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Hydrologic Response and Radionuclide Transport Following Fire at Semiarid Sites

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ABSTRACT

Infrequent, high-impact events such as wildfires, droughts, biological shifts, floods, and mechanical disturbances can greatly change land surfaces, including vegetative cover and soil characteristics, which in turn can trigger high rates of hydrologic erosion and associated transport of sediments and sediment-sorbed contaminants. Where persistent soil contamination exists, infrequent mobilization of contaminants may dominate in determining long-term risks to human and ecological receptors. Among these infrequent events, fire stands out as having the capacity to cause large increases in sediment transport. This study measured runoff, sediment yield, and mobility of sediment-sorbed contamination (^{137}Cs) on burned and unburned plots at the Waste Isolation Pilot Plant, New Mexico (WIPP), and the Rocky Flats Environmental Technology Site, Colorado (RFETS). Results showed that ^{137}Cs transport from burned plots was up to 22 times greater than that from unburned plots at WIPP and 4 times greater at RFETS. Associated runoff was up to 12 times greater on burned plots at WIPP and sediment yields up to 6 times greater. Further, ^{137}Cs concentrations in transported sediments were enriched compared with parent soils (expressed as *enrichment ratio*) by a factor of 2.3 at WIPP, and 1.3 at RFETS. However, enrichment ratios were not significantly different in sediments from burned and unburned plots. Our results provide new data on the effects of fire on the transport of sediment-sorbed contaminants, and demonstrate that rare events such as fire can greatly increase contaminant mobility.

Risk levels posed by contaminants in the environment are largely determined by basic processes such as erosion and sediment transport that can mobilize contaminants from source areas and carry them to receptors. These erosion and contaminant transport processes vary over time and may be subject to large shifts induced by infrequent, high-impact events such as wildfire, drought, biological activity, mechanical disturbances, floods, and extraordinary wind or precipitation events. However, such events are not routinely incorporated into contaminant risk assessment, or only cursorily so, mainly due to lack of information on how they affect basic contaminant mobility. Further, infrequent disturbances that induce episodes of accelerated contaminant movement are not included in most risk assessment models that typically assume steady-state surface conditions and thus may underpredict risk over long time frames (Whicker et al., 1999).

Specific examples of infrequent events that triggered concerns about increased contaminant mobility occurred in the summer of 2000 when forest and rangeland wildfires burned in the western USA at three nuclear weapons facilities. Wildfires occurred at the Los Alamos National Laboratory, New Mexico (approximately 3000 ha burned), the Hanford Site, Washington (approximately 24 300 ha burned), and the Idaho National Engineering and Environmental Laboratory, Idaho (approximately 16 000 ha burned). These wildfires burned on and near radiological waste areas and raised heightened concerns over post-fire transport of radiological contamination by wind and water. As a result, new risk assessments were initiated to assess if accelerated transport of contaminants was occurring in post-fire conditions, and if risk levels were raised by fire. Further, these risk assessments are being conducted in the context of an increasingly accepted view that at nuclear weapons sites, radiological and nonradiological wastes will remain, posing potential risk to humans and the environment for tens or even hundreds of thousands of years (National Research Council, 2000). Over these long time frames, wildfire can reoccur many times in semiarid landscapes (Swet-

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Abbreviations: RFETS, Rocky Flats Environmental Technology Site; WIPP, Waste Isolation Pilot Plant.

nam and Betancourt, 1998) and in addition, current risk of wildfire occurrence appears to have increased in some locations due to land use and fire suppression policies that have allowed excessive buildup of forest and rangeland fuels (Covington et al., 1994; Mast et al., 1999; Moore et al., 1999). Taken together, these suggest that fire will occur many times in the future at contaminated waste sites in semiarid locations, causing mobilization and transport of radionuclides and raising potential for risks to human and ecological receptors.

Fire stands out among infrequent events that cause increased contaminant mobility because of its capability to greatly reduce vegetation ground cover and alter soils (DeBano et al., 1998). Following fire, a burned hillslope has less ground cover, consequently allowing for less impedance of overland flow during rain and thereby increasing runoff. In addition, less vegetative cover reduces protection of soil from compaction and sealing effects cause by rain, thereby reducing infiltration and further contributing to runoff (DeBano et al., 1998; Bryan, 2000). Ground cover also protects against erosive forces by shielding the soil surface from direct transfer of kinetic energy from raindrops (interill erosive forces) and from shear stress of overland flow (rill erosive forces) (Lane et al., 1997; Weltz et al., 1998; Bryan, 2000). The cumulative effect of fire on most landscapes is increased runoff, erosion, and sediment transport that can mobilize and transport trace metals, nutrients, and radionuclides sorbed to sediments.

