

Vertical Distribution of ^{137}Cs in the Vicinity of the Waste Isolation Pilot Plant

The distribution of ^{137}Cs activity with depth in soil profiles is utilized in this study to evaluate the importance of transport mechanisms affecting the fate of radionuclides in the arid environment of the northern extremity of the Chihuahuan desert where the WIPP site is located (Fig. 18). Fallout from aboveground nuclear testing was the primary source of ^{137}Cs in soils (Ritchie, 1990), although a potential source in the near vicinity of the WIPP site is the Gnome test site, about 8.8 km southwest of the WIPP site. This test involved a 3.1 kiloton yield nuclear underground detonation in 1961 from which venting to the atmosphere occurred (Faller, 1994).

Kirchner et al. (2002) utilized surface soil data (2-cm depth) collected using a grid scheme to characterize radionuclide concentrations in soils near the WIPP. The activity concentrations of $^{239,240}\text{Pu}$, excluding one value that was less than the minimum detection level (MDL), ranged from 0.015 to 0.51 Bq kg⁻¹. The ^{137}Cs activity concentrations for the surface soils, also excluding one value <MDL, ranged from 0.31 to 15 Bq kg⁻¹, a range of more than an order of magnitude. The excluded values were associated with the same sample. The mean ^{137}Cs activity concentration from soils collected in the near vicinity of the WIPP was 3.1 (S.E. = 0.23) Bq kg⁻¹ and was slightly higher for soils at a reference site approximately 20 km SE of the WIPP site, where mean ^{137}Cs activity concentration was 6.2 (S.E. = 0.52) Bq kg⁻¹. Variability in surface soil concentrations reported in this study was attributed to redistribution of contaminated soil particles via resuspension and erosion, and perhaps from variation in rates of vertical transport.

The variability in ^{137}Cs activity concentrations reported by Kirchner et al. (2002) in surface soil points to the dynamic complexity of the soil landscape. The landscape surrounding WIPP is dominated by sandy soils on undulating plains and low hills. The soils developed in noncalcareous, wind-worked sandy deposits are highly susceptible

to wind and water erosion (Chugg et al., 1971). Understanding the relative role of redistribution of soil within a system, and how that redistribution depends on episodic events, is directly relevant to addressing the general issue of contaminant transport in semiarid ecosystems (Whicker et al., 2002).

The objective of this investigation was to evaluate whether some of the variability in surface concentrations of radionuclides could be explained by spatial variability in vertical transport and soil erosion processes, and whether a sufficient change in surface concentrations due to erosion could occur as to be misinterpreted as an indication of a release of contaminants from the WIPP site.

$^{239,240}\text{Pu}$ is the radionuclide of most concern in terms of a potential release from the WIPP. $^{239,240}\text{Pu}$ is generally considered to have low mobility in the environment because it tends to have low solubility. However, the vertical distribution of ^{137}Cs was investigated rather than the distribution of $^{239,240}\text{Pu}$ because the activity concentrations of ^{137}Cs in soils from fallout are considerably greater than those for $^{239,240}\text{Pu}$. In addition, ^{137}Cs is a gamma-emitter and it requires little effort to prepare samples for analysis. Although chemically quite different, ^{137}Cs is expected to behave similarly to $^{239,240}\text{Pu}$ in soils because it adheres tightly to cation-exchange surfaces and is relatively inert chemically (Coppinger et al., 1991).

^{137}Cs from fallout has frequently been used to characterize soil particle transport (Schimmack et al., 2002; Pimentel et al., 1995). Cesium can adsorb to organic matter as well as soil mineral particles. However, cesium adsorption on clay minerals is strongly specific, whereas adsorption on organic matter is considered to be non-specific (Staunton and Levacic, 1999). Therefore, even when organic matter is responsible for a large proportion of soil cation exchange capacity trace amounts of radiocesium will be preferentially adsorbed on clay minerals. Nevertheless, organic matter can play a role in the mobility of cesium in the soil. There is increasing evidence that

bioavailability increases with increasing organic matter content.

Given that ^{137}Cs binds preferentially to soil mineral particles, and assuming that deposition onto a landscape was uniform, then the current distribution of ^{137}Cs reflects integrated effects of water, wind, soils, vegetation, and topography on erosion and deposition during the 35 to 45 years after the major deposition period (Walling and Quine, 1991). The horizontal movement of ^{137}Cs in soils by biological and chemical processes is small in comparison with movement of ^{137}Cs by physical processes (water and wind) that transport soil particles across the landscape (Ritchie and McHenry, 1990). Coppinger et. al. (1991) examined patterns and assessed transport mechanisms of soil erosion and deposition in a native sagebrush steppe landscape in south central Wyoming using ^{137}Cs . Results showed that wind was the major factor responsible for small-scale (0.5-10 m) transport of soils in this landscape and that there was little or no intermediate-scale (approximately 100 m) transport occurring.

