DISSERTATION

HYDROLOGIC EROSION AND REDISTRIBUTION OF ^{137}CS FOLLOWING FIRE AT SEMIARID SITES

Submitted by

Mathew P. Johansen

Department of Radiological Health Sciences

In partial fulfillment of the requirements

for the Degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Summer 2002

UMI Number: 3063997

ProQuest Information and Learning Company
300 North Zeeb Road
P.O. Box 1346
Ann Arbor, MI 48106-1346

COLORADO STATE UNIVERSITY

June 19, 2002

WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY MATHEW P. JOHANSEN ENTITLED "HYDROLOGIC EROSION AND REDISTRIBUTION OF ^{137}CS FOLLOWING FIRE AT SEMIARID SITES" BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

Committee on Graduate Work

Advisor

Co-Advisor (if applicable)

Department Head

ABSTRACT OF DISSERTATION

Hydrologic Erosion And Redistribution Of ^{137}Cs Following Fire At Semiarid Sites

Of the few natural processes that reconcentrate dispersed environmental contaminants, landscape fire stands out as having potential to rapidly reconcentrate and redistribute contaminants, and do so on large scales. This study was conducted to quantify changes in concentration of a widely dispersed environmental contaminant –fallout ^{137}Cs– in soils and runoff following landscape fires.

Measurement of changes in fallout ^{137}Cs concentration after fire was conducted in grassland, shrubland, and forest ecosystems. At each site, burned and unburned plots (3.0 x 10.7m) were subjected to simulated rainstorms using a 16m rotating-boom rainfall simulator. Burned conditions ranged from a controlled, low-severity fire in grassland, to a high-severity wildfire in ponderosa pine forest.

A series of reconcentration and redistribution processes occur during and after fire. Ashing of biomass during fire resulted in elevated ^{137}Cs concentrations in burned soils, from 46% in grassland to about 300% higher in ponderosa forest where large amounts of ash were deposited. After fire, ^{137}Cs concentrations in runoff from burned plots were elevated one order of magnitude higher than in runoff from unburned plots, and two orders of magnitude higher in post-fire runoff from a small watershed. The greatest surface water transport of ^{137}Cs from plots, up to 11.6 KBq ha^{-1} per mm rainfall, occurred after severe burning in ponderosa pine forest where up to 80% vegetation cover was removed compared to yields from grassland and shrubland that were an order of magnitude less. ^{137}Cs increases in runoff were associated with increased sediment transport after fire, and, further, these sediments were enriched in ^{137}Cs by factors ranging from 1.4 to 2.9 compared to parent soils. However, enrichment ratios (^{137}Cs in runoff sediments compared to parent soils) were not affected by burning.

Study results provide evidence of order of magnitude increases in reconcentration and redistribution of a sorbed contaminant following fire that has relevance to a wide range of ecosystem dynamics, geophysical, fire management, and risk assessment studies.

Mathew P. Johansen
Department of Radiological Health Sciences
Colorado State University
Fort Collins, CO 80523
Summer 2002

ACKNOWLEDGEMENTS

This work was guided by Dr.'s F. Ward Whicker, Thomas E. Hakonson, Shawki A. Ibrahim, James C. Ascough II, and David D. Breshears. To work with these fine scientists is truly an honor.

Field experiments were performed by Dr. Hakonson, Jim Stone, Roger Simanton, Jeffery Stone, Audrey Hayes, David Breshears, Karen Agogino, and Susan Johnson. Geoff Warren, Heather Asmus, and Wendy Kuhne provided laboratory data analysis. Michael Ebinger provided use of lab space and equipment. Great thanks to Dave Englert who provided comparative data. Special thanks to my co-authors of journal articles, Dr.'s Whicker, Hakonson, Breshears, Simanton, and Stone. Thanks to Joe Vozella for supporting me start-to-finish by sponsoring my Fellowship with the U.S Department of Energy. This work was made possible by a grant from the U.S. Department of Energy, Environmental Management Science Program (EMSP98-4). I have appreciated continued help and support from Julie Asmus.

A big gushy thanks to Andrea who not only helped in many, many ways, but told me she is proud of this accomplishment. Special appreciation to Trisha and Caryn who helped with field work, but more importantly, inspire their dad.

PREFACE

Each Chapter in this dissertation is presented as a stand-alone document suitable for submission for publication. Chapter one focuses on hydrologic response following fire and provides a foundation for the following chapters on that focus on contaminant redistribution. Chapter one was published in *Hydrologic Processes* 15:2953-2965. Chapter two focuses on hydrologic response and ^{137}Cs redistribution following fire in grassland and shrubland (Rocky Flats Environmental Technology Site, CO, and the Waste Isolation Pilot Plant, NM). Chapter two was published in *Journal of Environmental Quality* 30:2010-2017. Chapter three focuses on concentration and redistribution of ^{137}Cs after wildfire in ponderosa pine forest at the Los Alamos National Laboratory, NM. Chapter three also provides some summary information synthesizing results from the grassland, shrubland, and forest ecosystems of all three study sites. Because the chapters are presented as stand-alone documents, readers my notice some repetition between chapters.

TABLE OF CONTENTS

Chapter 1 Introduction 1

Chapter 2 Post-fire runoff and erosion from rainfall simulation:

Contrasting forest with shrubland and grassland 6

Abstract 6

Introduction 7

Materials and Methods 10

Results 13

Discussion 21

Chapter 3 Hydrologic response and radionuclide transport

following fire at semiarid sites 29

Abstract 29

Introduction 30

Materials and Methods 34

Results 38

Discussion 45

Chapter 4 Pulsed redistribution of contaminants following fire:

Cesium-137 reconcentration and erosion in forest runoff 51

Abstract 51

Introduction 52

Materials and Methods 56

Results 59

Discussion and Conclusions 69

Chapter 5 Comprehensive discussion and conclusions 74

References 80

Appendix 89

LIST OF TABLES

Table 2.1 Surface characteristics of study plots in ponderosa pine near Los Alamos, New Mexico 14

Table 2.2 Sediment yields from rainfall simulation studies on burned and unburned plots 19

Table 3.1 Surface and soil characteristics of study plots at Waste Isolation Pilot Plant (WIPP) and Rocky Flats Environmental Technology Site (RFETS) 36

Table 3.2 Average runoff, sediment, and fallout ^{137}Cs yields from unburned and burned cover treatments at WIPP and RFETS 44

Table 3.3 Approximate enrichment ratios for Nutrients and Plutonium associated with various land use and locations in the United States . . . 49

Table 4.1 ^{137}Cs concentrations in soil, ash, and vegetation 60

Table 4.2a Average ^{137}Cs concentrations in runoff from rainfall simulation 62

Table 4.2b ^{137}Cs concentrations in runoff from the Pajarito Watershed 62

Tabel 4.3 Enrichment ratios of ^{137}Cs in runoff sediments from rainfall simulation 66

LIST OF FIGURES

Figure 2.1 Sediment yields and runoff ratios for burned and unburned plots (three runs per plot: dry, wet, and very wet) 16

Figure 2.2 Runoff hydrographs on burned and unburned plots 17

Figure 2.3 Sediment yields from rainfall simulations on comparably sized burned and unburned plots over a range of vegetation types 22

Figure 2.4 Conceptual relationship between sediment yield with percentage bare soil (a), conceptual response from grassland and ponderosa pine ecosystems (b) 27

Figure 3.1 Average enrichment ratios of ^{137}Cs in sediments from burned and unburned plots at WIPP and RFET 42

Figure 3.2 Changes in sediment particle size fractions in runoff from burned plots at RFETS as antecedent soil moisture levels increased 43

Figure 4.1 ^{137}Cs concentrations at the ground surface before and after the Cerro Grande wildfire 61

Figure 4.2 Concentrations of ^{137}Cs and total suspended solids in runoff 64

Figure 4.3 Post-fire concentrations of ^{137}Cs in unfiltered runoff during cumulative rainfall 65

Figure 4.4 Sediment yields and ^{137}Cs yields from rainfall simulations on burned and unburned plots over a range of vegetation types 67

Figure 4.5 Enrichment ratios in runoff sediments from rainfall simulation in a range of ecosystem types 68

Figure 4.6 ^{137}Cs reconcentration processes and transport pathways

following fire 69

Chapter 1. Introduction

Certain processes, such as bioaccumulation, have been shown to reconcentrate dispersed environmental nutrients and contaminants. Such reconcentration processes stand out in contrast to the many natural processes that dilute concentration levels such as dispersion, chemical change, and radioactive decay (Whicker and Shultz, 1982; Kendall and McDonnell, 1998; Peles et al., 2000). Of the processes that reconcentrate, those that occur during and after landscape fires are thought to stand out in their capability to concentrate and redistribute nutrients and contaminants on large scales and in rapid time frames (Debano et al., 1998). However, data is lacking in this area, particularly with respect to the reconcentration and redistribution of contaminants. Examination of fire as an agent in concentrating and mobilizing environmental contaminants has relevance to a wide range of current interests including: ecosystem dynamics (Baird et al., 1999; Thomas et al., 1999); effects of increased potential for landscape fires from climate change (IPCC, 2001; Piñol et al.,1998) and from increased forest fuel buildup (Sackett and Haase, 1999; Mast et al., 1999; Moore et al., 1999); relative to forest burning as a land use practice (Kauffman et al., 1993); and relative to mobility of dispersed contaminants, particularly radionuclides (Kashparov et al., 2000; Johansen et al., 2001b).

