass depth is a more fundamental

quantity to use for purposes of computing the above-ground dose rate since the density term affects the attenuation properties of the soil. Factors to convert a unit inventory of ^{137}Cs to exposure rate in air at 1 m above the ground for various values of α/ρ and for a representative soil composition have been published by Beck (1980). These data have been fitted to a smooth curve and converted to dose rate in air, which we denote as Γ. The results are presented in Fig. 2. Although Γ is somewhat dependent on the soil attenuation properties, typical changes in the relative amounts of the major elements in the soil do not greatly affect the computed dose rate. Despite the fact that there were marked differences in the apparent composition of the various soils collected in this study, particularly the surface soils, the measured mass attenuation coefficients varied by <5% about the mean, with no consistent difference between forest and field samples.

For the ^{137}Cs depth distributions that were encountered, roughly 65 to 85% of the dose rate above ground is attributable to the activity contained in the top 10 cm of soil. It is therefore important to fit the profile in this depth range to obtain a realistic value of Γ. Although the measured profile may depart from this idealized fit at greater depths, the predicted dose rate will still be reasonably accurate. In light of this, a convenient method for obtaining a value of α/ρ is to plot the logarithm of the integrated inventory

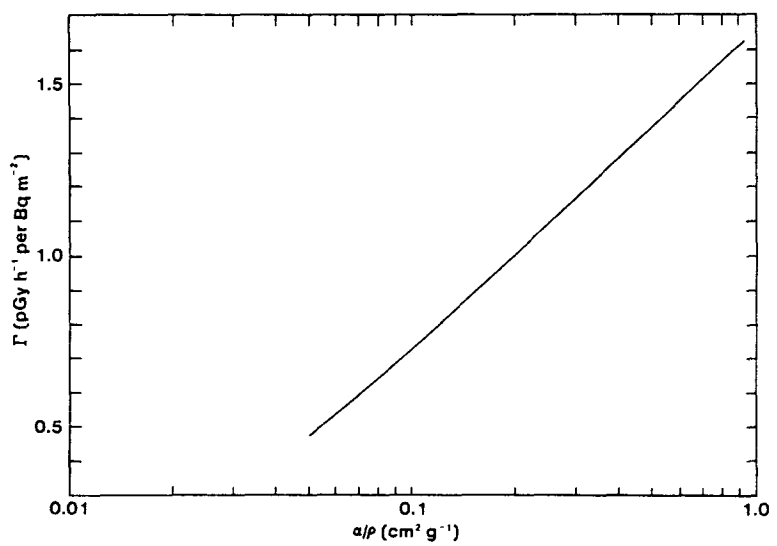

Fig. 2. Dose rate in air at one meter above the ground as a function of the depth parameter α/ρ for a unit inventory of ^{137}Cs in the soil.

below mass depth ρz, versus ρz. Then, the top two or three data points can be used to fit a straight line. Except for the Sandy Hook field site, this technique has been applied using a two-point fit for the forest cores and a three-point fit for the field cores. In each case, the top data point is simply 1·0, the normalized value of I_0. Although a more detailed depth profile could be constructed using finer sampling intervals, the use of the above exponential fit is convenient to apply and provides a means to infer quickly the dose rate. A sensitivity study was performed to determine whether or not finer detailing would substantially affect the calculated dose rate. At one site (Chester, New Jersey), two additional cores, with soil sections measuring 2 cm in thickness, yielded an exponential fit where the average value of α/ρ was within 5% of the value inferred from the fit to the coarser size profiles. In terms of the corresponding values of the dose rate conversion factor, the coarse and fine depth profiles yielded values of Γ which were only 2% different.

In the two field cores from Sandy Hook, the ^{137}Cs profile was characterized by a depleted inventory in the top soil section to the extent that the application of our standard exponential fit would be unrealistic. For these cases, a different approach was taken. For any arbitrary source depth distribution, one can break the soil half-space into a series of finite thickness elements and perform a numerical integration using the results for an infinite plane source for various heights above the ground (Beck & de Planque, 1968). In effect, this is the same as burying a series of plane sources in the ground at different depths, and the approximation holds since the mass attenuation and scattering properties of air are very similar to those of soil. The effects of low-energy scattering will produce a slight overestimation in the dose rate using this method, but only on the order of a few percent. The measured ^{137}Cs inventories in the soil sections can be fit to a smooth curve, or as a step function with mass depth, to provide the source strength term for each soil thickness element. This numerical integration method was also used as a cross-check against our exponential approximation for one forest core (Richmondtown, New York) and one field core (Sagamore Hill, New York). Although the profiles in these two cases were not fit perfectly by an exponential function, the values of Γ computed by the two methods differed by only 2% and 7%, respectively.