Downward migration of Cs is dependent on numerous factors: sorption and desorption processes in soil, soil mineral composition, soil moisture and precipitation, particle size and specific surface area, soil type, pH, organic matter content, cation competitive effects, etc. (Barisic et al., 1999). Increased concentrations of competing ions such as Na, K and H slightly decrease ^{137}Cs adsorption (Ritchie and McHenry, 1990). Adsorption of ^{137}Cs on soil particles is considered to be rapid, yielding a distribution in undisturbed soil profiles that shows an approximately exponential decrease with soil depth (Ritchie and McHenry 1990; Barasic et. al., 1999). Isaksson et. al. (2001) examined the depth distribution of ^{137}Cs in labeled soils annually over a six-year period in southern Sweden. They found great similarities between years, indicating that once deposited, ^{137}Cs migrated into the topsoil, and thereafter showed a slow migration.

This study examines the vertical distribution of ^{137}Cs and the characterization of four soil profiles in the vicinity of the WIPP site. These distributions can provide insight into the dynamics of cesium transport over the

last four decades in a desert environment. In addition, results of an experiment are reported that was conducted to estimate the rate of vertical transport of cesium in WIPP soils due to the infiltration of water. In this experiment, a thin layer of soil spiked with ^{134}Cs was applied to the surface of eight columns of soil and leached with a volume of artificial rainfall approximating 30 years of rainfall. The soil was typical of the sands that dominate the WIPP soil environment. This experiment helps to evaluate whether vertical transport via desorption, diffusion or flow, and re-adsorption is likely to be responsible for the observed depth distributions of ^{137}Cs .

Methods

Field site profiles

Four soil profiles were sampled in the spring of 2001 for the purpose of defining the distribution of ^{137}Cs , as a function of depth. Individual profiles (Fig. 19) were sampled using the depth increments listed in Table 13. When an indurated layer was encountered at a depth shallower than 150 cm, sampling was terminated within the last unconsolidated layer at the assigned sampling thickness.

Sampling site locations were determined by the nature of the surface (minimal disturbance, absence of brush vegetation, etc.) and within a 50-m square centered on a designated grid node marker. Two sites were located within a grid surrounding the WIPP site and two sites were located within a reference grid approximately 20 km SE of the WIPP site (Kirchner et. al., 2002), (Fig. 18).

All sampling equipment (buckets, scoops, etc.) was non-metallic with the exception of excavation equipment (shovels, chipping bars, hammers). The vertical profile face was scraped clean with a plastic trowel after excavation. Samples were collected by depth increment by removing a 0.25m^2 area beginning at the profile face and working back into the undisturbed profile (Fig. 20). A 25-cm alley was removed from around the sampled area to the depth sampled before the next depth increment was collected. Soil samples were sealed in plastic containers for transport to the lab.

Soil samples were air-dried in the laboratory and sieved through a 2-mm sieve. Particle-size distribution was determined using the pipette method described by Gee and Bauder (Gee and Bauder, 1986). Specific conductivity and pH were determined on 1:2 (soil:deionized water) suspensions. Activity concentrations of ^{137}Cs in the soil samples were measured with high purity Ge gamma-spectroscopy.

Column experiment

Eight soil aliquots were prepared by spiking them with ^{134}Cs . The ^{134}Cs activity in the spiked aliquots ranged from 1310 to 1380 Bq. The spiked aliquots were subjected to three wetting and air-drying cycles with stirring performed at least five times per day. Adsorption of the ^{134}Cs to the soil particles was verified by extracting an aliquot of the water used in the last wetting cycle, filtering it and determining its ^{134}Cs activity. Three wetting and drying cycles were determined to be sufficient to result in the absorption of all but negligible quantities of the spike.

Eight columns were prepared using soil collected from the vicinity of the WIPP site (from the soil surface to a depth of 2 cm). The soil was sieved to remove roots, leaves and rock fragments and then the columns were filled to approximately 14 cm of depth (Fig. 21). Each column consisted of a piece of PVC pipe 20 cm in diameter and approximately 22 cm long. A perforated PVC plate was attached to the bottom of the column and a fine stainless steel wire screen placed on top of the plate within the column. The columns were pretreated by leaching with 1.6 L of 0.001 mole/L CaCl_2 solution each day for seven days. A thin "marker layer" of white sand (washed) was added to the top of each column followed by a thin layer of soil that had been spiked with ^{134}Cs . This was overlain with a filter paper, a layer of the clean, white sand, and another filter paper that were used to minimize disturbance of the soil when water was being applied. Approximately 900 cm of water (the equivalent of about 30 years of rainfall) was then applied at a rate of 3.2 L d^{-1} .

Sampling of the columns was conducted using a special tool designed to enable thin layers to be sliced from the surface of the

column. The tool consisted of a circular PVC plate cut to just fit within the cylinders. An angled slit was cut in the plate from the outer edge to near the center and a tapered cutter blade mounted below the plate along one edge of the slit. The cutter blade could be adjusted to change the width of the slit. A metal barrier on the slicing tool segregated approximately 1 cm of soil on the outer edge of the column. The outer edge was segregated because it was the portion most likely to be disproportionately contaminated by any flow down the sides of the column. The plate assembly was attached to a threaded steel rod that passed through the center of a transparent plexiglass plate that rested on the top of the soil columns. The plexiglass plate had a collar attached that ensured that the plate would be centered on the top of the column. Locking nuts on the threaded rod were used to control the depth of cut. A handle was attached to the top of the threaded rod to enable it to be turned by hand to slice a layer of soil.