While many studies have focused on post-fire increases in runoff and sediment transport, fewer studies have coupled these effects with transport of sediment-sorbed constituents. Increases in concentrations of metals such as manganese, copper, and zinc have been observed in runoff sediments after fire (Auclair, 1977; Chambers and Attiwill, 1994), as well as increases in levels of nutrients such as potassium, phosphorous, and nitrogen (Tiedemann et al., 1979; Shindler, 1980; Belillas and Roda, 1993; Parra et al., 1996). Fallout ^{137}Cs can be concentrated in ash after fire and the solubility of ^{137}Cs in ash decreased compared with its solubility when bound in unburned material (Amiro et al., 1996). A decreased inventory of ^{137}Cs after fire at a site in a Canadian boreal forest was attributed in part to transport away from the study site by runoff (Paliouris et al., 1995). Independent of fire effects, studies on transport of contaminants by sediment movement often focus on fallout radionuclides such as ^{137}Cs, which is ubiquitous in environmental soils and sediments, relatively easily to measure, and typically tightly bound to sediments. In fact, the strong affinity of many radionuclides for soil provides a reliable method of using them as tracers to study soil erosion processes (McHenry and Ritchie, 1977; Whicker and Schultz, 1982). Conversely, erosion processes driven by wind and water can play a key role in controlling the long-term fate and effects of soil actinides (Lane and Hakonson, 1982; Watters et al., 1983; Lane et al., 1986; Hakonson and Lane, 1993).

In addition, few studies have investigated the effect of particle sorting by runoff and its influence on enrichment of contaminant concentrations in sediments transported by water (Lane and Hakonson, 1982). As used in this study, this effect is expressed as an *enrichment ratio*, and relates the concentration of a contaminant being transported by sediment particles to the concentration in the parent soil. Enrichment of sediment-sorbed constituents results from preferential detachment and transport of fine-grained particles that are generally more chemically active (i.e., typically contain greater concentrations of sorbing constituents) compared with coarser particles (Massey and Jackson, 1952; Graf, 1971; Menzel, 1980; Lane and Hakonson, 1982). Organic matter is included along with mineral particles in the types of sediments that can be preferentially entrained in runoff and contribute to enrichment (Flanagan and Nearing, 1990).

Lane and Hakonson (1982) analyzed sediment transport rates by particle size classes in alluvial channels and derived the following expression:

$$\text{Enrichment ratio} = \frac{\Sigma[Cs(i) \cdot Qs(i)]}{Cs\Sigma[Qs(i)]} \qquad [1]$$

where $Cs(i)$ is the contaminant concentration of particle size class i, $Qs(i)$ is sediment transport (mass/time) for particles in size class i, and Cs is mean contaminant concentration in parent soils over all particle size classes. This equation suggests that if all particle size fractions in transport are in the same proportion as they exist in the parent material, an enrichment ratio of unity results. Typically, however, smaller-sized particles are preferentially entrained by runoff, and higher enrichment ratios occur (Lane and Hakonson, 1982; Flanagan and Nearing, 1990; Quinton et al., 2001). Enrichment ratios are important for estimating contaminant concentrations in runoff moving away from waste sites. However, they are generally unavailable for sites where risk assessment is performed and entirely unavailable for disturbance scenarios such as wildfire.

In summary, few studies have focused on the effects of fire on erosion and transport of sediment-sorbed contaminants, particularly the effects of removal of vegetative ground cover on transport of contaminants. Also lacking is information on enrichment ratios for sediment-sorbed contaminants and specifically the effects of fire on these enrichment ratios. The main objective of our study was to quantify transport of sediment-sorbed contaminants following fire and relate this to fire-induced changes in ground cover. We sought to quantify fire's effects on both contaminant transport rates and on the enrichment of contaminants in runoff.

MATERIALS AND METHODS

Study Areas

Studies were conducted during 1998 at the Waste Isolation Pilot Plant (WIPP) and during 1999 at the Rocky Flats Environmental Technology Site (RFETS). The main difference between these sites was their soil textures, with WIPP having sandy soils (approximately 91% sand) and RFETS having clayey soils (approximately 44% clay).