The source geometry in a forested area is complicated by the presence of trees, which act to attenuate the radiation field through shielding, and which may also serve as a source of radiation from any activity contained within the leaves and wood. However, these effects would not be expected to substantially alter the dose rate estimates. Attenuation effects would become sizable only next to massive tree trunks where the solid angle due to shielding becomes a non-negligible fraction of the total. To some

degree, the shielding effects would be compensated by the presence of the additional activity in the canopy and understory. This component would not be expected to be very large since it is generally accepted that plant uptake of ^{137}Cs is low in most soils where there is sufficient available potassium. This is likely the case in the study area here. Measurements at sites in the eastern US have indicated that the inventory contained in trees and other vegetation is a small fraction of that in the soil (Van Voris & Dahlman, 1976; Hay & Ragsdale, 1978; Rickard *et al.*, 1982). The samples of live leaf matter from the Chester site used in this study showed that the ^{137}Cs activity per unit area was on the order of 0·01% of the soil inventory.

RESULTS

Soil characteristics for the top core sections (0–5 cm) are listed in Table 1. The analysis of the duplicate samples showed close agreement, so averages are given for these data. Also presented in Table 1 are mean annual precipitation amounts for each site based on the 1951–1980 records for nearby weather stations (NOAA, 1986, 1987). The measured inventory data for the soil cores at each site are plotted against the mass depth in Figs 3–9. In order to facilitate a comparison between sites, the depth data are given for dry soil. Samples were counted long enough in each case to achieve a 1σ Poisson counting error of $<5\%$ of the inventory value. Except for the Sandy Hook field area, reproducibility of the depth profile between duplicate cores was generally good.

Table 2 gives the corresponding values of Γ for each field and forest core sample. The ratio of Γ between the forest and field areas at each site is given using averages where duplicate cores were taken. Again, to facilitate comparison, the values are for dry soil. Soil moisture content varied in the samples since they were collected at different times under conditions that ranged from moist to dry. To obtain a correction factor to these values of Γ, the attenuation from soil moisture can be treated on an average basis. Any addition of water makes the soil more dense, which in effect buries the source deeper on a mass basis. For example, at a dry density of 1·2 g cm^{-3}, an addition of water to the point of 25% in the total soil–water mass raises the density to 1·6 g cm^{-3}. For a uniform profile with depth, the effect is like that of a source-term dilution. Although the γ-ray attenuation and scattering properties of water are slightly different from those of soil, as a reasonable approximation one can simply reduce the above-ground dose rate in air by the same percentage that the density of the soil increases, in this case 25%. However, the closer the activity is to the soil surface, the less the effect. Using the curve in Fig. 2, we find that the same

TABLE 1
Precipitation—Characteristics of Top Soil Sections (0–5 cm)

Site	*Mean annual precipitation (cm)*	*Type*	*Density ($g\ cm^{-3}$)*	*Size fractions (%)*			*Organic (%)*
				Gravel (>2 mm)	*Sand (2–0·06 mm)*	*Silt and clay (<0·06 mm)*	
Chester, NJ	128	Field	0·99	1	54	45	10
		Forest	0·54	6	41	53	24
Upton, NY	116	Field	1·22	0	73	27	7
		Forest	0·30	6	58	36	36
Shawangunk, NY	116	Field	0·69	NM	NM	NM	NM
		Forest	0·66	NM	NM	NM	NM
Richmondtown, NY	101	Field	0·97	8	66	26	7
		Forest	0·45	19	48	33	16
Sagamore Hill, NY	112	Field	0·83	5	40	55	12
		Forest	0·58	3	47	49	14
Cheesequake, NJ	112	Field	0·95	3	40	57	13
		Forest	0·45	0	79	21	27
Sandy Hook, NJ	108	Field	0·99	0	94	6	5
		Forest	0·90	0	97	3	12

NM, No measurement.