In the first column sampled, the first 30 mm of soil below the white marker layer were removed in 1 mm increments. Subsequent sampling was conducted by alternating 9 mm and 1 mm layers until the bottom of the column was reached. These layers were then gamma-counted using a well detector. The results from this first set of counts indicated that the ^{134}Cs had not moved more than a few millimeters. The remaining columns were therefore sampled only to a depth of 3 cm in 1 mm slices but were preserved for additional sampling until radiometric analysis confirmed that the activity could not be detected in the deepest layers and that approximately 90% of the total activity could be accounted for in the slices and the discarded outer rim of soil.

Results

Soil Profiles

Profiles 1, 2, and 3 have similar ^{137}Cs activity distributions (Table 14, Fig. 22). The maximum activity in each of these profiles occurs in either the third or fourth cm below the soil surface ranging from 7.12 to 8.66 Bq/kg, and concentration declines approximately exponentially below the peak. The activities drop off sharply within 10 cm of

the soil surface and are at or below minimum detectable concentrations (MDC) within 20 cm below the soil surface. There is no indication that radionuclides have mobilized to any substantial degree within these profiles. It is possible that the increase in ^{137}Cs activities with a slight increase in depth could be the result of leaching. However, it is also possible that the immediate surface is characteristic of a "mixing zone" in which the sandy surface could be depositional, erosional, or both.

The radionuclide distribution in profile 4 appears to be quite different from the other three profiles (Table 14, Fig. 23). An initial interpretation of the profile might be that a pulse of ^{137}Cs activity has moved to a depth of 20 cm. However, a profile inventory of the profile would suggest otherwise. Using an estimated soil bulk density of 1.4 g/cm^3 , the total ^{137}Cs activity in the first three profiles ranged from $503 (\pm 10.0)$ to $797 (\pm 8.0) \text{ Bq/m}^2$, two to four times higher than the profile inventory reported by Collins et al. (2001) for soils located in more equatorial latitudes. The total inventory of ^{137}Cs activity in the fourth profile is almost three times higher than the other three profiles with an activity concentration of $2,263 (\pm 30.1) \text{ Bq/m}^2$ (Table 15).

Pedogenically, three of the four profiles were quite similar. Profiles 1, 2, and 3 had relatively thick sandy surfaces underlain by weakly developed subsoil horizons that are distinguished from the surface by a distinct increase in clay (loamy, mixed, thermic Arenic Ustalfic Haplargid). The clay percentages presented in Table 14 ranged from approximately 2 to 5 % in the upper 6 cm of the profiles. Clay maxima in the underlying subsoil horizons ranged from approximate 12 to 15 %. Profile 4 was unique in that the clay increase in the subsoil was absent (siliceous, thermic Typic Torripsamment). The clay content observed in Table 14 for Profile 4 reflects a uniform distribution of approximately 2 % clay, increasingly only slightly (<1 %) with depth. Because the WIPP soils are dominated by sand and because Cs has an affinity for binding to clay particles, the distribution of clay in the soil could be a factor affecting the distribution of ^{134}Cs in the soil. However, there appears to be no relationship

between the percentage of clay in a soil layer and the concentration of ^{134}Cs in that layer (Fig. 23).

The salinity of these sandy soils is relatively low as illustrated by the low specific conductivity values in Table 13, given the occurrence of saline and gypsiferous outcrops in the vicinity (Chugg et al., 1971). Specific conductivity values in the profiles are generally $<70 \mu\text{S}$ in the upper 40 cm of the soil profiles, with the exception of the immediate surface (upper 2 cm). The concentration of salts at the soil surface reflects the capillary rise and evaporation of moisture in the soil profile. Substantial increases in salinity occur at depths below 50 cm with specific conductivity values exceeding $100 \mu\text{S}$, and is indicative of the effective leaching depth in these profiles. The maximum value was observed in the deepest sample collected from Profile 1, having a specific conductivity value of $190 \mu\text{S}$. To put things into perspective, a soil is not considered saline until the specific conductivity of a saturated paste exceeds $4,000 \mu\text{S}$ (USDA, 1961).

The pH values observed in each of the profiles are uniform throughout for the most part, with pH values generally falling between 6.5 and 7.5. The majority of pH values in Profiles 2, 3, and 4, occur in the pH range of 6.5 to 7.0, while pH values in Profile 1 are dominantly between 7.0 and 7.5. The extremes observed included a low pH value of 6.26 in the second cm-increment of Profile 3 and a high pH value of 7.55 in the deepest horizon sampled in Profile 1.