Better information on fire-induced concentration of contaminants can be particularly useful in performing risk and dose assessments. Routine risk and dose assessments, and the models used for these assessments, may under-predict risks and doses over long time frames when they do not include key fundamental mechanisms such as fire that induce episodes of accelerated contaminant movement (Whicker et al., 1999). The importance of wildfire in mobilizing radionuclides was highlighted in the summer of 2000, when separate forest and rangeland wildfires burned a total of about 43,300 ha at three nuclear weapons sites located in the western U.S. These wildfires burned on and near radiological waste areas in Idaho, New Mexico, and Washington states and heightened concerns over post-fire transport of radiological contamination by wind and water. After the Cerro Grande fire burned over portions of the Los Alamos National Laboratory in New Mexico in May of 2000, the U.S. Environmental Protection Agency issued a statement warning against the use of post-fire ash in vegetable gardens citing concerns over concentrated contaminants including the fallout radionuclide strontium-90. A subsequent, more detailed, risk analysis calculated that risk levels through the water pathway had been elevated above pre-fire conditions. However, in this case, dilution of post-fire runoff after it had entered a major waterway had reduced risks to low levels (awaiting RAC report).

These issues are being raised in the context of an increasingly accepted view that radiological and non-radiological wastes will remain at nuclear weapon production sites, 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 (Swetnam and

Betancourt, 1998). 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 fuels (Covington et al., 1994; Mast et al., 1999; Moore et al., 1999), implying continued reconcentration and redistribution of environmental contaminants by fire into the future.

Ecosystem redistribution and concentration processes have been extensively quantified using tracers, such as fallout radionuclides that are widely dispersed in world ecosystems (Kendall and McDonnell, 1998). However, few such measurements have been made following landscape fires that can greatly amplify both the amounts and rates of redistribution. In a Canadian boreal forest, elevated concentrations of ^{137}Cs were found in surface soils at an area that had been burned years before, indicating reconcentration from burned vegetation to deposited ash, (Paliouris et al., 1995). Amiro et al., 1996, found that on a unit weight basis, the ash deposits after fire were enriched in cesium from 4 to 25 times during field burns, and over two orders of magnitude in laboratory burns compared to the original vegetation. These studies suggest potentially large concentration factors of fallout radionuclides in ash and soils after forest wildfires.

Subsequent to deposition of concentrated radionuclides in ash on the ground surface, erosion and transport of concentrated radionuclides by surface water runoff are expected to occur. In the Paliouris et al, 1995 study, the total ^{137}Cs inventory of the burned area was less than that of an unburned control area, indicating redistribution away from burned areas. The authors attributed ^{137}Cs losses from the burned area to volatilization and fly-ash processes during fire, and runoff after fire.

The importance of surface water runoff as a post-fire redistribution process was further suggested by ^{137}Cs inventories in both burned and unburned areas that showed highest inventories in the organic soil layer (up to 55 KBq ha^{-1}) at the ground surface where erosion processes occur, compared to the lesser above-ground inventory (up to 14 KBq ha^{-1}). However these inventory measurements were made years after fire occurrence, not during transport processes, thus considerable uncertainty remains regarding rates of reconcentration and redistribution, and the underlying factors controlling these process on post-fire landscapes.

Many studies have indicated that the magnitude of hydrologic response (and associated contaminant transport) after fire depends largely on the effectiveness of the fire to remove groundcover (Simanton et al., 1990; Emmerich and Cox, 1994; Hester et al., 1997; Emmerich, 1998; Wilson, 1999). Ground cover aides infiltration by impeding overland flow, thereby increasing the frequency and depth of ponding, and by protecting the soil surface from compaction and from sealing effects that can inhibit infiltration (Wilcox et al., 1988; Bryan, 2000). Ground cover also protects against detachment and entrainment of soil particles 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; Bryan, 2000).

Part of the increase in ^{137}Cs transport during runoff and erosion is thought to be from enrichment of ^{137}Cs in runoff. Enrichment can occur during erosion and runoff through preferential entrainment of fine particles such as clay-sized particles which have greater affinity for cationic contaminants such as ^{137}Cs (Graf, 1971; Menzel, 1980; Lane and Hakonson, 1982; Watters et al., 1983; Lane et al., 1986;

Hakonson and Lane, 1993; Weigand et al., 1998). Lacking are measurements of the enrichment effect following landscape fires where soil heating can reach high temperatures and thus potentially affect enrichment processes by changing soil characteristics.

The above studies suggest a series of processes during and after fire consisting of ashing of biomass, amplified surface water erosion and sediment transport, and enrichment during surface water erosion that result in concentration of fallout radionuclides and subsequent rapid redistribution away from burned areas. However, lacking are data that quantify these processes after landscape fires. Particularly lacking are data on concentrations of fallout radionuclides in surface water runoff and the underlying factors that govern radionuclide enrichment and transport after fire. Also unavailable is information relating specific fire effects, such as removal of vegetation, to radionuclide transport.

The primary objective of this study was to quantify changes in the concentrations of a fallout radionuclide—^{137}Cs— in soils and runoff sediments after fire in ecosystems ranging from grassland, to shrubland, to forest. Specifically, data is used from rainfall simulation plots and monitoring data from adjacent areas, including one watershed, to evaluate post-fire changes in (1) ^{137}Cs concentrations in soils, (2) ^{137}Cs concentrations in surface water runoff including sediments (both changes in concentration and duration of those changes), and (3) associated changes in the enrichment of ^{137}Cs in transported sediments compared to their parent soils. Special emphasis is placed on underlying factors, such as fire-removal of vegetation, and their effects on transport of contaminants.

Chapter 2. Post-fire Runoff and Erosion from Rainfall Simulation: Contrasting Forest with Shrubland and Grassland

2.1 Abstract

Rainfall simulations allow for controlled comparisons of runoff and erosion among ecosystems and land cover conditions. Runoff and erosion can increase greatly following fire, yet there are few rainfall simulation studies for post-fire plots, particularly after severe fire in semiarid forest. We conducted rainfall simulations shortly after a severe fire (Cerro Grande) in ponderosa pine forest near Los Alamos, New Mexico, U.S., which completely burned organic ground cover and exposed unprotected soil. Measurements on burned plots showed 74% of mineral soil was exposed compared to an estimated 3% exposed prior to the fire. Most of the remaining 26% surface area was covered by easily moveable ash. Rainfall was applied at 60 mm hr^{-1} in three repeated tests over two days. Runoff from burned plots was about 45% of the total 120 mm of applied precipitation, but only 23% on the unburned plots. The most striking difference between the response of burned and unburned plots was the amount of sediment production; burned plots generated 25 times more sediment than unburned plots (76 kg ha^{-1} and 3 kg ha^{-1} respectively per mm rain). Sediment yields were well correlated with percentage bare soil ($r = 0.84$). These sediment yields were more than an order of magnitude greater than nearly all comparable rainfall simulation

studies conducted on burned plots in the U.S., most of which have been in grasslands or shrublands. A synthesis of comparable studies suggests that an erosion threshold is reached as the amount of soil exposed by fire increases to 60-70%. Our results provide sediment yield and runoff data from severely burned surfaces, a condition for which little rainfall simulation data exists. Further, our results contrast post-fire hydrologic responses in forests with those in grasslands and shrublands. These results can be applied to problems concerning post-fire erosion, flooding, contaminant transport, and development of associated remediation strategies.

2.2 Introduction

Hydrological processes such as runoff and erosion at the hillslope scale are sensitive to changes in land surface properties that can be greatly altered by fire. In particular, reductions in the amount of ground cover and changes in soil characteristics resulting from fire can produce amplified hydrologic responses, especially for fires that are intense (DeBano et al., 1998; Sackett and Haase, 1999). Intense fires are of particular concern for the semiarid ponderosa pine forests of the western U.S., where the probability of such fires is currently high. Fires occurred historically in these forests at frequent intervals, and probably at low intensities, through the latter part of the 1800s (Swetnam et al., 1999). Beginning in the late 1800s, suppression of fire—which resulted initially from grazing and later from direct fire fighting efforts—led to excessive build-up of canopy fuel and organic material on the ground surface (Campbell et al., 1978; Covington et al., 1994; Sackett and Haase, 1999; Mast et al., 1999; Moore et al., 1999). Consequently, risk of catastrophic, intense fires has increased greatly along with the consequent potential for such fires to alter runoff and erosion processes drastically. Understanding these post-fire processes is fundamental for assessing risks and determining remediation strategies associated with flooding, erosion, sediment transport, nutrient dynamics, and contaminant transport (Foster and Hakonson, 1987; DeBano et al., 1998).

Hillslope runoff and erosion rates before and after fire depend on numerous factors that include surface properties, slope, rainfall intensity and amount, and the size and composition of the area of interest. Consequently, it is important to control some of these factors during field studies to allow evaluation of the effects of other parameters and to allow comparisons among different studies. An effective means of exerting this control is through the use of rainfall simulators, which allow for repeatable rainfall amounts on plots of a given size and slope. Hydrologic response varies with spatial scale (Seyfried and Wilcox, 1995; Lane et al., 1997; Reid et al., 1999), and hence rainfall simulations performed on larger plots are more likely to be representative of hillslope-scale processes than those performed on plots at the scale of vegetation patches. Large rainfall simulators that can be applied to plots of ~3 m x ~10 m have been used extensively for evaluating hydrologic and erosional responses from hillslopes (Renard, 1986; Lane et al., 1986; Hakonson et al., 1986; Hakonson, 1999). Although rainfall simulation studies are limited in their ability to replicate rainfall patterns and energies completely, they allow for more direct comparison among different ecosystems and different site conditions (e.g., burned and unburned) than studies based on natural precipitation (Renard, 1986; Lane et al., 1997).