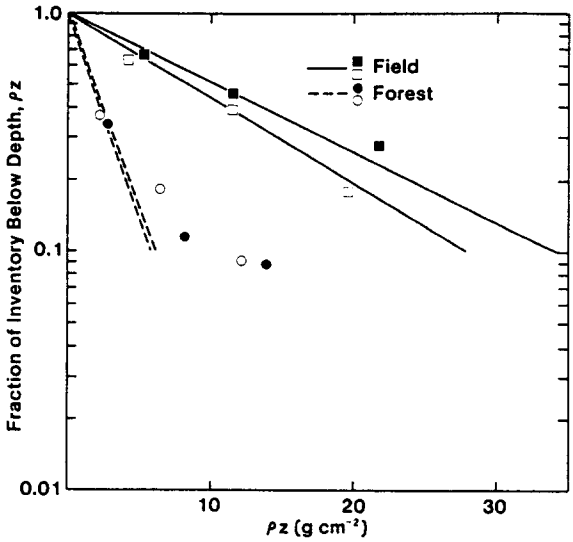

Fig. 3. ^{137}Cs inventory depth profiles in dry soil at Chester, New Jersey

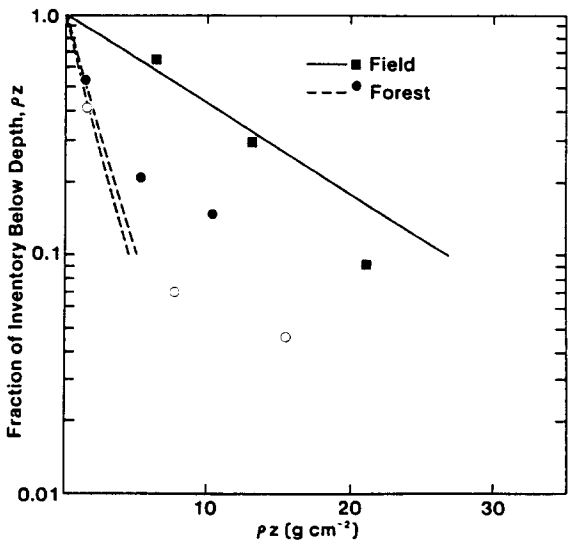

Fig. 4. ^{137}Cs inventory depth profiles in dry soil at Upton, New York.

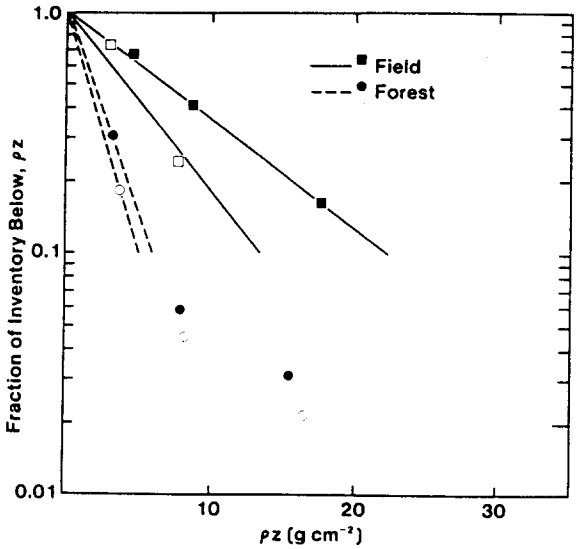

Fig. 5. ^{137}Cs inventory depth profiles in dry soil at Shawangunk, New York.

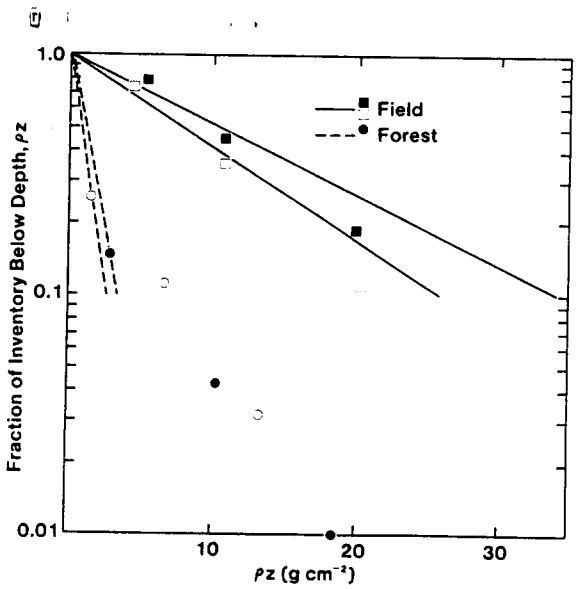

Fig. 6. ^{137}Cs inventory depth profiles in dry soil at Richmondtown, New York.

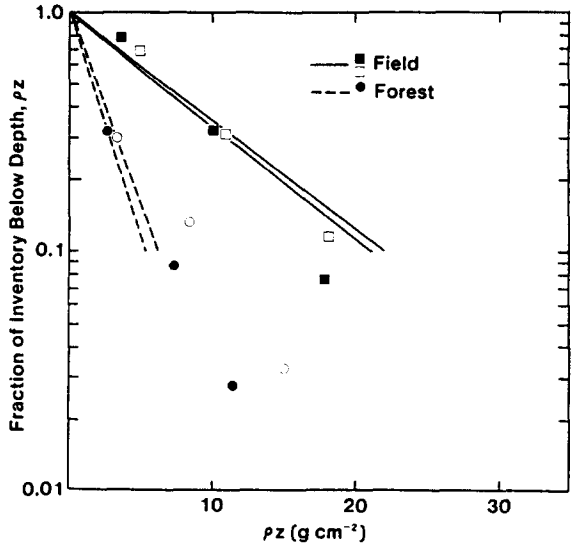

Fig. 7. ^{137}Cs inventory depth profiles in dry soil at Sagamore Hill, New York.