The WIPP study site is located about 15 km east of the main WIPP facility and about 60 km from Carlsbad, NM. The

Table 1. Surface and soil characteristics of study plots at the Waste Isolation Pilot Plant (WIPP) and Rocky Flats Environmental Technology Site (RFETS).

	WIPP		RFETS	
Particle size distribution, %				
Sand	91.1 (±2.7)		34.3 (±5.6)	
Silt	3.8 (±2.0)		21.2 (±4.6)	
Clay	5.1 (±1.7)		44.4 (±6.8)	
Dry bulk density, g cm^{-3}	1.34 (±0.2)		1.30 (±0.3)	
Organic matter, %	0.4 (±0.2)		2.6 (±0.6)	
CEC†, $cmol_c$ kg^{-1}	8.8 (±2.2)		27.5 (±2.6)	
Random roughness‡, cm^2	0.7		1.8	
	Unburned	Burned	Unburned	Burned
Canopy cover, %				
Forbs	12	0	25	0
Grass	69	0	40	0
Shrub	1	0	5	0
None	17	0	27	0
Standing dead	1	0	3	0
Ground cover, %				
Bare soil	22	46	28	36
Gravel	9	8	3	3
Rock (>20 mm)	0	0	1	1
Nonpersistent litter	33	35	3	18
Persistent litter	1	0	33	16
Basal vegetation	35	11	32	26

† Cation exchange capacity.
‡ Expressed as standard deviation of height measurements.

study area has a semiarid climate with an average annual precipitation of about 300 mm, of which most occurs during summer thunderstorms (U.S. Department of Energy, 1997). Average annual temperature is 17°C, with daily mean minimum and maximum temperatures of 8.8 and 29.9°C, respectively. Surface geology is dominated by stabilized sand dunes overlaying Mescalero caliche (U.S. Department of Energy, 1997). Soils at the WIPP study site were classified as sand to loamy sand with relatively low organic matter and cation exchange capacity (Table 1). The study area was probably subjected to grazing by cattle in the past, although evidence of such was not visible. The average slope of the plots was 6.2% (±0.6%). The dominant grass was black grama [*Bouteloua eriopoda* (Torr.) Torr.].

The RFETS study area is located about 1 km from the southeastern boundary of RFETS near Westminster, CO. The site has a semi-arid climate with an average annual precipitation of 370 mm, of which approximately 40% occurs in the spring and approximately 30% in the summer (Tysdal, 2000). Average annual temperature is 9.7°C, with daily mean minimum and maximum temperatures of −8.8 and 31.2°C, respectively. Site soils are clay to clay loam with relatively high cation exchange capacity (Table 1). Light grazing by horses had occurred recently at the study area. The average slope of the rainfall simulation plots is 9.1% (±0.5% standard deviation). Vegetation is shortgrass steppe with dominant species including blue grama [*Bouteloua gracilis* (Kunth) Lag. ex Griffiths, nom. illeg.], western wheatgrass [*Pascopyrum smithii* (Rydb.) A. Löve], smooth brome (*Bromus inermis* Leyss.), and intermediate wheatgrass [*Elytrigia intermedia* (Host) Nevski subsp. *intermedia*].

Experimental Design

Six plots of 3.0 by 10.7 m (10 by 35 ft) were established in pairs at each site. Vegetation cover and organic litter were removed from one plot of each pair by a controlled grass fire at WIPP and a controlled grass fire aided by a propane torch at RFETS. Vegetation canopy cover, ground cover, and surface roughness were characterized with 245 point frame measurements per plot (Levy and Madden, 1933). Soil textures were determined by pipette analysis. Soil bulk density measurements were made at six locations per site and three soil samples (5 cm depth) were taken per plot prior to each rainfall simulation to determine antecedent soil moisture content.

Rainfall simulations, in lieu of natural storms, were used to provide control and repeatability of experimental treatments. Rainfall simulations were conducted about 2 d following burn treatments. A Swanson 16-m-diameter, rotating-boom rainfall simulator was used to apply rainfall of approximately 60 mm h^{-1} on plot pairs (Swanson, 1965). The drop size distribution from the rainfall simulator nozzles was similar to that from natural rainfall, but the drops impacted the ground surface with about 80% of the kinetic energy of natural rain (Swanson, 1965). Large rainfall simulators have been used extensively for evaluating hydrologic and erosional responses of crop and rangeland sites (Renard, 1985; Simanton et al., 1990; Lane et al., 1986) and at locations having contaminated waste sites to investigate runoff transport of radionuclides (Hakonson, 1999; Hakonson et al., 1986; Nyhan et al., 1990; Essington and Romney, 1986).