Of the reported rainfall simulation studies that focused on post-fire hydrologic response, most indicate that the magnitude of hydrologic response depends largely on the effectiveness of the fire to remove groundcover (Simanton et al., 1990; Emmerich and Cox, 1994; Hester et al., 1997; Emmerich, 1998; Wilson, 1999). Ground cover aides infiltration by impeding overland flow, thereby increasing the frequency and depth of ponding, and by protecting the soil surface from compaction and from sealing effects that can inhibit infiltration (Wilcox et al., 1988; Bryan, 2000). Ground cover also protects against detachment and entrainment of soil particles 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; Bryan, 2000).

Another factor besides ground cover that can affect post-fire runoff and sediment yield is soil alteration resulting from fire. In particular, water repellent soils can develop during fire when organic matter at the soil surface is volatilized. The volitalized organic matter can move downward as vapor and condense as a hydrophobic coating on soil particles, thereby reducing infilitration (DeBano, 1981). Such reductions in infiltration are thought to change hydrologic response, producing greater than initial runoff from sites where hydrophobicity is a factor than from sites where it is not (Hester et al., 1997; Robichaud, 2000). Other effects of fire on soil properties include combustion of organic matter (Hester et al., 1997; Marcos et al., 2000) and reductions in soil aggregate sizes (Emmerich and Cox, 1994), both of which can affect soil resistance to erosive forces.

Few rainfall simulation studies are reported for conditions following intense fire. Intense fire can result in greatly amplified hydrologic responses because of the potential for large reductions in ground cover and alteration of soils. The limited number of post-fire rainfall simulation studies that have been conducted to date encompass a diverse set of ecosystems and various burn severities, but a synthesis of these studies is lacking. We sought to address these gaps, in part by measuring runoff and sediment following an intense fire in ponderosa pine forest –the Cerro Grande fire that occurred near Los Alamos, northern New Mexico, U.S., in May, 2000—and by contrasting our results with other post-fire rainfall simulation studies. The Cerro Grande fire caused extensive exposure of soil that had been previously protected by duff. Our major objective was to use rainfall simulation to quantify runoff and sediment yield as related to ground cover following severe fire in this ponderosa pine forest. In addition, we compared our hillslope-scale results for ponderosa pine forests to results from other studies at similar spatial scales in different ecosystems that included grasslands, shrublands, and forests. Our results, which document amplified hydrologic responses in runoff and erosion at the hillslope scale following intense fire in ponderosa pine forest, contrast with relatively small hydrological responses observed in grasslands and shrublands.

2.3 Materials and Methods

Study site

The study was located within the Los Alamos National Laboratory on the Pajarito Plateau, thirty-five miles northwest of Santa Fe, NM, U.S. The study was conducted predominantly in intercanopy spaces of ponderosa pine forest. The site has a semi-arid, temperate mountain climate with an average annual precipitation of ~50 cm, with a major portion of the precipitation falling in July and August (Bowen, 1990). Average annual temperature is 10.0 °C, with daily mean minimum and maximum temperatures ranging from –6 to 29.8 °C. Prior to the Cerro Grande fire, there was extensive duff beneath trees and mixed grass and Gambel oak (*Quercus gambelii*) in open interspaces. The hydrology of a nearby site in a ponderosa pine forest has been extensively characterized with respect to actual precipitation events. At this companion study site, surface runoff may comprise 3% to 11% of the annual water budget (Wilcox et al., 1997). Subsurface shallow water flow, or interflow, is another component of water loss documented for this system. This flow has been observed only in response to winter snowmelt, for which it can comprise as much as 20% of snowpack (Wilcox et al, 1996, 1997; Wilcox and Breshears, 1998). The soil water dynamics for this site reflect the two large seasonal inputs of precipitation and a period of high evapotranspiration following each (Brandes and Wilcox, 2000), with soil evaporation extending to a depth of ~10 cm and downward flux of ~0.02 cm per year (Newman et al., 1997). Surficial soil on the study plots is loam consisting of 40% sand, 47% silt, and 13% clay.

The fire history of the Pajarito Plateau has been extensively documented (Foxx, 1984; Allen, 1989; Swetnam and Baisan, 1996; Touchan et al., 1996; Allen and Breshears, 1998; Swetnam et al., 1999), with three landscape-scale fires occurring within the last 25 years: the La Mesa fire in 1977 (Foxx, 1984), the Dome fire in 1996 (Cannon and Renaeu, 2000) and the Cerro Grande fire in 2000. Several studies document amplified hydrologic responses at the watershed scale to these fires: White and Wells (1984) and White (1996) for the La Mesa fire, and Cannon and Renaeu (2000) for the Dome fire. The most recent of these fires, the Cerro Grande fire, burned about 17,400 ha with the most severely burned areas located in dense forests, and created water repellent soil conditions in some locations (Interagency BAER, 2000).

Experimental design

Four plots, each 3.03 m by 10.7 m (10 by 35 feet), were established in areas with relatively uniform vegetation and surface slopes. Two plots were established on a severely-burned area, and two control plots were established approximately 150 m away in an unburned area. The soil series, scale, slope, and amount and intensity of rainfall received were similar for the control plots and the burned plots. Vegetation canopy cover, ground cover, and surface roughness were characterized with a point frame having 245 pin drops per plot (Levy and Madden, 1933). Average plot slope was ~4.5% for unburned plots and ~7.0% for burned plots. Soil bulk density was measured in the field at six locations along plot edges. Water drop tests (DeBano, 1981) were used to assess the extent of water repellent soils on nearby sites and on the study plots. Soil texture was measured using wet sieve analysis.

For rainfall simulation, a Swanson-type (Swanson, 1965) 16-m diameter, rotating-boom rainfall simulator was used to apply rainfall of 60 mm hr^{-1}, which represented 100-year recurrence interval at Los Alamos for a 1-hour storm event. 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).

Three rainfall simulations were performed on each plot pair as follows: a one-hour rainfall application at 60 mm hr^{-1} (labeled *Dry* run for its antecedent moisture condition) followed by a twenty-four hour between-run interval, a second rainfall event, one-half hour in duration (*Wet* run), followed by a one-half hour between-run interval, and a final third event (*Very wet* run). Rain applied to each plot totaled about 120 mm.

Data were normalized to correct for minor variations in actual rainfall intensities and durations associated with the simulations. Three soil samples (to a depth of 5 cm) were taken adjacent to each plot just prior to each simulation to measure antecedent soil moisture. Antecedent moisture was 4.6% by weight (± 0.02%) for Dry runs, and progressed to 13.5% (± 0.02%) for Wet runs, and 18.9% (± 0.01%) for Very wet runs.

The downslope end of each plot was fitted with an end-plate and gutter to collect and channel runoff and sediment through a calibrated flume where runoff depth was measured every 2-4 min using a bubble gage flow meter (ISCO, Lincoln, NE). Flow depth measurements were used to construct runoff hydrographs. Samples of runoff (water and sediment) were taken every 2-4 min at the flume exit during each simulation to facilitate calculation of sediment yields.

2.4 Results

Changes in plot characteristics

Organic ground cover on burned plots at Los Alamos decreased greatly as a result of fire, which consumed litter, duff, and ground vegetation. The amount of bare soil increased from an estimated 3% (± 3%) prior to burning, to 74% (± 8%) after burning. The ground cover on the remaining 26% of plot area was mostly moveable, non-persistent ash which was counted as ground cover when found in deposits sufficiently thick (approximately 1 cm) to be easily distinguished from the blackened mineral soil. Only 6% of the post-fire surface had persistent ground cover such as rock, persistent litter, and basal vegetation, including burned root crowns (Table 2.1).

Table 2.1 Surface characteristics of study plots in ponderosa pine near Los Alamos, New Mexico

Characteristics	*Unburned*		*Burned*	
	Plot 1	*Plot 2*	*Plot 3*	*Plot 4*
Ground Cover †				
Bare soil (%)	38	58	69	80
Rock (>20mm) (%)	1	0	2	1
Non-persistent litter (%)	18	18	27	13
Persistent litter (%)	17	8	0	0
Vegetation (%)	26	17	2	7
Surface Roughness ‡ (cm^2)	4.7 (1.4)	3.7 (1.1)	6.0 (1.8)	4.2 (1.3)

† Average from 5 transects per plot, with 49 measurements per transect. Standard deviations of transect averages for plots 1-4 were 8, 5, 7, and 3 respectively.
‡ Expressed as standard deviation of height measurements from ground surface along a transect to a reference elevation. Standard deviation between transects is given in parentheses

Prior to burning, the thickness of the layer of litter and duff on burn plots, which was estimated from post-fire measurements of discoloration on stationary rocks, averaged ~ 2.2 cm, ranging from 0 to 4 cm. Grass and oak also provided limited pre-fire cover (living plants and litter) estimated at 10% and 5% respectively based on post-fire distribution of root crowns and charred oak stems (burned up to 2 cm diameter). As an indicator of fire severity, the closest pine trees, averaging 12 m height, were fully consumed, i.e., all needles and small branches consumed, and dead fuels up to 20 cm in diameter lying on the ground were fully consumed.