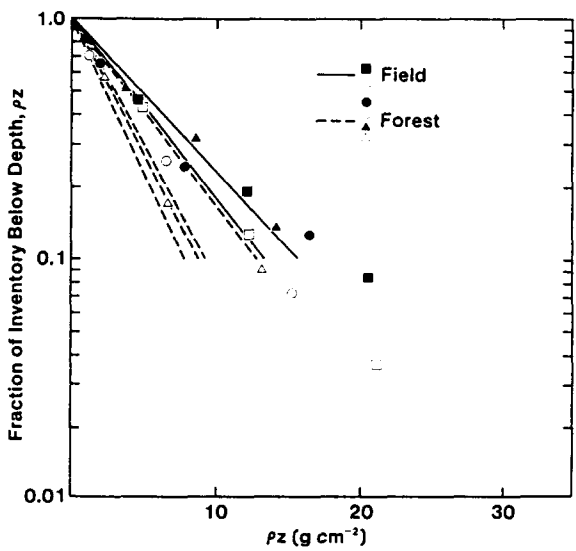

Fig. 8. ^{137}Cs inventory depth profiles in dry soil at Cheesequake, New Jersey

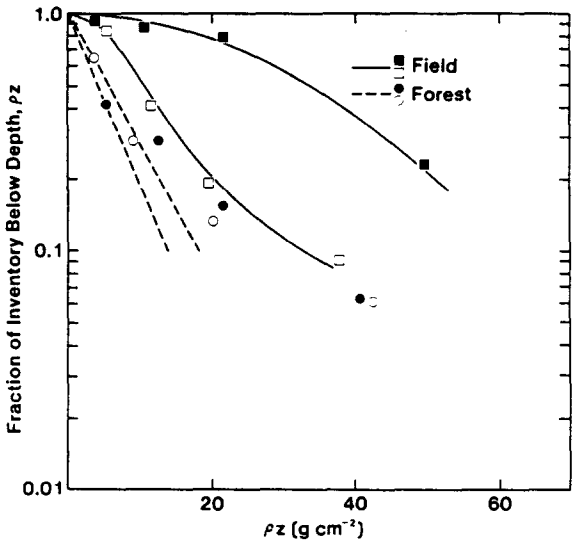

Fig. 9. ^{137}Cs inventory depth profiles in dry soil at Sandy Hook, New Jersey.

addition of water to a soil with an exponential depth profile for ^{137}Cs where $\alpha/\rho = 0{\cdot}1$ (dry) would lower this value to 0·075 and cause the dose rate to fall by 15%. A more shallow profile would have the value of α/ρ lowered from 0·5 to 0·375, resulting in the dose rate falling by only 8%.

For the soil cores collected in this study, the moisture content mean and standard deviation was $19 \pm 7\%$ for the forest sites and $17 \pm 7\%$ for the field sites. However, the moisture profile was not uniform with depth as typically the upper layers contained more water. The moisture content for the 0–5 cm section averaged $31 \pm 11\%$ for the forest and $20 \pm 11\%$ for the field. Despite this difference, the value of Γ based on the wet depth profile was found to average 12% lower than the dry values given in Table 2 for both the forest and field. This resulted from the fact that, as discussed above, a given moisture content has a greater effect on a deeper profile, as found in the field, than it does for a shallow profile. Although it is not considered here, it should be noted that snow cover will have a substantial effect on the external dose rates, and would therefore have to be taken into account for areas where it exists for a large fraction of the year.

In Table 3, we present our final estimates of the current values of Γ, taking into account the mean 12% soil moisture reduction factor. For comparison, we also give values of Γ based on previously published results of ^{137}Cs wet depth profile measurements that were performed within the last 10 years in other parts of the USA.

TABLE 2
^{137}Cs Dose in Air Conversion Factors for Dry Soil

Site	Γ *(Field)*[a]	Γ *(Forest)*[a]	*Average forest/average field*
Chester, NJ	0·57	1·28	2·08
	0·66	1·28	
Upton, NY	0·66	1·35	2·08
	—	1·40	
Shawangunk, NY	0·95	1·28	1·57
	0·73	1·35	
Richmondtown, NY	0·57	1·49	2·41
	0·71	1·59	
Sagamore Hill, NY	0·76	1·30	1·68
	0·76	1·26	
Cheesequake, NJ	0·88	1·11	1·18
	0·95	1·16	
	—	1·09	
	—	0·95	
Sandy Hook, NJ	0·55	0·85	2·39
	0·19	0·92	
Mean ± SD	0·69 ± 0·20	1·23 ± 0·20	

[a]pGy h^{-1} per Bq m^{-2} at 1 m above the ground.