Three rainfall simulations were performed on each plot pair as follows: a 1-h rainfall application at about 60 mm h^{-1} (labeled *Dry* run for its antecedent moisture condition) followed by a 24-h recovery, then two 0.5-h rainfall events at 60 mm h^{-1} separated by a 0.5-h recovery period (labeled *Wet* and *Very Wet* runs, respectively). Rain applied to each plot totaled about 120 mm.

The downslope end of each plot was fitted with an end plate and gutter to collect runoff and sediment. Runoff flow measurements were made at a calibrated flume using a bubble gage flow meter (ISCO, Lincoln, NE). Samples of runoff (water and sediment) were taken at 2- to 4-min intervals at the flume exit during each simulation to provide for calculating sediment and radionuclide yields. The texture of sediment samples was determined by pipette analysis and by wet sieving after shaking the samples for 1 h to break up aggregates formed during drying. Concentrations of ^{137}Cs in sediment and soil were measured using an HPGE gamma ray spectrometer (EG&G ORTEC, Oak Ridge, TN) with counting times sufficiently long to reduce counting error to less than 12%.

Table 2. Average runoff, sediment, and fallout ^{137}Cs yields from unburned and burned cover treatments at the Waste Isolation Pilot Plant (WIPP) and Rocky Flats Environmental Technology Site (RFETS).

	Antecedent moisture	Runoff		Sediment yield		^{137}Cs yield	
		Unburned	Burned	Unburned	Burned	Unburned	Burned
	%	mm mm^{-1} rain		kg ha^{-1} mm^{-1}		Bq ha^{-1} mm^{-1}	
WIPP							
Dry	0.0 (1.4)†	0.00 (0.00)	0.00 (0.00)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
Wet	5.6 (1.6)	0.00 (0.00)	0.04 (0.05)	0.0 (0.0)	1.3 (1.6)	0.0 (0.0)	33.2 (44.9)
Very Wet	13.3 (0.8)	0.02 (0.02)	0.16 (0.13)	0.8 (1.1)	3.5 (1.5)	4.5 (3.9)	62.2 (35.4)
Mean		<0.01 (<0.01)	0.07 (0.06)	0.3 (0.3)	1.6 (1.0)	1.5 (1.3)	31.8 (26.7)
RFETS							
Dry	12.5 (1.4)	0.29 (0.04)	0.37 (0.07)	1.8 (0.3)	4.3 (0.8)	36.4 (16.6)	106.1 (22.1)
Wet	28.8 (2.4)	0.43 (0.09)	0.49 (0.17)	1.9 (0.1)	4.3 (0.3)	52.5 (12.6)	202.3 (7.0)
Very Wet	35.4 (2.8)	0.67 (0.12)	0.66 (0.10)	2.2 (0.2)	5.1 (1.9)	72.4 (24.5)	276.2 (91.8)
Mean		0.46 (0.49)	0.50 (0.11)	1.9 (0.2)	4.6 (1.0)	53.8 (17.9)	194.9 (40.3)

† ±Standard error.

RESULTS

Ground Cover Changes from Burning

At both study sites, the major effects of burning were the complete removal of canopy cover, decreases in ground cover, and corresponding increases in percentages of bare soil (Table 1). At WIPP, the percentage of bare soil increased by 24% (from about 22 to 46%). Of the ground cover that remained after burning, most was nonpersistent litter such as ash or moveable detritus (35% of plot area), with lesser amounts of basal vegetation (11%) and gravel (8%). At RFETS, burned plots also had complete removal of canopy cover, and the percentage of bare soil increased by 8% (from about 28 to 36%). After burning, most ground cover was basal vegetation, mostly root crowns (26% of plot area), with additional ground cover of persistent and nonpersistent litter (16 and 18%, respectively), and rock and gravel (1 and 3%, respectively).