Water drop tests performed on the post-fire ground surface in the study area indicated that some of the soil had limited water repellenency, which was heterogeneous with respect to both surface extent and depth. Water drop tests performed on burned plots just prior to rainfall simulation indicated no water repellency at the ashy surface. However, moderate water repellency was observed at the 1-2 cm depth interval during all four tests performed at random locations on the burned plots one year after the fire.

By comparison, on unburned soils, only one of four tests indicated any degree of water repellency (weak repellency at the 1-2 cm depth interval). Rather than rely completely on water drop test, which can be subjective and difficult to quantify, we sought to observe water repellency effects in the hydrologic response.

Runoff

After approximately 120 mm of applied rain, the two burned plots yielded 71 and 35 mm runoff, and the unburned plots yielded 26 and 27 mm runoff (Fig. 2.1 shows runoff ratios). Runoff volume was positively correlated to percent bare soil ($r = 0.76$) over twelve runoff events, and the highest runoff and the largest percentage of bare soils were both observed in plot 4 (burned). In contrast, runoff volumes were poorly correlated with surface roughness ($r = -0.12$). The time periods between start of rainfall and runoff initiation averaged 1.9 (±0.7) min for burned plots and 4.4 (±0.4) min for unburned plots. Times to runoff initiation were negatively correlated to percent bare soil ($r = -0.67$), with the shortest times from plot 4 (burned).

Hydrographs from burned and unburned plots indicated marked differences in their times to runoff initiation, and in the slopes of their rising limbs (Fig. 2.2). The regression coefficients of the rising limbs of the burned plots were consistently much steeper than those of the unburned plots (average of 5.3 mm min^{-1} (± 2.2) to 1.8 mm min^{-1} (± 1.2) respectively). Slopes of the rising limb portion of hydrographs are affected by both infiltration and surface features such as depression storage and ground cover that impedes flow (Frasier et al., 1998). If less infiltration and less surface impedence occurs, the result is steeper slopes of the rising limbs.

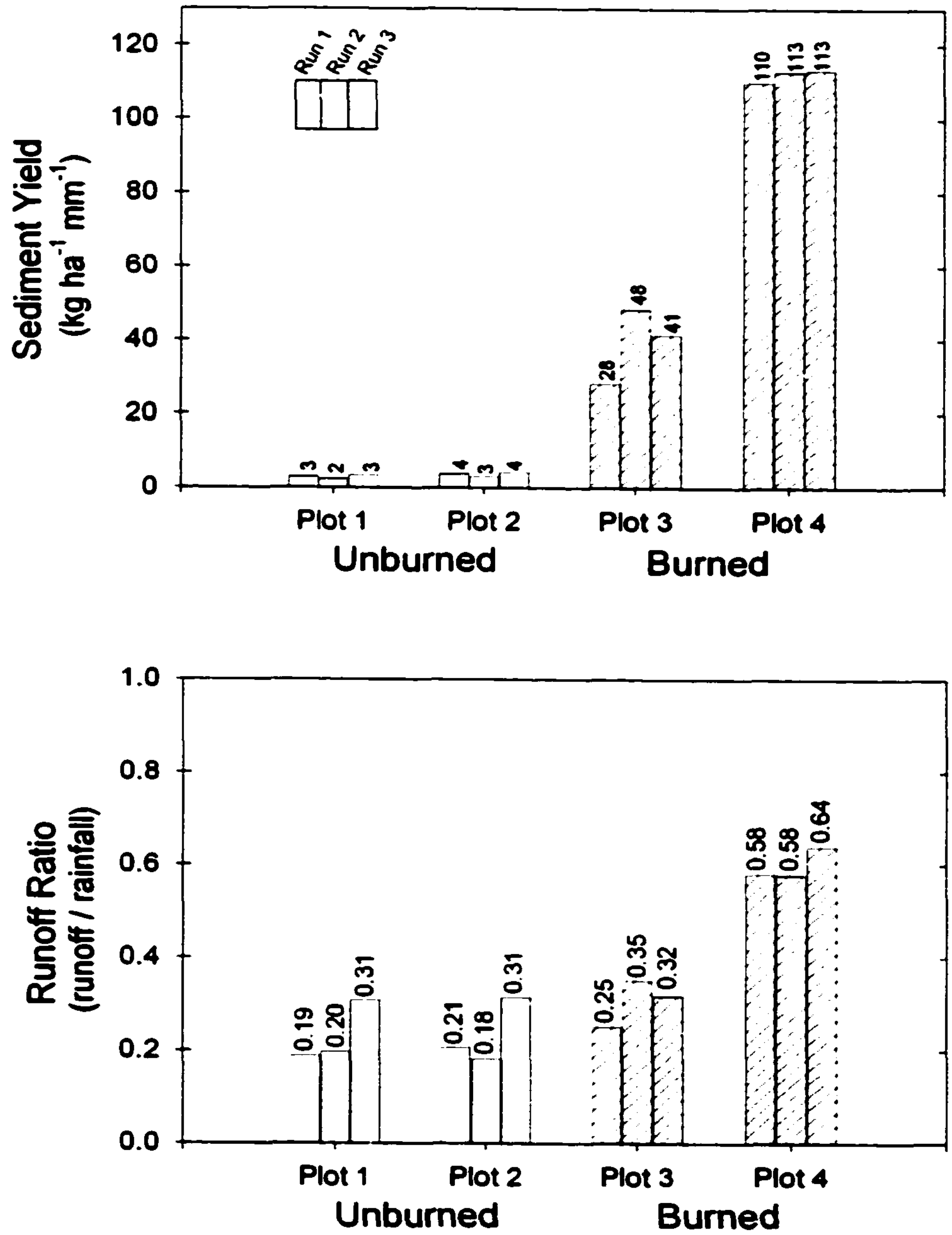

Fig. 2.1. Sediment yields and runoff ratios for burned and unburned plots (three runs per plot: Dry, Wet, and Very wet)

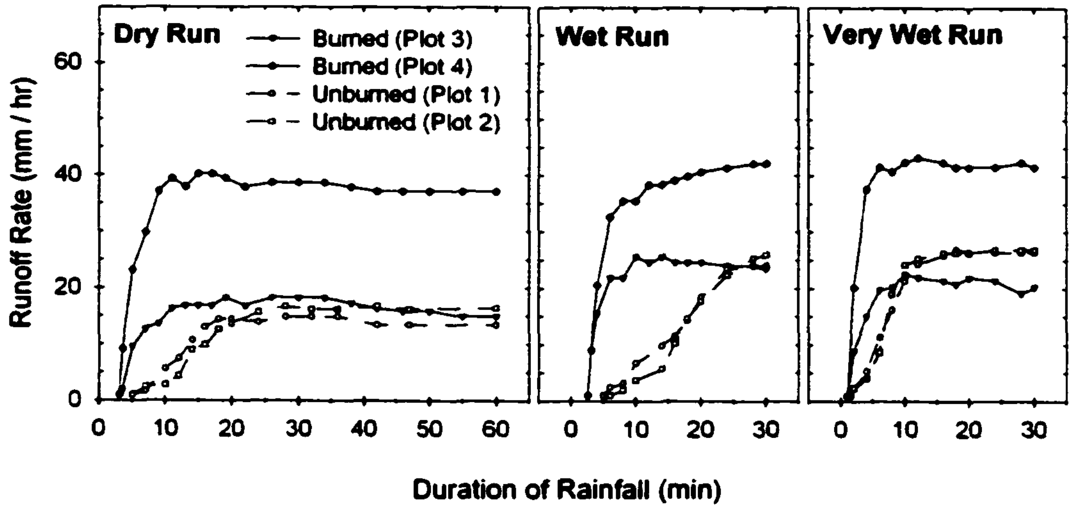

Fig. 2.2. Runoff hydrographs on burned and unburned plots

With respect to effects of water repellency, the shapes of the burned-plot hydrographs did not appear to confirm increases in runoff due to water repellency. In previous studies, water repellency was associated with a sharp initial spike in runoff rate with decreasing runoff rates thereafter (Shahlaee et al., 1991; Hester et al., 1997; Robichaud, 2000). These decreasing runoff rates were attributed to increases in infiltration as the soil became saturated, which has the effect of decreasing the degree of water repellency (DeBano, 1981). At Los Alamos, burned plots had relatively quick peaks in their runoff rates, but only slight reductions in runoff rates thereafter at slopes indistinguishable from those from the unburned plots ($P < 0.05$) (Fig. 2.2). One hydrograph associated with a burned plot even showed significant increase in runoff after initial saturation (Wet run, Fig. 2.2). This lack of characteristic behavior associated with hydrographs from water repellent soils implies limited effects of water repellency on runoff from burned plots at Los Alamos.

Sediment yields

Large increases in sediment yields were measured from burned plots compared to unburned plots (Fig. 2.1). Average sediment yields were 76 kg $ha^{-1}mm^{-1}$ (mm of rainfall) from burned plots compared to 3 kg $ha^{-1}mm^{-1}$ from unburned plots. The maximum sediment yield observed for all runs on burned plots was 113 kg $ha^{-1}mm^{-1}$.