TABLE 3
Average ^{137}Cs External Dose in Air Conversion Factors Measured in Recent Years for Wet (in-situ) Soil

Area	*Reference*	*Land type*	*Year*	Γ[a]
Northeastern US	This study	Field	1986–88	0·61
Northeastern US	This study	Forest	1986–88	1·08
Utah cities, towns	Beck & Krey (1980)	Irrigated lawns	1979	0·59
Southwestern US	Miller & Helfer (1985)	Natural terrain (field and forest)	1980–84	1·23

[a]pGy h^{-1} per Bq m^{-2} at 1 m above the ground.

DISCUSSION

The values of Γ listed in Table 2 vary from site to site. However, the standard deviation of the values for the forest samples (16% of the mean) is indicative of a reasonably tight distribution when one considers that the sites were geographically well separated and that the soil compositions were different. If the Sandy Hook site, which was atypical due to the unstable soil, is not included, the corresponding standard deviation would likewise be reasonably small (18%) for the field subset. More significantly, these data consistently show a difference in the value of Γ between neighboring field and forest areas. In part, this is due to the lower soil density which is typical of the upper forest soil layer, as can be seen from the values in Table 1. The value of ρ for dry soil in the 0–5 cm section averaged 0·54 g cm^{-3} for the forest as compared to 0·92 g cm^{-3} for the field. As such, the forest surface layer provides significantly less attenuation of the γ flux for the same linear depth profile.

Aside from the density effect, however, there are apparent differences in the linear depth profile data. For the forest areas, the fraction of the total inventory residing in the top 5 cm ranged between 30 and 85% and on average was 58%, while for the field areas the range was 10 to 59% with an average of 31%. This suggests that varying retention mechanisms are likely affecting the profiles. No strong correlation was found between the inventory fraction in the top soil layer and the soil characteristics listed in Table 1. Other researchers have found no correlation between these parameters and the levels of another fallout product, ^{129}I, in surface soil (Boone *et al.*, 1985). However, relatively high Cs sorption has been found for the very top layer (O horizon, raw humus) of some soils (Bachhuber *et al.*, 1982). It is also possible that the clay mineralogy differs among these soils, as clay can play a role in the ability of the Cs to bind. Other factors might include exchangeable potassium content, potential exchange capacity and pH value, all of which can influence uptake by vegetation and, thus, influence the profile (Kuhn *et al.*, 1984). The likely explanation, in the case of the locations used in this study, is the phenomenon whereby the forest litter layer acts as a buffer by absorbing water from precipitation, thus slowing the movement of trace constituents into and through the mineral soil layers (Adriano *et al.*, 1981).

We consider the Sandy Hook site atypical because it is a back-dune area characterized by a soil which is almost pure sand. Under these conditions, the ^{137}Cs would be expected to assume a depth profile that is deeper than usual due to more rapid penetration into the ground. Evidence of deeper penetration in sandy as opposed to finer textured soils has also been found in the course of studying erosion processes (Menzel *et al.*, 1987). Also, in

an open area with little vegetative cover, the effects of shifting sand would tend to redistribute the activity and, in time, possibly bury it in some areas. The depletion of the surface layer that can be seen in the data for this site demonstrates this phenomenon in general. However, the processes at work are such that one would expect a high degree of heterogeneity as is shown by the large difference in the two values of Γ for the field area at the Sandy Hook site.

Apart from any difference in dose rate which can be caused by the depth distribution of the ^{137}Cs in the soil, there is the question of the inventory itself and how it may vary in undisturbed terrain. In the past, there has been speculation as to whether a forested area might intercept more fallout than an open field site. This could occur in the form of dry deposition through a mechanism of enhanced turbulence in the atmospheric surface layer over the forest where the roughness length is much greater than that over a grassy plain. In areas where the climate is moist, the effect on global fallout inventory would tend to be minor since most deposition was in the wet form. The experimental procedures carried out were not specifically designed to yield a highly precise estimate of the inventory in that only two cores were taken at any one location. However, there were 16 forest cores and 13 field cores in total and it is possible to compare the average values. The forest sites had a mean inventory of 3·9 kBq m^{-2}, while the field sites averaged 3·8 kBq m^{-2}. This difference is not considered statistically significant since the average values for each set would be expected to have a coefficient of variation on the order of 6 to 7%. The fact that no substantial difference is demonstrable is consistent with a previous study performed in this area of the US (Krey *et al.*, 1973). Despite this finding for global fallout, interception by forest canopies has been demonstrated in the vicinity of a reprocessing facility (Adriano *et al.*, 1981) and a similar effect might occur downwind of any local source of atmospheric release if dry deposition is a dominant mechanism. Any resultant increase in inventory in a forested area would augment the forest/field dose rate ratio that is caused by depth profile differences.