Runoff

Burned plots generated earlier and greater amounts of runoff than unburned plots at WIPP (Table 2). Average time of rainfall application before runoff began was 81 min (during Wet run) on burned plots compared with 106 min (during Very Wet run) on unburned plots. Further, of the applied rainfall at WIPP, an average of 6.84% (±6.14%) exited burned plots as runoff, while unburned plots averaged much less runoff at 0.57% (±0.53%). At RFETS, increases in runoff from burned plots were smaller with average percent runoff from burned plots of 50.3% (±11.4%) compared with 46.1% (±8.5%) from unburned plots. At RFETS, average times to runoff initiation were about 5 min for both treatments. Runoff increases on burned plots generally correlated with ground cover removal ($r = 0.53$ at WIPP, $r = 0.49$ at RFETS); however, a larger population of tests on plots having a larger range of ground cover removal is needed to define runoff–ground cover relationships on burned surfaces. Increases in runoff from burned plots could have also been caused by water-repellent soils created during fire. However, this was thought to not be the case, particularly at RFETS where runoff initiation times were essentially the same for both burned and unburned plots. Had water-repellent soils been created at RFETS, quicker runoff times would have occurred on burned plots.

Comparing between WIPP and RFETS sites, runoff from sandy soils at WIPP was comparatively low with only 5 of 18 rainfall events resulting in measurable runoff compared with 18 of 18 at RFETS. Average runoff as a percent of total rainfall was 3.7% (±6.4%) for all plots at WIPP, compared with 48.2% (±15.4%) at RFETS (96% infiltration at WIPP vs. 52% at RFETS). Large differences were also seen in the average times from beginning of rainfall application to beginning of runoff, with 94 min occurring at WIPP and 5 min at RFETS. In addition, RFETS had higher antecedent moisture content (12.5–35.4%) compared with WIPP (0.0–13.3%), and runoff amounts generally increased as antecedent soil moisture increased during sequential runs (Table 2). These differences between sites were mainly associated with differences in soil texture, with the clayey soils at RFETS providing less infiltration and consequently more runoff than sandy soils at WIPP ($r = -0.95$ correlation between infiltration and percent clay for all plots); however, the slightly greater slope at RFETS may have also contributed to the greater runoff observed there.

Sediment Yields

Relative to the effects of burning, sediment yields at WIPP averaged 0.27 ± 0.34 kg ha^{-1} mm^{-1} from unburned plots, and a factor of six higher from burned plots at 1.63 ± 1.05 kg ha^{-1} mm^{-1}. Similarly at RFETS, average sediment yields were 1.94 ± 0.20 kg ha^{-1} mm^{-1} for unburned plots, and higher at 4.56 ± 1.01 kg ha^{-1} mm^{-1} for burned plots. Sediment yields generally correlated with percent bare soil at both sites ($r = 0.80$ at WIPP, $r = 0.52$ at RFETS). Comparing between sites, average sediment yields from all plots at RFETS were 3.4 times greater than those measured at WIPP. Similar to runoff, sediment transport increased with increasing antecedent soil moisture as sequential runs were conducted.

Cesium-137 Transport

The average yield of ^{137}Cs from burned study plots at WIPP was about 22 times higher than the amount transported from unburned plots. At RFETS, average ^{137}Cs transport was about 4 times higher for burned plots. Yields of ^{137}Cs generally correlated with the percentage of bare soil on study plots ($r = 0.69$ at WIPP, $r = 0.73$ at RFETS), and consistent with previous studies, correlated strongly with sediment yields ($r = 0.95$ at WIPP, $r = 0.96$ at RFETS). Comparison between yields at WIPP and RFETS shows 7.5-times-greater average ^{137}Cs transport at RFETS, primarily associated with the greater runoff and sediment yields at RFETS and the higher percentage of fine-grained material in RFETS sediments.

The amount of ^{137}Cs transported from study plots was compared with the inventory of total ^{137}Cs in the top 5 cm of parent soil for purposes of determining loss rates of sediment-sorbed radionuclides. Only small fractions of the total ^{137}Cs plot inventories were lost during testing, even though simulated rainfall applications of 60 mm h^{-1} for 1 h represent large storms of greater than a 100-yr recurrence interval at WIPP and about a 12-yr recurrence interval at RFETS. For example, during Wet and Very Wet runs (total of 60 mm rain) the amount of ^{137}Cs lost from the top 5 cm was less than 0.1% at both study sites. During Wet and Very Wet runs at WIPP, ^{137}Cs yields were 1498 Bq ha^{-1} while parent soils to a depth of 5 cm contained approximately 5.3 MBq ha^{-1}. At RFETS, ^{137}Cs corresponding yields were 9051 Bq ha^{-1} while parent soils contained approximately 22.3 MBq ha^{-1}. These inventory loss rates fall within the range (0.02–3.4%) reported in a study on ^{137}Cs transport from hillslopes at the Nevada Test Site (Essington and Romney, 1986). However, considerable variation in inventory loss rates are expected among different sites as a result of differing distributions of contaminants in soil, with greater initial loss rates where contaminants are near the surface compared with deeper distributions.