Increases in sediment yields between burned and unburned plots (about a factor of 25) were disproportionately large compared to increases in runoff amounts (about a factor of 2). In addition, variation in sediment yield was observed between burned plots; with plot 4, having the least amount of ground cover (20% ground cover), yielding nearly 3 times more sediment than plot 3 (31% ground cover). Sediment yields were positively correlated with percent bare soil ($r = 0.84$); however this relationship is not expected to be well represented by simple correlation, as discussed below.

Post-fire rainfall simulations from different ecosystems

Rainfall simulation studies that focus on the effects of fire have been conducted in various grassland and shrubland systems, and to a lesser extent, in forest systems (Table 2.2). Of these studies, our results in ponderosa pine forest following an intense fire stand out as having the largest measured sediment yields, which are in most cases greater than other reported values by more than an order of magnitude (Table 2.2). A previous study in eucalyptus forest (Wilson, 1999) also demonstrated high sediment yields; that study was also conducted following severe burning by a wildfire that caused extensive removal of ground cover. In Table 2.2, larger sediment yields were associated with higher fire intensities, whereas smaller sediment yields were associated with studies of areas subjected to low-intensity fire, often which was prescribed burning

Table 2.2. Sediment yields from rainfall simulation studies on burned and unburned plots (grouped by plot size).

Location	*Dominant species*	*Soil type*	*Plot size*	*Rainfall intensity / duration*	*Burned condition*[†]	*Burned sed. yield / (% bare soil)*	*Unburned sed. yield / (% bare soil)*	*Slope*	*Time lapse after fire*	*Reference*
			meters	*mm/hr (min)*		*kg ha^{-1}mm^{-1} (%)*	*kg ha^{-1}mm^{-1} (%)*	*(%)*		
NE California	sagebrush-juniper	gravelly sandy loam	3 x 10.7	65 (30 – 60 min)	intense, controlled	**0.0 – 14.0** (40 – 45)	0.0 - 3.7 (15 – 17)		within days	Simanton et al. 1986
					intense, controlled	**9.5 – 22.0** (30 – 40)			7 months	
					low-intensity	**3.0 – 7.3** (19 – 20)			1 year	
SE Arizona (Santa Rita)	introduced grass	gravelly loam	3 x 10.7	55 (45 min) 110 (15 min)	low-intensity	**0.4 – 0.6**	0.3 – 0.4	5 – 6	same day	Emmerich & Cox 1992
SE Arizona (Empire)	native grass	gravelly sandy loam	3 x 10.7	55 (45 min) 110 (15 min)	low-intensity	**1.6 – 4.2**	1.3 – 2.4	5 – 7	same day	
SE New Mexico	Chihuahuan Desert grass	sand – loamy sand	3 x 10.7	60 (30 – 60 min)	low-intensity	**0.0 – 5.3** (41 – 48)	0.0 – 2.3 (17 – 23)	6 – 7	1 day	Johansen et al. 2001a
Central Colorado	shortgrass steppe	clay – clay loam	3 x 10.7	60 (30 – 60 min)	low-intensity	**3.5 – 7.2** (28 – 40)	1.5 – 2.3 (21 – 35)	9 – 10	1 day	Johansen et al. 2001a
NW N. Carolina	hardwood, pine	loam	3 x 7.5	100 (30 min)	low severity	**0.2 - 0.4** (7)		30		Robichaud, & Waldrup 1994
NW N. Carolina	hardwood, pine	loam	3 x 7.5	100 (30 min)	high severity	**0.2 – 27.7** (63)		30		
N. New Mexico	ponderosa pine	loam	3 x 10.7	60 (30 – 60 min)	severe, wildfire	**28.2 – 113.3** (69 – 80)	2.3 – 4.2 (38 – 58)	4 – 8	80 days	Johansen, *this study*
* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *
Tasmania Australia	eucalyptus	loamy sands	15 x 20	35 – 162 (10 – 49 min)	intense wildfire	**0.0 – 80.1** (>85)	(0.1 – 14.1) (<50)[*]	26	6 months	Wilson 1999

Georgia	hardwood, pine	sandy clay loam	1 x 5	78 – 102 (30 min)	slash burn	**0.95 – 2.6**	--	10 – 30	0 –12 months	Shahlaee et al. 1991
Leon Prov. NW Spain	shrub	loamy sand	1 m^2	180 (5 min)	low-severity	**0.6 – 13.3 (30 – 99)**	2.3	10	days - 1½ yr.	Marcos et al. 2000
N. Idaho	mixed conifer	silt loam	1 m^2	50 (30 min)	prescribed burn	**2.5 – 11.0**		13 - 27	within days	Robichaud et al. 1994
E. Nevada	pinyon-juniper	loamy, mixed	0.83 m^2	84 (60 min)	prescribed coppice	**4.0 – 9.7 (19 – 80)**	1.6 – 3.3 (1-17)	5 – 8	1-2 months	Roundy et al. 1978
					prescribed coppice	**3.1 – 12.8 (6 – 91)**			1 year	
	pinyon-juniper				prescribed interspace	**7.9 – 26.0**	10.1 – 21.6		1-2 months	
					Prescribed interspace	**11.9 – 32.0**			1 year	
Texas	juniper	silty clay	0.5 m^2	203 (50 min)	prescribed low-sev.	**9.5 (100)**	0.2 (0)	4	5 months	Hester et al. 1997
	oak					**22.2 (100)**	.01 (0)			
	bunchgrass					**22.0 (100)**	1.5 (32)			
	shortgrass					**28.4 (100)**	6.4 (57)			
Texas	whitebrush	sandy loam	0.4 m^2	203 (30 min)	prescribed	**6.0 (16)**	7.1 (19)		10 months	Knight et al. 1983
Texas	mesquite	clay loam		203 (30 min)	prescribed	**12.7 (12)**	19.2 (11)		10 months	

[†] Terms describing burn conditions vary between reported studies.

(Roundy et al., 1978; Knight et al., 1983; Simanton et al., 1990). Further, in one natural rainfall study after controlled, low-intensity fire was applied, a higher intensity wildfire occurred by chance, and allowed for comparison of the relative effects of low severity and higher severity burning (Soto and Diaz-Fierros, 1998). Results showed sediment yields 8.5 times greater on unplanned wildfire plots compared to unburned control plots, whereas sediment yield increased only slightly on low-intensity, prescribed burn plots relative to control plots.

We plotted results from a subset of rainfall studies with similar precipitation intensities (most were approximately 60 mm per hour) and plot scales (most were approximately 3 by 10 meters), after normalizing on a per mm rain basis (Fig. 2.3). These results show sediment yields from comparable studies across a wide range of ecosystem types (grasslands, shrublands, and forests) and fire intensities. Our results from study plots in the ponderosa pine forest show sediment yields an order of magnitude greater than most others from burned plots in other ecosystems (Fig. 2.3). Further, a curvilinear relationship is seen between bare soil and sediment yield, with little change in sediment yield as percent bare soil varies between 0% and ~60-70%, and sharp increases in sediment yields when the amount of bare soil is greater than 60 70%.

2.5 Discussion

High sediment yields following severe fire in ponderosa pine forest

Study results document large increases in sediment yields following severe fire in ponderosa pine forest. The observed sediment yields were well correlated with ground cover, which was greatly reduced by the wildfire. The surface of the burned

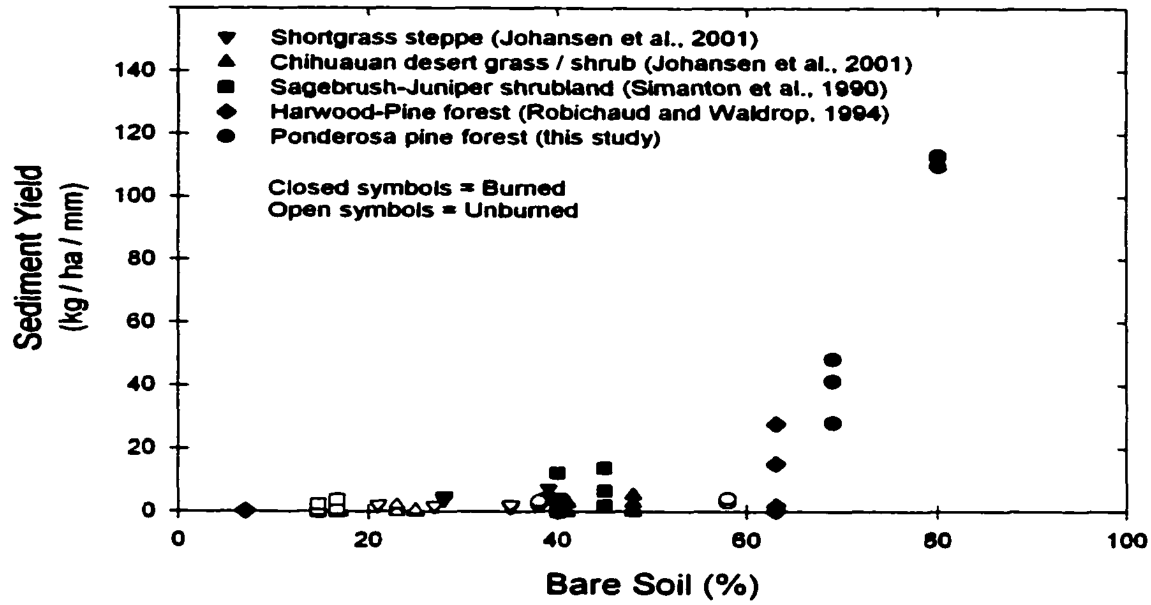

Fig 2.3 Sediment yields from rainfall simulations on comparably sized burned and unburned plots over a range of vegetation types

plots had 26% ground cover, most of which consisted of moveable ash, with only 6% considered persistent cover. The burned plots, with lower amounts of ground cover than the unburned plots, had larger amounts of runoff, which in turn provided greater erosive forces and greater potential for sediment transport by overland flow. In addition, much more soil was exposed to raindrop splash and shear erosive forces (up to 42% more on burned plots than unburned plots), further contributing to the large sediment yields observed.