An alternative method of inferring the dose rate delivered by ^{137}Cs in the soil is in-situ γ-ray spectrometry (Helfer & Miller, 1988). This technique was used at one of our sites (Chester) at several spots in both the field and forest areas. Using a portable germanium spectrometer at 1 m above the ground, the full absorption peak count rate at 662 keV was observed to be a factor of 2·24 higher on average in the forest relative to the field. This count rate is essentially a measure of the primary γ flux density, which is closely related to the dose rate. An exact conversion to dose rate would require a precise knowledge of the source geometry,

which is rather difficult to ascertain in a complex situation such as a forest. However, since most of the inventory is in the soil, one can invoke the normal half-space geometry. As such, the application of the appropriate conversion factors for the measured ^{137}Cs depth profiles yields a forest/field dose rate ratio at the soil sampling spots of 1·90. This value compares favorably with the ratio of 2·08 listed for this site in Table 2 that was based on the soil sample analysis. Some disagreement can be expected since the in-situ measurements were performed at a different time from the soil sampling, and soil moisture conditions may not have been the same. However, the lower ratio for the in-situ case is consistent with the fact that the surrounding trees provide some measure of attenuation which therefore lowers the value of Γ in the forest. This experimental check thus appears to offer confirmation that the shielding effect is indeed small.

The data given in Table 3 for sites in the western US further attest to the differences in dose rate that can arise depending upon the type of land under consideration. Although there was a fairly wide distribution of depth profiles reported for natural terrain sites in the western US (Miller & Helfer, 1985), on average the ^{137}Cs was still relatively close to the soil surface as compared to the case of irrigated lawns (Beck & Krey, 1980). Since there is significantly less rainfall in these western states, one may infer from the data in Table 3 that field areas subject to a moderate to heavy throughput of water from either natural or anthropogenic sources will tend to have a lower value of Γ because the Cs is driven further into the soil. On the other hand, unwatered field areas in arid regions or forested areas where the litter layer acts as a buffer will tend to be characterized by higher Γ values due to the Cs residing close to the soil surface. Aside from the value of Γ, it should be remembered that the dose rate from ^{137}Cs will also change significantly across regions of the US due to variations in the soil inventory that result from differences in precipitation. Thus, although Γ may be high for arid areas, the dose rate will still be low because relatively little fallout was deposited.

The dose rate differences observed at present, 25 years after the last significant input of global weapons test fallout, are representative of the middle period in the decay history of ^{137}Cs in the environment. In Fig. 10 we present a limited set of historical dose rate data based on measurements of soil samples collected in the past at two of the sites in this study as well as a third site, a forest clearing in North Eastham, Massachusetts. By plotting the dose rate per unit inventory at the time of each sampling, the effect of radioactive decay is removed. The value of 2·2 pGy h^{-1} is also plotted for the year 1963 as an approximate peak value at the time of heaviest fallout. This value corresponds to the dose rate from a plane source, which,

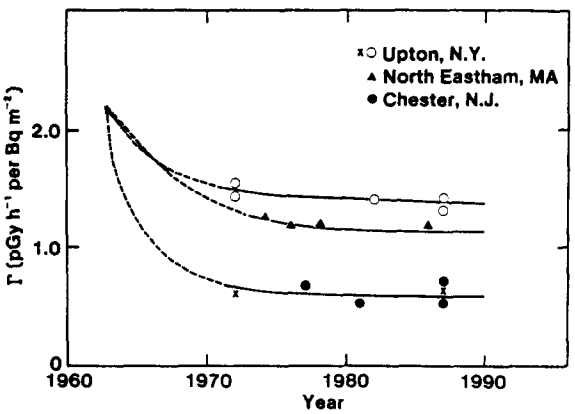

Fig. 10. Dose rates in air per unit inventory of ^{137}Cs in the soil inferred from historical soil sample data at three different locations at various times since assumed peak value in 1963. Upper two curves are for forest sites and lower curve is for field sites.

because of ground roughness, can be considered to have a negative exponential profile where $\alpha/\rho = 6{\cdot}25\,cm^2\,g^{-1}$. This type of profile has been used in the past to measure fresh fallout (Helfer & Miller, 1988). It is apparent from the data in Fig. 10 that the dose rate has been decreasing very slowly, if at all, in recent years. Although the penetration of ^{137}Cs into the soil probably continues to some degree, the effect on the above-ground dose rate appears to be rather small. The interpretation of these historical data is complicated by the fact that the fallout was not delivered in the form of a short-term pulse, but was rather spread out over a number of years. Some small but measurable amounts of additional fallout were still being deposited throughout the 1970s. Despite this and the fact that there are no experimental exposure rate data for the 1960s in Fig. 10, it is nonetheless apparent that there had to have been a sharp falloff from the peak value at the time of deposition, based on the relatively low values of Γ seen in the early 1970s. This is substantiated to some degree by fallout measurements in the early 1960s at open field sites that showed a linear relaxation depth of 3 cm, i.e. $\alpha/\rho = 0{\cdot}21\,cm^2\,g^{-1}$, where $p = 1{\cdot}6\,g\,cm^{-3}$ (Beck, 1966). Measurements of ^{134}Cs in till made in Sweden 5·5 months after the Chernobyl fallout event showed relatively quick penetration into the soil as well, with the average 1/2 value depth being 2·8 cm (Carbol *et al.*, 1988). Also the indication in Fig. 10 that the value of Γ for the forest falls off more slowly than that for the field is consistent with findings of a more shallow profile in wooded land as opposed to an open field site that was reported in the late 1960s at a site in Maryland (Alexander, 1967).