Enrichment Ratios

Average enrichment ratios for burned and unburned treatments at WIPP were 2.6 (±2.1) and 2.1 (±1.2), respectively, and for corresponding treatments at RFETS 1.4 (±0.3) and 1.2 (±0.1) (Fig. 1). The slightly higher enrichment ratios for burned treatments at both sites were not significantly different from unburned treatments ($p < 0.05$). Average enrichment ratios at WIPP were nearly two times greater than those at RFETS. Maximum enrichment ratios of 3.8 and 1.5 were measured for individual simulator runs at WIPP and RFETS, respectively.

Study data shows how particle sorting can contribute to enrichment where, at WIPP for example, ^{137}Cs was preferentially bound to <50-μm clay- and silt-sized particles in both sediments and soils (in sediment, 42.2 and 13.7 Bq kg^{-1} for <50- and ≥50-μm fractions, respectively; in soils 37.4 and 5.1 Bq kg^{-1}, respectively). When runoff occurred, the percentage of the clay- and silt-sized particles increased to 14.3% (±3.6%) in sediments compared with 8.9% (±1.9%) in the parent soils. This sorting of particles increased amounts of fine-grained material in sediments, and thus increased concentrations of sorbed ^{137}Cs (17.5 Bq kg^{-1} sediment vs. 7.7 Bq kg^{-1} parent at WIPP, and 38.2 Bq kg^{-1} sediment vs. 28.9 Bq kg^{-1} parent at RFETS). Sorting was not limited to specific size fractions, in fact, 51% of the enrichment at WIPP occurred within the ≥50-μm size fraction (i.e., average sand particles in sediments were smaller and had higher associated ^{137}Cs concentrations). This study did not distinguish between organic and mineral particles and a portion of the observed enrichment may have been associated with organic material that, similar to mineral particles, can be preferentially entrained by runoff and can also sorb with cations such as ^{137}Cs.

Enrichment ratios for both treatments at RFETS increased with increases in antecedent soil moisture ($R^2 = 0.92$). This appears to be related to increases in the proportion of fine-grained material in successive runs as antecedent soil moisture increased (Fig. 2 and Table 2). This result is somewhat counterintuitive in that successive runs also had greater runoff flow rates, which are expected to entrain more coarse material and reduce enrichment. However, we speculate that the observed increases are related to the high clay content of soils at RFETS. Specifically, breakdown of clay-dominated soil aggregates during increasing saturation as rainfall simulations progressed allowed for greater availability of fines, and thus greater enrichment. A similar effect was not observed at WIPP where soils contain little clay.

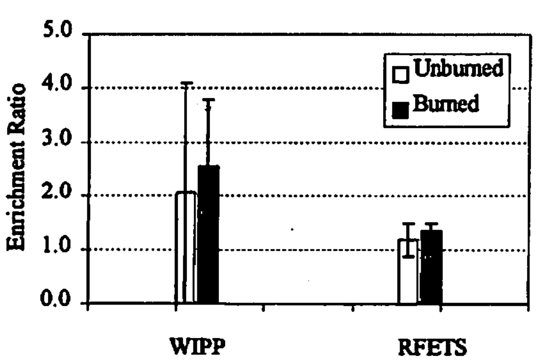

Fig. 1. Average enrichment ratios of ^{137}Cs in sediments from burned and unburned plots at the Waste Isolation Pilot Plant (WIPP) and Rocky Flats Environmental Technology Site (RFETS) (error bars of ±1 standard deviation).

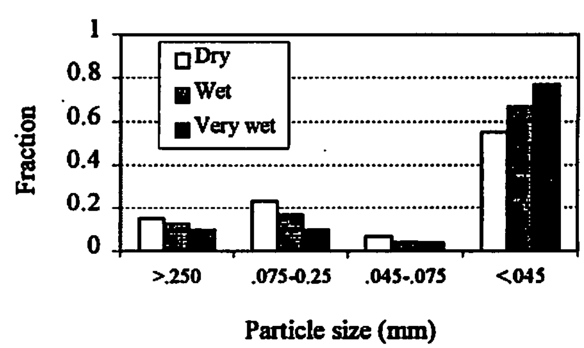

Fig. 2. Changes in sediment particle size fractions in runoff from burned plots at Rocky Flats Environmental Technology Site (RFETS) as antecedent soil moisture levels increased. Average antecedent soil moisture levels were 12.5% for the Dry run, 28.8% for the Wet run, and 35.4% for the Very Wet run.