The removal of ground cover by fire exposed previously protected soil that was highly susceptible to erosion. The soil at the study site was apparently covered for at least 60 years by a duff layer, which would have protected the soil surface from the effects of compaction by rainfall and of armoring from erosion. Other studies have shown that soil compaction and pavement cover, (i.e., armoring of the soil surface that can increase over time) provide resistance to erosive forces independent of ground cover (Simanton and Emmerich, 1994; Hakonson, 1999). For example, rainfall simulation studies in southern Arizona showed that sediment yield increased greatly when gravel and rock cover was removed (Simanton et al., 1986). This increase appeared much larger than would be explained by reduction in ground cover alone, supporting the concept that sudden exposure of uncompacted, unarmored soil increases sediment production, independent of cover amount. This is consistent with practices applied in erosion equations and models, such as the Revised Universal Soil Loss Equation, where a greater soil erodibility factor is used to model greater susceptibility to soil erosion (Renard, 1986).

Ground cover effects appeared to be more important in explaining hydrologic response than either surface roughness or slope. Burned and unburned plots in the ponderosa pine forest that we studied had similar surface roughness (Table 2.1) even though sediment yields were much greater on burned plots. In addition, surface roughness was poorly correlated to runoff volumes. Similarly, differences in plot slope also were not thought to be a major determinant of runoff volume, as slope and runoff volume were poorly correlated. Previous observations indicate that the effect of slope is likely to be small over the range of slopes we studied (Wilcox et al., 1988). Of the two burned plots, plot 3 had the greater slope but generated only about half the runoff volume of plot 4.

Water repellency also may have affected runoff, but we believe that this effect was not large for our study. Water repellency was evident near the site and may have been present at depth on burned plots based on surface burn characteristics. If present, water repellency would contribute to the observed high sediment yields by decreasing infiltration rates and thus contributing to runoff. In support of this, the rising limbs of the burned plot hydrographs were very steep compared to unburned plots, indicating less initial infiltration. However, hydrographs from the burned plot did not match the characteristic shape of hydrographs from water repellent soils (Robichaud, 2000). In addition, water repellent soil, when saturated, conducts water almost as rapidly as wettable soil (DeBano, 1981; Shakesby et al., 2000). Thus, because burned plot soils became quickly saturated during the rainfall simulations, the effects of water repellency, if any, were likely small.

Rainfall simulation results from different ecosystems of varying burn severities

Our results from ponderosa pine forest, in conjunction with those from grasslands, shrublands, and other forest ecosystems, highlight two general trends related to post-fire hydrologic response. First, the data indicate that post-fire sediment yields increase non-linearly as percent bare soil increases. Specifically, sediment yields increase little, if at all, when percent bare soil varies from 0% up to about 60-70%. This observation is supported by similar rainfall simulation data gathered from undisturbed plots in southern Arizona for which the relationship between cover and sediment yield had a slope near zero (vegetation cover varied between 17% and 66%; Simanton et al., 1986). Our data from ponderosa pine forest are consistent with this observation, but also indicate that when percent bare soil exceeds a threshold of ~60-70%, a sharp increase in sediment yield can occur. This threshold range is similar to that observed at the catchment- and watershed-scale. For example, Campbell et al. (1978) reported little sediment yield from unburned and moderately burned ponderosa pine forest watersheds having 8% and 61% bare soil, respectively, but high sediment yield (> 3800 kg ha^{-1} yr^{-1}) from a severely burned watershed with 77% bare soil. Our hillslope-scale results are consistent with these catchment- and watershed-scale observations of a threshold effect, and provide further support for the occurrence of such a threshold through the controlled conditions provided by rainfall simulation. Thresholds in runoff and sediment yield are likely related to the proportion and connectivity of small patches of bare cover, which generate most of the runoff (Dietrich et al., 1993; Davenport et al., 1998; Reid et al., 1999). More specifically, a threshold can be crossed when there is shift from low to high connectivity among patches (Davenport et al., 1998). Such a shift is likely to occur as the amount of bare area approaches the threshold range of 60-70% that we observed. The threshold range is similar to that suggested by mathematical percolation theory (Stauffer, 1985). If ground cover is viewed as a grid of cells, some of which generate runoff and some which do not, the probability of cells of a given type (e.g., bare cells that generate runoff) becoming highly connected at the hillslope scale exhibits a non-linear, threshold-like response when the proportion of bare cells is near 60% (Stauffer, 1985; Davenport et al., 1998).

The threshold-type response of erosion to fire severity appears to be a function primarily of reduction in ground cover and secondarily due to changes in soil properties (DeBano et al., 1998; Marcos et al., 2000). We hypothesize a curvilinear relationship between sediment yield and ground cover (Fig. 2.4a, bottom curve). We further hypothesize that this relationship can be modified by fire in at least two ways: rapid exposure of previously protected soils such that erosion rates for a given level of ground cover are increased; and, creation of water repellency or breakdown of soil aggregates that can further amplify erosion when fire is sufficiently intense to alter surface soil properties (Fig. 2.4a, top curve). This contrast suggests that forests have a relatively greater vulnerability to increases in post-fire erosion (Fig. 2.4b). Forests may have much higher sediment yields than would be predicted by extrapolation of post-fire data from grasslands or shrublands. In grassland and shrubland ecosystems, residence times of fire and fire intensities are typically low, generally causing relatively small changes in ground cover and consequently causing only small increases in erosion (e.g., several studies in Table 2.2). Because fire intensities are typically low, water repellency and other soil changes appears to be uncommon in grassland and many shrubland ecosystems. In contrast, intense fire in a ponderosa pine forest can simultaneously reduce ground cover, expose susceptible soils, and potentially create water repellent conditions. Consequently, the hydrologic response in these systems can shift from very low sediment yield in pre-fire conditions to extremely high sediment yields after fire (Fig. 2.4b).

Additional research is needed to assess the persistence of increases in post-fire sediment yields. Studies in Arizona show that elevated erosion rates on burned desert shrub lands can persist for at least 5 years after fire (Simanton and Emmerich, 1994). The conceptual model displayed in figure 2.4a may, with more supporting data, be useful in describing reductions in sediment yields over time during recovery. Better estimates of how post-fire sediment yields change over time and the relative importance of these processes can be made when more understanding is achieved regarding rates of revegetation, soil compaction and armoring, and breakup of water repellent soils.

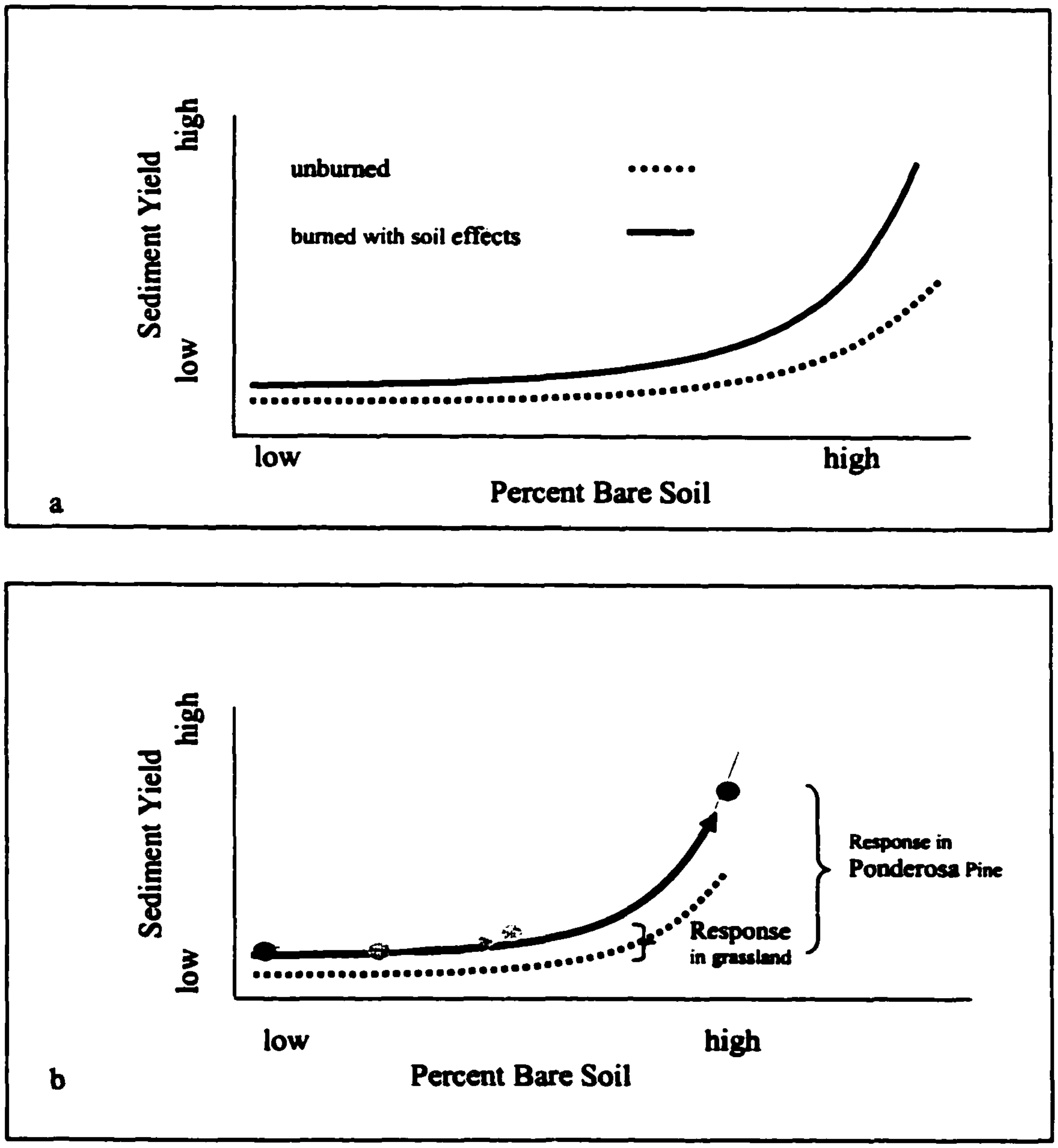

Fig. 2.4. Conceptual relationship of sediment yield with percentage bare soil (a), and conceptual response from grassland and ponderosa pine ecosystems (b).