The dose rate data set for this study covers a limited time frame. However, it is still possible to estimate the dose commitment because of the apparent stabilization of the ^{137}Cs in the soil a few years after deposition. To do this, we have assumed that the dose rate per unit deposition (i.e. a constant source with no radioactive decay) decreases exponentially with time from its peak value at the time of deposition to a saturation value where the ^{137}Cs has reached a more or less fixed profile in the soil and the dose rate changes negligibly. This can be expressed by the function

$$\Gamma = \Gamma_0 + (19{\cdot}3 - \Gamma_0)e^{-t/\tau} \tag{2}$$

where

Γ = the dose rate in air (nGy year^{-1}) per unit inventory in the soil (Bq m^{-2}),
Γ_0 = the saturation value of the dose rate per unit inventory,
t = time (years),
τ = the time constant for the exponential decrease toward the saturation value (years), and
19·3 = the dose rate per unit inventory at the time of deposition.

Using the extrapolated part of the curves in Fig. 10, we have estimated the value of τ at 2·5 years for field sites and 5·0 years for forest sites. The respective values of Γ_0 for 1 Bq m^{-2} are taken to be 5·4 and 9·5 nGy year^{-1}, based on the data in Table 3 for field and forest. Using these constants in eqn (2), we present in Fig. 11 the integrated doses as a function of time, taking into account both the reduction due to penetration into the soil and the radioactive decay of the ^{137}Cs. Also presented in Fig. 11 are the integrated doses for an open field area where the soil has been plowed to depths of 15 and 30 cm. Repeated annual tilling would tend to thoroughly mix the soil. One study of a cultivated vegetable patch has shown that a more or less uniform profile with depth for fallout radionuclides existed down to 25–30 cm (Hardy *et al.*, 1977). Thus, the values of Γ were derived by using a deposit of 1 Bq m^{-2}, mixed homogeneously in soil at a density of 1·6 g cm^{-2}, and calculating the dose rate in air at 1 m above the ground using previously published data (Beck & de Planque, 1968). To compute the integrated dose, Γ was then taken to be constant and only a radioactive decay term was applied.

Measurements of Chernobyl fallout in Germany revealed that the deposited activity quickly assumed an exponential depth profile with a relaxation length on the order of 1 cm (Gogolak *et al.*, 1986). At a soil density of 1·6 g cm^{-3}, this would result in a dose in air for ^{137}Cs about 40% lower than we have assumed at time of deposition. This quick penetration into the soil may have been related to rainfall intensity and soil conditions

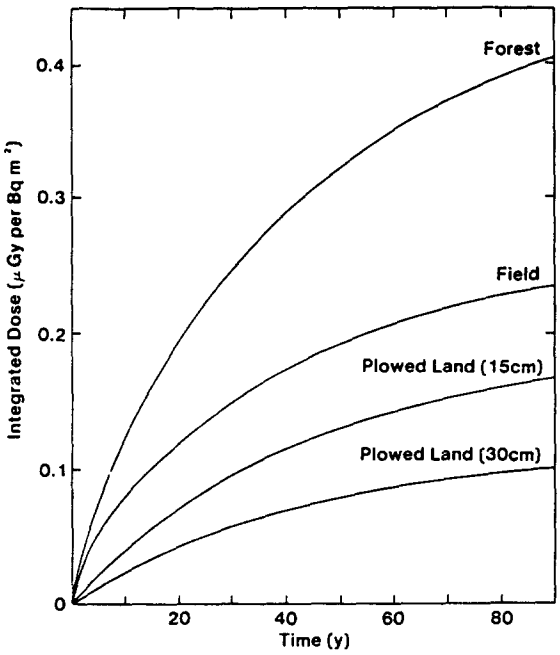

Fig. 11. Predicted integrated doses for external gamma radiation per unit deposition of ^{137}Cs for different land types taking into account both radioactive decay and penetration into the soil.

at the time of fallout input. While this type of finding is important for predicting dose for short-lived isotopes, it should be emphasized that the manner in which we have computed the dose for ^{137}Cs provides a result which is not highly sensitive to the values of τ or the dose rate at the time of deposition. Rather, it is mostly dependent upon the value of Γ_0. In time, further studies may indicate that some modification to this treatment is necessary, particularly if the dose rates show significant departure from the assumed saturation values. We intend to continue following the ^{137}Cs depth profile in future years at the Chester, New Jersey, site which is a dedicated regional monitoring station for EML. However, it is likely that the marked differences between the various types of sites, computed with the model given here, would be sustained even though the absolute values of the doses may change.