DISCUSSION

Increased Cesium-137 Transport Related to Fire

Our results show that fire at arid sites can cause many times greater sediment transport and corresponding greater mobility of sorbed contaminants. Burned plots at WIPP yielded up to 22 times greater ^{137}Cs than unburned paired control plots. At both WIPP and RFETS, transport of ^{137}Cs was closely related to sediment yield and, in addition, was in proportion to percentage ground cover removed by fire. More specifically, burning increased the amount of bare soil subject to rainfall splash effects (related to interrill erosion), and increased the amount of unprotected soil subject to overland flow (related to rill erosion). Burning also caused slight decreases in average infiltration of 6 and 4% at WIPP and RFETS, respectively, probably due in part to less ponding and impedance where ground cover was removed.

While ^{137}Cs transport correlated with ground cover removal at the two sites studied, the percentage of ground cover removed by fire at these grass-dominated sites was relatively low compared with observations following fire in other ecosystems (24 and 8% at WIPP and RFETS, respectively). In general, percentages of ground cover reductions by fire are expected to be less in grass-dominated systems compared with many brush and forest systems where fire can burn hotter and at longer residence times. Our results that relate reductions in ground cover to contaminant transport on grass-dominated surfaces imply that greater transport of sediment-sorbed contaminants can occur in brush and forest ecosystems where fire can remove greater percentages of ground cover.

In addition, the data from WIPP show how fire can shift the contaminant transport response on a hillslope from practically no transport in an unburned condition—even when subjected to very large storms—to the occurrence of ^{137}Cs transport after fire at earlier times during storms, and from smaller rainfall amounts. For some ecosystems that are usually not prone to sediment runoff, the WIPP data imply potentially large increases in sediment-sorbed contaminant transport after fire where practically no transport would occur without fire.

Increased Cesium-137 Transport Related to Soil Texture Differences

In addition to the effects of ground cover removal by fire, our results show that soil texture was important in determining amounts of runoff, sediment yield, and associated ^{137}Cs transport. Soil textures at the two sites were very different, with the clayey soils at RFETS (approximately 44% clay) allowing for much less infiltration, and associated greater runoff amounts and longer runoff durations, than the sandy soils at WIPP (approximately 91% sand). Yields of ^{137}Cs were positively correlated to percent clay in plot soils (r = 0.69), and the clayey RFETS plots yielded about 7 times more ^{137}Cs than WIPP plots. Those plots at RFETS having both clayey soil and removal of vegetative cover by fire had the largest yields of ^{137}Cs of all plots, approximately 4 times greater than paired unburned plots at RFETS, approximately 6 times greater than burned plots at WIPP, and approximately 130 times greater than unburned plots at WIPP.

Inventory Loss Rates and Enrichment Ratios

While our results show greater transport of sediment-sorbed ^{137}Cs on burned plots, the percentage of ^{137}Cs removal from the inventory in the parent soils was small for all plots. The maximum percentage of ^{137}Cs inventory lost from the top 5 cm of soil was less than 0.1% even after application of 120 mm of rain, which is equivalent to infrequent, large storms at the study sites. While our data show that the rate of transport of contaminants can be greatly affected by fire, and while these rates of transport may be significant to downstream receptors, it appears difficult for even large storms on burned surfaces to remove large portions of the inventory of contaminants dispersed in soils. This indicates the potential for waste areas having dispersed contaminants to continue to act as source areas over long time frames. However, our study plots had relatively small slopes and low percentages of ground cover removal compared with what is possible at other locations where greater inventory losses may occur. In addition, inventory loss rates depend highly on the depth distribution of contaminants in soils. Where contaminants are concentrated on the ground surface, greater initial loss rates may occur.

When assessing contaminant transport relative to downstream receptors, the concentrations of contaminants in runoff and sediment are often more important than loss rates from waste sites. Our results show that the sediment leaving from plots was enriched in ^{137}Cs concentrations compared with parent soils, with average enrichment ratios of 2.3 ($\pm$1.7) at WIPP and 1.3 ($\pm$0.7) at RFETS. However, no statistically significant differences in enrichment ratios were measured between burned and unburned treatments at either site. This result, which was not found to have been previously documented in literature, suggests that while fire at a site can deposit ash and thus increase levels of fallout radionuclides in soils and in associated runoff (Paliouris et al., 1995; Amiro et al., 1996), the enrichment of radionuclides in runoff occurs in the same ratio before and after burning. However, this study used low-intensity fire and did not determine if high-intensity fire may affect enrichment ratios. One possible cause of greater enrichment after fire was thought to be associated with breakdown of soil aggregates by soil heating, which could provide for more fines available for erosion, and consequently more enrichment. Our results of low severity fire experiments, however, did not support the occurrence of this effect.