The distribution of ^{137}Cs seen in profiles 1, 2 and 3 is similar to that reported by Owens et al. (1996) on field and laboratory experiments in Devon, UK. These experiments were carried out to provide empirical information on the fate of ^{137}Cs applied to different soil types over a 10-month period in order to simulate transport of fallout during individual storm events. The results suggested that most of the ^{137}Cs was contained in the top few millimeters of the soil profile, with activity concentrations declining approximately exponentially with depth. The longer-term net effect of ^{137}Cs sorption and movement within the soil profile since fallout began in the

1950's was investigated by examining contemporary soil profiles collected from stable non-eroding locations. Unlike experimentally leached columns, which represent the fate of applied radiocesium after shorter periods of time, the peak in ^{137}Cs concentration was not located at the soil surface, but a few cm below (1.5 to 3.0 cm). The location of the peak below the surface is considered to reflect the interaction of internal soil processes (such as diffusion, translocation and bioturbation) operating within the soil profile since the initiation of ^{137}Cs fallout inputs in the early 1950's, and the fact that the input of ^{137}Cs to the soil surface has declined through time since the mid-1960's.

The most likely explanation for the distribution of profile 4 is deposition and burial of the original profile after it received nuclear fallout during the 1950's and early 1960's. This is not to be unexpected in a landscape dominated by sand dunes. If the ^{137}Cs pulse is bisected at the maximum (assuming a similar profile distribution to the first three profiles), and only the underlying ^{137}Cs activities considered, the total ^{137}Cs is $855 (\pm 17.9) \text{ Bq/m}^2$, which is much more in line with the other three profiles (Table 15). The presence of a buried peak in ^{137}Cs activity was also interpreted as a former surface in studies of an area in southern Zambia reported by Collins et al. (2001).

The activities of ^{137}Cs in profile 4 between the surface and a depth of 15 cm range from 2.419 to 3.309 Bq/kg. This would suggest that the source of deposition is from eroded surface soil higher in ^{137}Cs relative to deeper, subsoil activity concentrations which are generally less than 1.0 Bq/kg, but somewhat lower than the maximum concentrations observed in the near surface environments of profiles 1, 2 and 3. It would be expected that the depositional material would be a combination of material relatively high in ^{137}Cs with materials lower in activity depending on the severity of the erosion from which the depositional sand originated.

The maximum depth at which ^{137}Cs could be detected further supports the argument that the original surface of profile 4 receiving radionuclide fallout has been buried. Profiles 1, 2 and 3 had detectable concentrations of

^{137}Cs down to 20 cm. Profile 4 had detectable ^{137}Cs activity down to 40 cm. If the ^{137}Cs activity peak in profile 4 that begins at a depth of 20 cm is assumed to be the original surface, this profile would have received 20 cm of erosional sand. Accounting for this burial, the depth to which ^{137}Cs has penetrated the buried profile falls in line with the other three profiles at 20 cm.

Discontinuities in the vertical distribution of properties that affect the mobilization of ^{137}Cs , such as pH and salinity, could result in the concentration of ^{137}Cs near the discontinuity. The relative uniformity in pH and salinity in the subsurface soil does not offer an explanation for the occurrence of subsurface maxima in ^{137}Cs concentrations. In addition, the high level of conductivity in the surface layer is indicative of vertical movement of water through evaporation and capillary flow, which would be expected to work in opposition to a downward diffusion of ^{137}Cs .

Column Study

All of the activity contained in the eight soil columns appeared to be in the top few mm of soil, i.e. there was virtually no movement of the ^{134}Cs (Table 16, Fig. 24). Only 56% of the total inventory could be accounted for in Column 1. It was postulated that the remaining inventory might be found in the white marker layer and in the soil of the outer rim, both of which were discarded in the initial sampling process of Column 1. The white marker layer, positioned between the spiked soil and the underlying soil column, was sampled in the remaining columns and all of the soil removed (soil of the outer rim, cover sand above spiked soil, filter covering spiked soil) was preserved for counting. Recovery of the ^{134}Cs spike ranged from 87.3 to 96.5% in the remaining columns. All of the remaining columns also showed a very shallow ^{134}Cs profile, similar to Column 1. It is believed that most of the remaining activity may be attached to soil that could not be removed from the filter paper that was placed directly above the labeled soil. The slight indication of ^{134}Cs movement in the columns could possibly be due to soil colloids moving out of the spiked soil or possibly leaching. Regardless

of the mechanism, the miniscule movement of ^{134}Cs into the soil columns, 7 mm, could in no way account for the ^{137}Cs activity observed in the soil profiles at depths of 30 to 40 mm.

Conclusions

In assessing both the soil profiles and soil columns, it is apparent that leaching and colloidal transport are not major factors in affecting the vertical movement of ^{137}Cs or ^{134}Cs in the soils found in the vicinity of WIPP. The lateral movement of soil by wind erosion is, by far, more responsible for the redistribution of the radionuclides in this ecosystem. The maximum ^{137}Cs activity concentrations observed in the soil profiles, ranging from 6.73 to 8.66 Bq/kg, were slightly higher than the mean values reported by Kirchner et al. (2002) of 3.1 and 6.2 Bq/kg for soil samples sampled to a depth of 2.5 cm. However, the values were well within the range of activity concentrations reported, 0.31 to 15 Bq/kg. Nevertheless, there exists the potential under erosional conditions to expose soils having higher concentrations of ^{137}Cs , and potentially higher concentrations of other

radionuclides such as $^{239,240}\text{Pu}$, than the concentrations currently present in surface soils.