In summary, our results document high sediment yields from rainfall simulation in severely burned ponderosa pine forest. These results contrast with results from grassland and shrubland ecosystems. Although these general relationships have been noted for watersheds and catchments, our study is among the first to quantify these responses systematically, particularly in a manner that controls for scale and rainfall effects, thereby allowing direct comparison of hydrologic response in grassland, shrubland, and forest ecosystems.

Chapter 3. Hydrologic Response and Radionuclide Transport Following Fire at Semiarid Sites

3.1 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 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, NM (WIPP), and the Rocky Flats Environmental Technology Site, CO (RFETS). Results showed that ^{137}Cs transport from burned plots was up to 22 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 to 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 demonstrates that rare events such as fire can greatly increase contaminant mobility.

3.2 Introduction

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 under-predict 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 U.S. at three nuclear weapons facilities. Wildfires occurred at the Los Alamos National Laboratory, NM (~3,000 ha burned), the Hanford Site, WA (~24,300 ha burned), and the Idaho National Engineering and Environmental

Laboratory, ID (~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 non-radiological 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 (Swetnam 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 (Parra et al., 1996). Fallout ^{137}Cs can be concentrated in ash after fire and the solubility of ^{137}Cs in ash decreased compared to 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 easy 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 to 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:

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

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.

3.3 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 (~91% sand) and RFETS having clayey soils (~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 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 to 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 3.1). The study area was likely 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 gramma (*Bouteloua eripoda*).

The RFETS study area is located about 1 km from the southeastern boundary of RFETS near Westminster, CO. The site has a semiarid climate with an average annual precipitation of 37 cm, of which ~40% occurs in the spring and ~30% in the summer (Tysdal, 2000). Average annual temperature is 9.7°C, with daily mean minimum and maximum temperatures of 8.8 to 31.2°C respectively. Site soils are clay to clay loam with relatively high cation exchange capacity (Table 3.1). Light grazing by horses had occurred recently at the study area. The average slope of the rainfall simulation plots is 9.1% (± 0.5%). Vegetation is shortgrass steppe with dominant species including blue gramma (*Boutelaoua gracilis*), western wheatgrass (*Agropyron smithii*), smooth brome (*Bromus inermis*), and intermediate wheatgrass (*Agropyron intermedium*).

Table 3.1. Surface and soil characteristics of study plots at Waste Isolation Pilot Plant (WIPP) and Rocky Flats Environmental Technology Site (RFETS).

	WIPP		RFETS	
Soil 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 (meq/100g)	8.8 (± 2.2)		27.5 (± 2.6)	
Random Roughness† (cm^2)	0.7		1.8	
Canopy Cover	***unburned***	***burned***	***unburned***	***burned***
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 (>20mm) (%)	0	0	1	1
Non-persistent litter (%)	33	35	3	18
Persistent litter (%)	1	0	33	16
Basal Vegetation (%)	35	11	32	26

† Expressed as standard deviation of height measurements

Experimental Design

Six plots of 3.0 m by 10.7 m (10 by 35 feet) 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 two days following burn treatments. A Swanson 16 m diameter, rotating-boom rainfall simulator was used to apply rainfall of ~60 mm hr^{-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., 1994; 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 one-hour rainfall application at about 60 mm hr^{-1} (labeled *Dry* run for its antecedent moisture condition) followed by a twenty-four hour recovery, then two one-half hour rainfall events at 60 mm hr^{-1} separated by a one-half hour 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 minute intervals at the flume exit during each simulation to provide for calculating sediment and radionuclide yields. Sediment samples were fractioned by wet sieving after shaking the samples for one hour to break up aggregates formed during drying. Concentrations of ^{137}Cs in sediment and soil were measured using a HPGE gamma ray spectrometer (EG&G ORTEC, Oak Ridge, TN) with counting times sufficiently long to reduce counting error to less than 12%.

3.4 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 3.1). 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 non-persistent 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 non-persistent 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 3.2). Average time of rainfall application before runoff began was 81 minutes during the Wet run on burned plots compared to 106 minutes during the

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 to 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 are 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, 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 to 18 of 18 at RFETS. Average runoff as a percent of total rainfall was 3.7% (± 6.4%) for all plots at WIPP, compared to 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 minutes occurring at WIPP and 5 min at RFETS. In addition, RFETS had higher antecedent moisture content (12.5% B 35.4%) compared to WIPP (0.0% - 13.3%), and runoff amounts generally increased as antecedent soil moisture increased during sequential runs (Table 3.2). 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 0.63 (± 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.

^{137}Cs 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 to 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 hr^{-1} for one hour 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 ~5.3 MBq ha^{-1}. At RFETS ^{137}Cs corresponding yields were 9051 Bq ha^{-1} while parent soils contained ~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, NV (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 to 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. 3.1). 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 μm 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 to 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

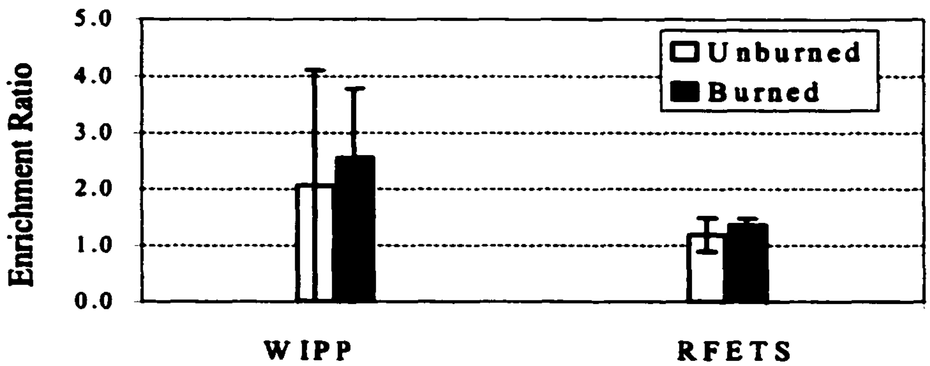

Fig. 3.1. Average enrichment ratios of ^{137}Cs in sediments from burned and unburned plots at the Waste Isolation Pilot Plant and the Rocky Flats Environmental Technology Site (error bars of ± 1 standard deviation).

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 fines (<45 μm) in successive runs as antecedent soil moisture increased (Fig. 3.2 and Table 3.2). This result is somewhat counterintuitive in that successive runs also had greater runoff flow rates which are expected to entrain more course 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

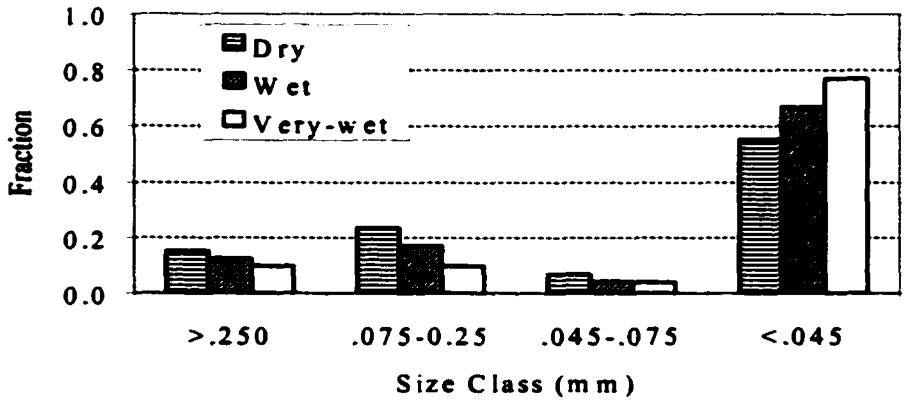

Figure 3.2. Changes in sediment particle size fractions in runoff from burned plots at RFETS as antecedent soil moisture levels increased. Average antecedent soil moisture levels were 12.5% for the Dry run, 28.8% for the Wet run, and 35.4% for the Very-wet run.

Table 3.2. Average runoff, sediment, and fallout ^{137}Cs yields from unburned and burned cover treatments at WIPP and RFETS.