The enrichment ratios calculated at RFETS increased with increasing soil moisture on both burned and unburned plots. As soil moisture levels increased, the proportion of fines in the runoff increased, resulting in proportionally greater enrichment of ^{137}Cs. The reason for the increase of fines with antecedent soil moisture was not determined, but may be related to the break-

Table 3. Approximate enrichment ratios for nutrients and plutonium associated with various land use and locations in the United States (adapted from Lane and Hakonson, 1982).

Land use and location	Approximate enrichment ratios: Mean	Range	Tracer measured
Cropland, USA†	4.5	2.5–7.4	nitrogen
	3.6	2.6–6.0	phosphorus
Rangeland, USA†	2.6	1.1–6.7	nitrogen
	7.1	2.7–17	phosphorus
Cropland, USA‡	1.6	1.1–2.5	fallout plutonium
Pasture, USA‡	2.3	0.8–4.0	fallout plutonium
Mixed cropland, USA§	2.5	1.2–4.0	fallout plutonium in perennial river
Semiarid, USA¶	5.5	1.4–13.3	waste effluent plutonium in ephemeral streams
Agricultural, Europe	1.7	0.4–5.0	fallout ^{137}Cs
This study#			
Chihuahua Desert grass/shrub, USA (WIPP)			
Unburned	2.1	0.6–3.5	fallout ^{137}Cs
Burned	2.6	1.4–4	
Prairie shortgrass steppe, USA (RFETS)			
Unburned	1.2	0.5–1.8	fallout ^{137}Cs
Burned	1.4	0.63–2.0	

† Small agricultural watersheds (5.2–18 ha) at Chickasha, Oklahoma.
‡ Small agricultural watersheds (2.6–2.9 ha) near Lebanon, Ohio.
§ Great Miami River (drainage area = 1401 km²) at Sidney, Ohio.
¶ Los Alamos watersheds (176–15 000 ha) near Los Alamos, New Mexico.
# Waste Isolation Pilot Plant (WIPP) and Rocky Flats Environmental Technology Site (REFETS).

down of clay-dominated soil aggregates as soil moisture increased during rainfall simulations. No similar effect was seen at WIPP where soils contained only small percentages of clay.

Enrichment ratios calculated in this study were consistent with those for nutrients and radionuclides that have been estimated for agricultural and rangeland sites (Table 3). Enrichment ratios that vary from 2.6 to 7.1 have been measured for soil nutrients and radionuclides in runoff from small agricultural areas. Ratios measured for fallout plutonium in runoff from agricultural watersheds range from about 1.6 to 2.5 while ratios in ephemeral stream channels at Los Alamos, New Mexico, ranged from 1.4 to 13.3 with a mean of 5.5.

One key question is, what degree of correlation exists between enrichment ratios derived from small plots to those for larger scales such as watersheds? Average enrichment ratios measured in this study compared well with those measured for plutonium and nutrients in a variety of site and radionuclide source conditions (Table 3). However, measures of erosion processes are highly scale dependent (Lane et al., 1997), and enrichment on a watershed scale may increase or decrease substantially compared with the small plot scale.

In summary, this study demonstrated transport of sediment-sorbed contaminants (represented by ^{137}Cs) in amounts up to 22 times greater following fire compared with unburned conditions. Burned plots consistently produced more ^{137}Cs than unburned plots, even though percentage of ground cover removed was relatively small on our grass-dominated plots compared with removal that can occur by fire in brush and forest ecosystems. Burning of vegetative ground cover at WIPP served as a catalyst that shifted conditions from practically no contaminant transport in large storms, to contaminant transport after fire that occurred earlier and in greater amounts. Enrichment of fines, and associated enrichment of sorbed radionuclides, was measured; however, burning at our plots did not affect the degree to which ^{137}Cs was enriched in sediments. Our results imply potentially large increases in radionuclide transport rates after wildfires, particularly where large percentages of ground cover are removed, and highlight the need to incorporate infrequent, high-impact events such as fire into long-term risk assessment.

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