Soil surface concentrations of radionuclides can also affect aerosol concentrations. Arimoto et al. (2002) reported that $^{239,240}\text{Pu}$ activity concentrations in total suspended particle (TSP) samples varied strongly with season (12 to 16 nBq m⁻³), with the highest values generally occurring in spring (March through May, a period commonly typified by frequent dust storms). The source for the plutonium was hypothesized to be from resuspended soil. The authors offered that the results of their study demonstrated that processes affecting the resuspension of $^{239/240}\text{Pu}$ and possibly other substances of concern are of great importance and must be known and quantified if any impacts of WIPP operations are to be properly evaluated. Thus the detection of increased concentrations of radionuclides in the WIPP soils or aerosols should not be automatically attributed to releases from the WIPP.

Table 13. Specific Conductivity and pH data as a function of depth for the four soil profiles

Lower depth (cm)	Specific Conductivity(μS)				pH			
	Profile 1	Profile 2	Profile 3	Profile 4	Profile 1	Profile 2	Profile 3	Profile 4
1	128	200	60	128	7.20	6.58	6.53	6.59
2	71	169	48	89	7.11	6.51	6.26	6.64
3	55	66	47	73	7.12	6.74	6.43	6.73
4	63	53	39	60	6.76	6.71	6.61	6.61
6	55	55	48	50	7.28	6.76	6.58	6.79
8	64	55	45	48	7.25	6.85	6.58	6.94
10	64	50	49	39	6.76	6.78	6.51	6.94
15	56	37	50	39	6.99	6.71	6.68	6.91
20	62	31	54	39	7.19	6.75	6.52	6.88
30	54	41	49	30	7.63	6.62	6.66	6.96
40	54	43	52	30	6.83	6.64	6.67	7.05
50	73	32	62	61	7.39	6.94	6.79	6.84
75	87	33	123	72	7.19	6.90	6.59	7.08
100	125	76		137	7.44	6.74		6.70
125	190			59	7.55			6.94
150				67				6.85

Table 14. Distribution of clay and ^{137}Cs as a function of depth in the four soil profiles

Lower Depth (cm)	Clay (%)				^{137}Cs (Bq/kg)			
	Profile 1	Profile 2	Profile 3	Profile 4	Profile 1	Profile 2	Profile 3	Profile 4
1	3.6	2.0	3.2	1.9	5.261	5.030	6.321	2.419
2	4.1	2.0	4.1	2.0	6.730	6.439	6.983	2.472
3	4.3	3.0	4.4	2.1	7.399	8.296	7.124	2.923
4	3.9	2.9	4.6	2.0	6.722	8.661	5.104	2.612
6	3.9	3.4	5.0	2.0	5.932	6.864	2.126	2.816
8	4.1	3.0	5.4	2.2	2.951	3.029	1.080	2.911
10	4.7	3.7	4.9	2.1	1.378	1.658	0.408	3.052
15	4.7	3.7	5.5	1.9	0.634	0.846	0.437	3.309
20	5.1	3.8	5.6	1.7	0.400	0.240	0.191	4.489
30	5.9	3.9	5.9	2.1	<MDC ¹	<MDC	<MDC	6.727
40	6.8	4.1	6.8	2.2	<MDC	<MDC	<MDC	2.741
50	8.3	4.2	9.0	2.3	<MDC	<MDC	<MDC	<MDC
75	9.3	5.0	12.0	2.5	<MDC	<MDC	<MDC	<MDC
100	10.4	13.0		2.8	<MDC	<MDC		<MDC
125	15.1			2.8	<MDC			<MDC
150				2.8				<MDC

¹<MDC = less than minimum detectable concentration**Table 15. Profile inventories for ^{137}Cs**

Profile	Inventory (Bq/m ²)	Variance (Bq/m ²)
1	725	7.5
2	797	8.0
3	503	10.0
4	2263	30.1
¹ 4	855	17.9

¹below the sample having the maximum concentration

Table 16. Column distribution of ^{134}Cs activity expressed as percent recovery of original spike

Column Depth (mm)	Recovered from Columns (%)							
	C1	C2	C3	C4	C5	C6	C7	C8
Total Recovery	56.4	96.3	88.6	87.3	89.6	96.5	89.6	90.5
Spiked Soil	55.8	85.8	82.8	82.0	84.1	91.3	83.6	84.7
Marker Sand	¹ ND	0.80	1.43	1.37	0.70	2.45	1.35	1.34
1	0.33	0.80	1.25	0.91	0.87	0.77	1.22	0.78
2	² <MDC	0.76	0.59	0.57	0.63	0.46	0.91	0.74
3	<MDC	0.16	0.27	0.21	0.51	0.25	0.37	0.41
4	<MDC	0.12	0.16	0.14	0.19	0.15	0.23	0.30
5	<MDC	0.12	0.08	0.06	0.12	0.06	0.12	0.15
6	<MDC	0.09	<MDC	0.09	0.09	<MDC	0.09	0.11
7	<MDC	0.09	<MDC	0.08	<MDC	<MDC	<MDC	0.06
8	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC
9	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC
10	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC
11	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC
12	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC	<MDC