Extending our integration to infinite time, we present in Table 4 estimated dose commitments for field, forest and plowed land. The value suggested by UNSCEAR (United Nations, 1982), based on an average 3 cm relaxation depth for an exponential ^{137}Cs profile and a mean life of 43·6 years, is also given. As a point of comparison, it should be noted that

the dose commitment that would result from assuming a plane source geometry with no penetration into the soil and no effects of ground roughness, certainly a gross overestimate, would be 0·97 μGy for 1 Bq m^{-2}.

These dose values are for the outdoor environment as typified by rural, suburban and to some extent non-central-city urban settings, where a large fraction of the land can be characterized as soil covered. In general, population dose would be lower still due to shielding provided by buildings. For conversion from air to tissue dose, UNSCEAR applies a factor of 0·3 to the value given in Table 4, which takes into account both

TABLE 4
Predicted ^{137}Cs External Dose Commitment for Various Land Types

Type	*Dose in air (μGy per Bq m^{-2})*
Forest	0·46
Field	0·27
Plowed land (to 15 cm depth)	0·19
Plowed land (to 30 cm depth)	0·12
UNSCEAR (United Nations, 1982)	0·39

the housing and body shielding effects. The results given in this paper should not be applied to the central-city urban environment where a large fraction of the land cover is pavement. Under these conditions, runoff from streets and building surfaces would substantially alter the source term as has been found to be the case for Chernobyl fallout (Jacob *et al.*, 1987). The behavior of fallout in the urban environment is further complicated by the apparent ability of some surfaces such as clay tile roofs to retain Cs (Nicholson, 1989).

Although the UNSCEAR value falls within the range of our data, it may not be representative for the US in that it is substantially higher than our value for open fields, which is probably most representative of the type of land (lawns) that surround US housing. In new home construction and sometimes in the case of existing dwellings, the soil surrounding a house is cultivated, which would result in an even lower dose, perhaps on the order of the value given for 15 cm deep plowed land. Residents of central cities, 30% of the US population (USDC, 1984), would be expected to have still smaller doses due to the flushing effect and the greater shielding provided by larger buildings. In the case of rural areas, which represent only 26% of

the population (USDC, 1984), the dose associated with plowed land would be most applicable to that fraction of the population living in agricultural regions. Apart from these average values, there clearly exist small population subsets which might receive even higher doses than the UNSCEAR value. Included would be residents of homes that abut woodland and workers who spend a large fraction of their time in forested areas such as park rangers and logging industry personnel.

SUMMARY AND CONCLUSIONS

The derived dose rates for the measured depth profiles of global fallout ^{137}Cs at the seven sites in this study ranged between a factor of 1·18 and 2·41 higher for forest as compared to neighboring field areas. On average, the present forest/field dose rate ratio is about 1·8. Based on historical dose rate data, which imply a rapid falloff from time of deposition to a near saturation value a few years later, the forest/field dose commitment ratio is estimated to be 1·7. Calculations indicate that the dose commitment for a forest site would be a factor of four higher than that for a deeply plowed field for the same amount of ^{137}Cs deposition.

To some degree, the observed differences in the dose rate between the forest and field sites can be explained by the lower soil density that characterizes the forest surface layers. However, it is also apparent that this layer tends to hold the ^{137}Cs, thus acting as a buffer against downward driving physical processes.

The soil data from this study, along with the results from previous sampling in the southwestern US, suggest that field areas in moist regions and irrigated land in general will tend to have deeper profiles of ^{137}Cs (and consequently lower external dose rates per unit deposition) as compared to field areas in dry regions or forested areas in general. Considering the type of land that surrounds its housing, the US population is receiving a dose commitment from weapons test fallout that is likely to be 30–70% lower than the worldwide average of 0·39 μGy per Bq m^{-2} suggested by UNSCEAR.

The findings of this present work can be considered representative of the northeastern US, but they may also be applicable to other areas with similar climate and land cover. This type of data is useful in estimating population doses associated with ^{137}Cs from nuclear weapons test fallout and also has implications for doses delivered on contaminated land following a fission product release from a nuclear reactor accident. We conclude that it would be important to collect similar histories of dose rates for different land types in regions on the European continent for better

establishing the expected external dose commitment from fallout associated with the Chernobyl accident.

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