¹ND – Not Determined²<MDC – Below Minimum Detectable Concentration

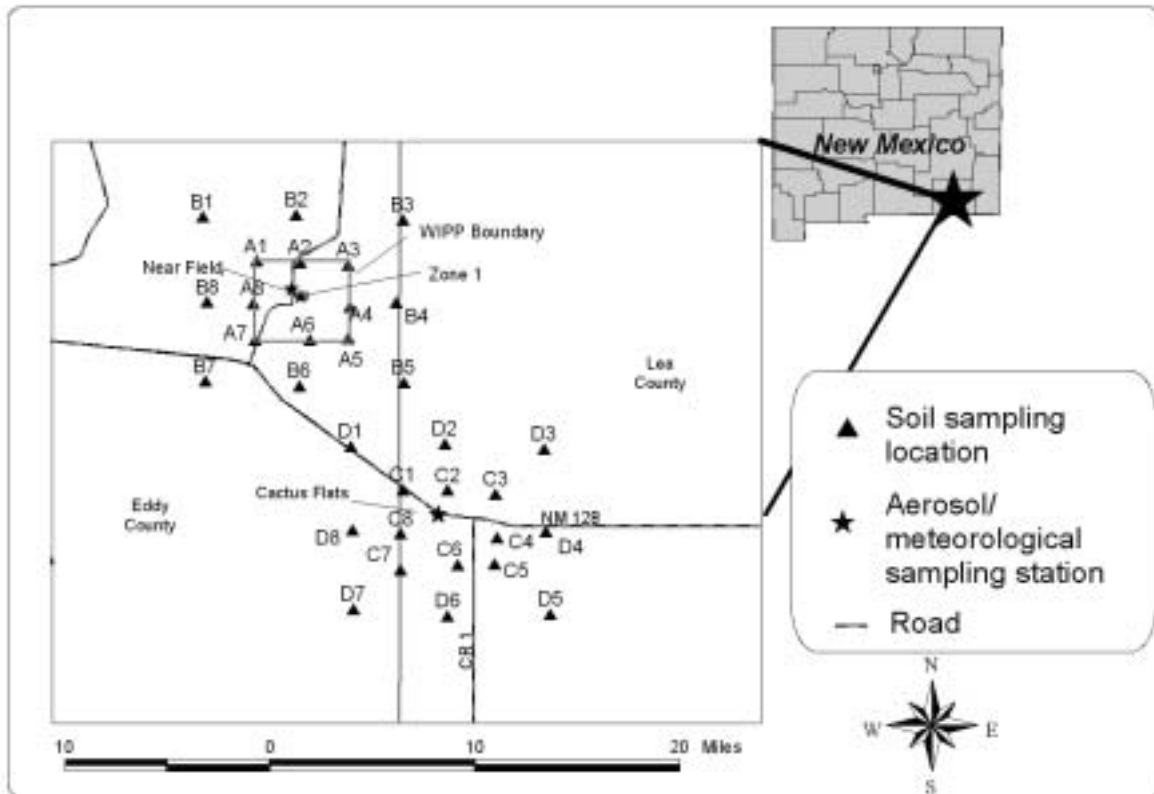


Figure 18. Site location map for profiles excavated at grid nodes A6, A8, C6, and D2



Figure 19. Soil Profile 3 before sampling



Figure 20. Soil Profile 4 after sampling

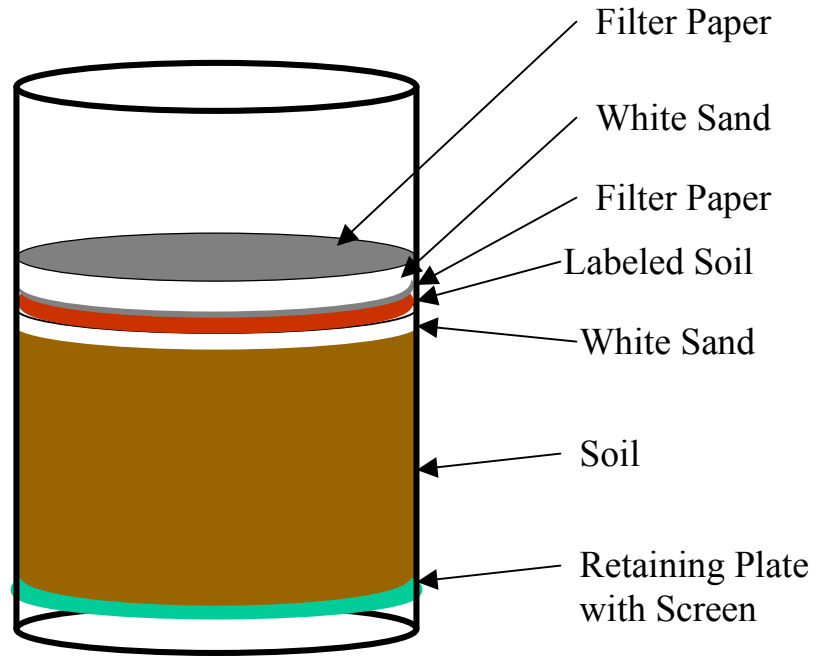


Figure 21. Schematic diagram of soil column construction

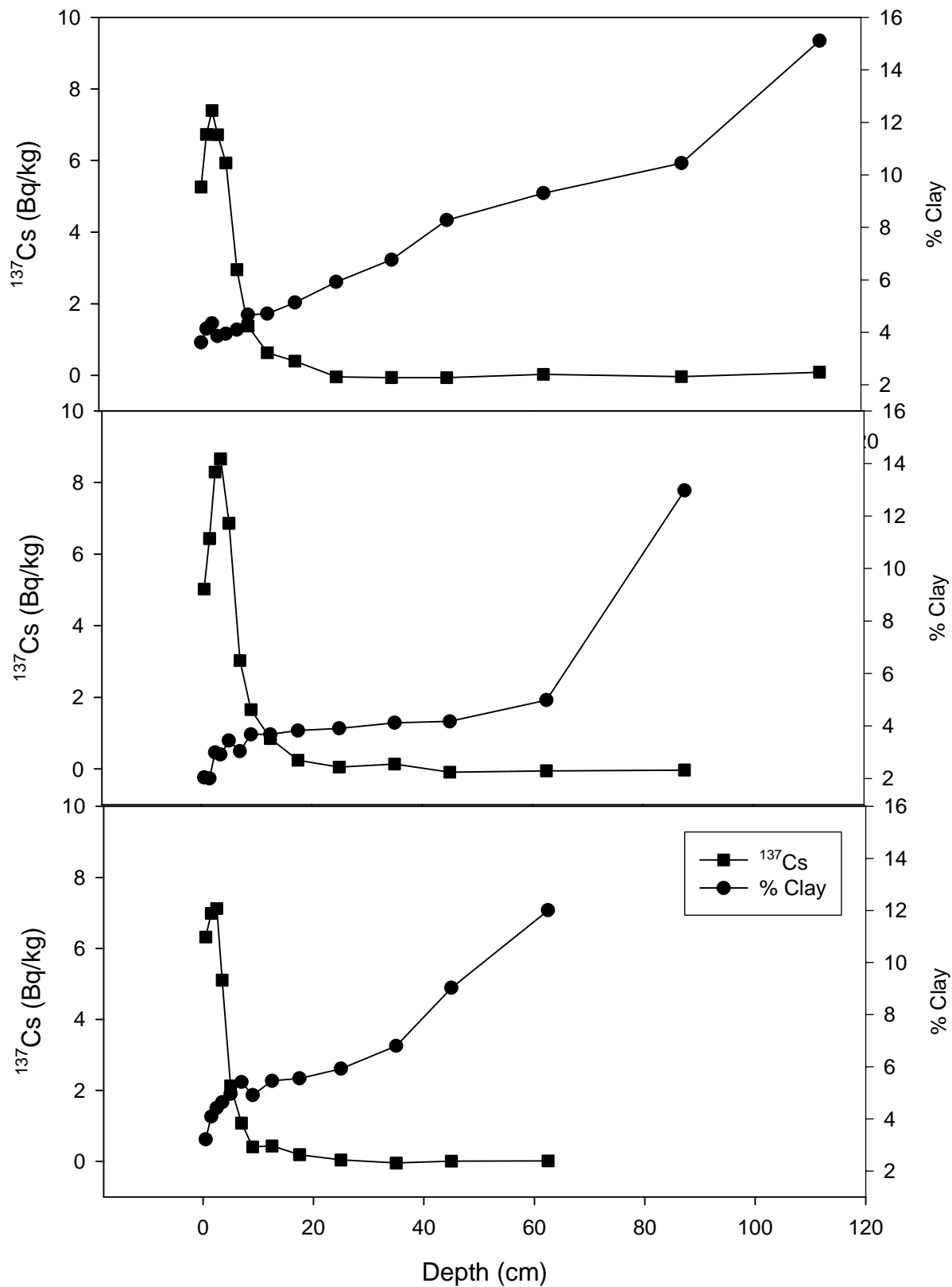


Figure 22. Distribution of ¹³⁷Cs and clay content expressed as a function of depth in Profiles 1, 2 and 3

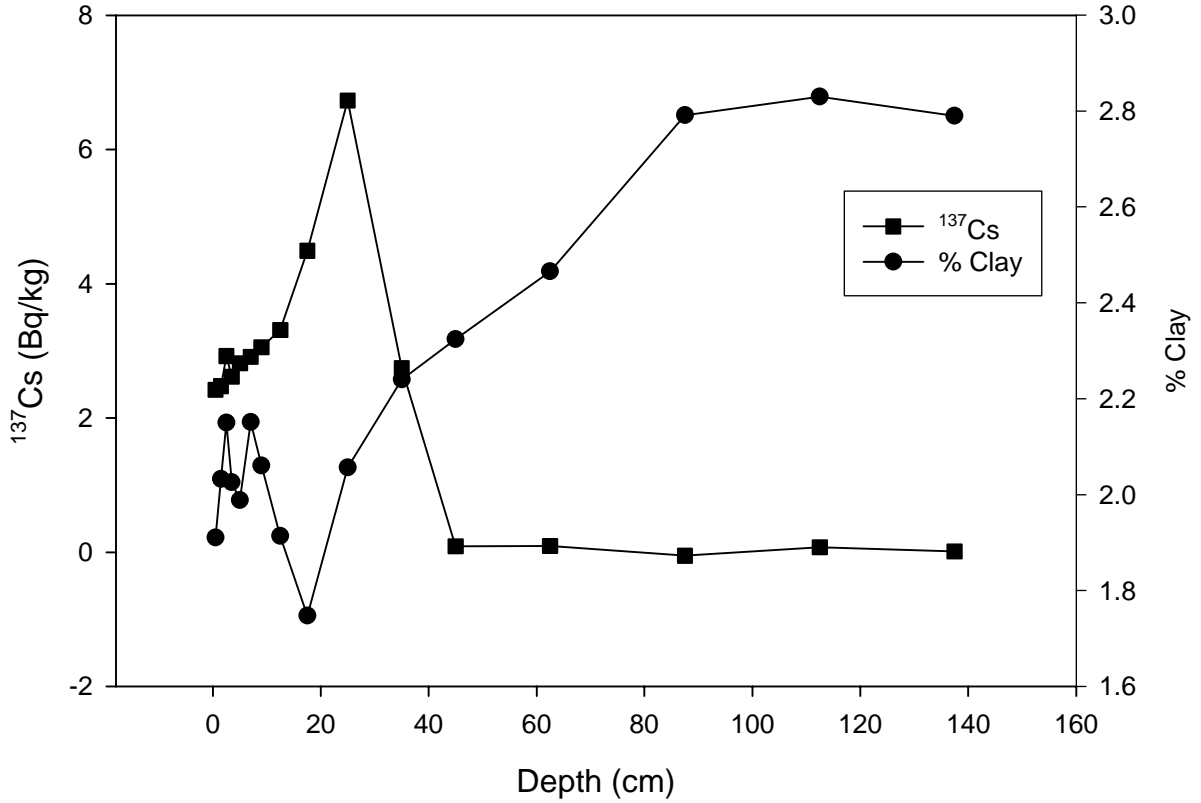


Figure 23. Distribution of ¹³⁷Cs and clay content expressed as a function of depth in Profile 4

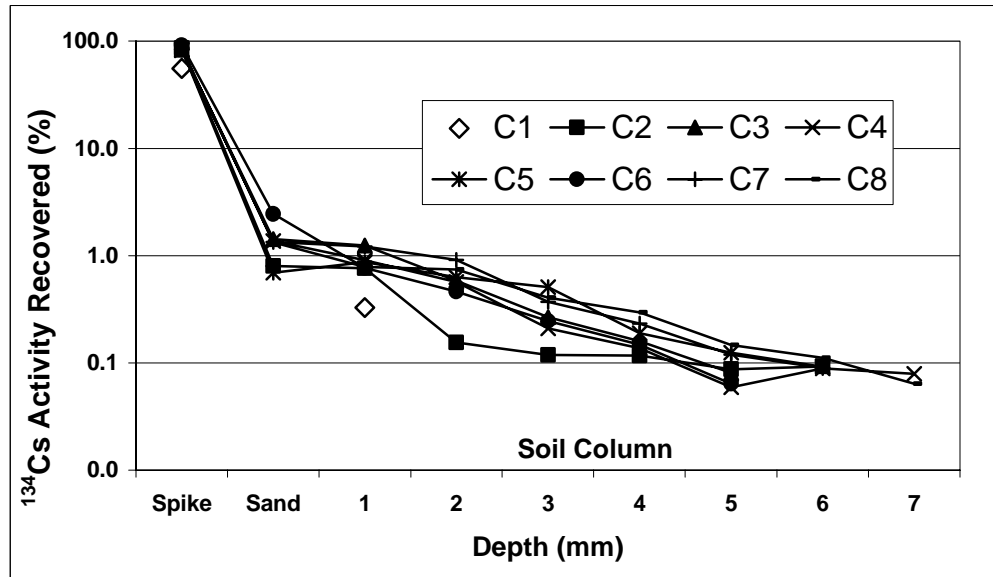


Figure 24. Distribution of ¹³⁴Cs activity as a function of depth in the soil columns