		Antecedent moisture	Runoff (mm mm^{-1} rain)		Sediment yield (kg ha^{-1}mm^{-1})		^{137}Cs yield (Bq ha^{-1}mm^{-1})	
		(%)	*unburned*	*burned*	*unburned*	*burned*	*unburned*	*burned*
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)
	V.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)
	V.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.

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.

3.5 Discussion

Increased ^{137}Cs 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, likely 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-dominate sites was relatively low compared to 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 to 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 less rainfall amounts. For some ecosystems that are usually not prone to sediment runoff, the WIPP data implies potentially large increases in sediment-sorbed contaminant transport after fire where practically none would occur without fire.

Increased ^{137}Cs transport related to soil texture differences

In addition to the effects of ground cover removal by fire, our results show 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 (~44% clay) allowing for much less infiltration, and associated greater runoff amounts and longer runoff durations, than the sandy soils at WIPP (~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, ~4 times greater than paired unburned plots at RFETS, ~6 times greater than burned plots at WIPP, and ~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 to 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 is 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 to parent soils, with average enrichment ratios of 2.3 (± 1.7) at WIPP and 1.3 (± 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), that the enrichment of radionuclides in runoff occurs in the same ratio before and after burning. However, this study utilized 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 breakdown 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 3.3). 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

Table 3.3. Approximate enrichment ratios for nutrients and plutonium associated with various land uses and locations in the U.S. (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
<u>This Study</u> Chihuahua Desert Grass/shrub, USA (WIPP)			
Unburned	2.10.	0.6 – 3.5	fallout ^{137}Cs
Burned	2.61.	0.4 – 4.0	
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, OK.
‡ Small agricultural watersheds (2.6 - 2.9 ha) near Lebanon, OH.
§ Great Miami River (Drainage area = 1401 km^2) at Sidney, OH.
¶ Los Alamos Watersheds

in ephemeral stream channels at Los Alamos, NM, 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 3.3). 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 to 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 to 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 to 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.

Chapter 4. Pulsed Redistribution of Contaminants Following Fire: Cesium-137 Reconcentration and Erosion in Forest Runoff.

4.1 Abstract

Of the few natural processes that reconcentrate dispersed environmental contaminants, landscape fires stand out as having potential to rapidly reconcentrate contaminants, and do so on large scales. This study was conducted to quantify changes in concentration of a widely dispersed environmental contaminant –global fallout ^{137}Cs– in soils and runoff following a major forest fire. Our study at Los Alamos, NM, U.S., uniquely combined rainfall simulation methods performed shortly after a forest wildfire with a baseline of pre-fire data. Concentrations of ^{137}Cs in post-fire ash deposits were up to one order magnitude higher than in pre-fire soils. In subsequent runoff, ^{137}Cs concentrations averaged about 20 times greater for burned plots compared to unburned plots, and up to two orders of magnitude higher (maximum of 4.0 of Bq L^{-1}) in runoff from the encompassing watershed compared to pre-fire watershed runoff. However, these elevated concentrations in runoff returned to pre-fire levels after only 24 cm of rainfall. A portion of the concentration effect occurred during surface water erosion, resulting in enrichment of sediments by factors of 1.4 to 2.9 compared to parent soils. The greatest surface water transport of ^{137}Cs, up to 11.6 KBq ha^{-1} per mm rainfall, occurred after severe burning where 80% of the post-fire ground surface was denuded.

Our results provide evidence of order of magnitude concentration increases of a fallout radionuclide as a result of forest fire, and rapid transport of radionuclides following fire that can have importance for a wide range of geophysical, ecosystem, fire management, and risk-based studies.

4.2 Introduction

Certain processes, such as bioaccumulation, have been shown to reconcentrate dispersed environmental nutrients and contaminants. Such reconcentration processes stand out in contrast to the many natural processes that decrease concentration levels, such as dilution, dispersion, chemical change, and radioactive decay (Whicker and Shultz, 1982; Peles et al., 2000; Kendall and McDonnell, 1998). Of the processes that reconcentrate, those that occur during and after landscape fires are thought to stand out in their capability to concentrate and redistribute nutrients and contaminants on large scales and in rapid time frames (Debano et al., 1998). However, data are lacking in this area, particularly with respect to dispersed environmental contaminants. Examination of fire as an agent in concentrating and mobilizing environmental contaminants has relevance to a wide range of current interests including: ecosystem dynamics (Baird et al., 1999; Thomas et al., 1999); effects of increased potential for landscape fires from climate change (IPCC 2001, Piñol et al., 1998) and from increased forest fuel buildup (Sackett and Haase, 1999; Mast et al., 1999, Moore et al., 1999); relative to forest burning as a land use practice (Kauffman et al., 1993); and relative to mobility of dispersed contaminants, particularly radionuclides (Kashparov et al., 2000; Johansen et al., 2001b, Whicker et al., 2002).

Better information on fire-induced concentration of contaminants can be particularly useful in performing risk and dose assessments. Routine assessments, and the models used for these assessments, may under-predict doses and risks over long time frames when they do not include key fundamental mechanisms such as fire that induce episodes of accelerated contaminant movement (Whicker et al., 1999).

Ecosystem redistribution and concentration processes have been extensively quantified using tracers, such as fallout radionuclides that are widely dispersed in world ecosystems (Kendall and McDonnell, 1998). However, few such measurements have been made following landscape fires that can greatly amplify both the amounts and rates of redistribution. In a Canadian boreal forest, elevated concentrations of ^{137}Cs were found in surface soils at an area that had been burned years before, indicating reconcentration from burned vegetation to deposited ash, (Paliouris et al., 1995). Amiro et al. (1996) found that on a unit weight basis, the ash deposits after fire were enriched in elemental cesium from 4 to 25 times during field burns, and over two orders of magnitude in laboratory burns compared to the original vegetation. These studies suggest potentially large concentration factors of fallout radionuclides in ash and soils after forest wildfires.

Subsequent to deposition of concentrated radionuclides in ash on the ground surface, erosion and transport of concentrated radionuclides by surface water runoff are expected to occur. In the Paliouris et al. study (1995), the total ^{137}Cs inventory of the burned area (total soils and biomass) was less than that of an unburned control area, indicating redistribution away from burned areas. These authors attributed ^{137}Cs losses from the burned area to volatilization and fly-ash processes during fire, and runoff after fire. The importance of surface water runoff as a post-fire redistribution process was further suggested by ^{137}Cs inventories in both burned and unburned areas that showed highest inventories in the organic soil layer (up to 55 KBq ha^{-1}) at the ground surface where erosion processes occur, compared to the lesser above-ground inventory (up to 14 KBq ha^{-1}). However, these inventory measurements were made years after fire occurrence, not during transport processes, thus considerable uncertainty remains regarding rates of reconcentration and redistribution, and regarding the underlying factors controlling these process on post-fire landscapes.

Fate of ^{137}Cs during rainfall and runoff after fire in grassland and shrubland was studied by Johansen et al. (2001b), who found runoff from burned plots yielded up to 22 times greater amounts of ^{137}Cs compared to unburned plots (maximum yields of up to 0.28 KBq ha^{-1} per mm rainfall). Most of the increased ^{137}Cs transport was associated with the amplified sediment transport from burned surfaces. Little ash was observed to remain after the controlled grassland and shrubland fires.

Part of the increase in ^{137}Cs transport from burned plots observed by Johansen et al. (2001b), was attributed to enrichment of ^{137}Cs in runoff. Enrichment can occur during erosion and runoff through preferential entrainment of fine particles such as clay-sized particles which have greater affinity for cationic contaminants such as ^{137}Cs (Graf, 1971; Menzel, 1980; Lane and Hakonson, 1982; Watters et al., 1983; Lane et al., 1986; Hakonson and Lane, 1993; Weigand et al., 1998). This enrichment effect appears to occur on both unburned and burned landscapes; however, lacking are measurements of the enrichment effect following landscape fires where soil can reach high temperatures and thus potentially affect enrichment processes by changing soil characteristics.

These earlier studies suggest a series of processes during and after fire consisting of ashing of biomass, amplified surface water erosion, enrichment during surface water erosion, and accelerated transport in runoff, that result in concentration of fallout radionuclides and subsequent rapid redistribution away from burned areas. However, lacking are data that quantify these processes after a major landscape fire. Particularly lacking are concentration measurements of fallout radionuclides in surface water runoff and the underlying factors that govern radionuclide enrichment and transport after fire. Also unavailable is information relating specific fire effects, such as removal of vegetation, to radionuclide transport.

Our primary objective was to quantify changes in the concentrations of a fallout radionuclide –^{137}Cs in soils and runoff sediments after a major forest fire. Specifically, we used data from rainfall simulation plots and monitoring data from the watershed encompassing the plots to evaluate post-fire changes in (1) ^{137}Cs concentrations in soils, (2) ^{137}Cs concentrations in surface water runoff (both changes in concentration and duration of those changes), and (3) associated changes in the enrichment of ^{137}Cs in transported sediments compared to their parent soils. In addition, we compared ^{137}Cs transport from rainfall simulations for burned and unburned conditions in forest, shrubland, and grassland ecosystems. This comparison provides insight to the underlying processes governing post-fire movement of fallout ^{137}Cs, and similarly acting radionuclides, metals, and nutrients, and is expected to be relevant to a wide range of geomorphic, ecosystem dynamics, risk, and fire management applications.

4.3 Materials and Methods

Study Site

The study site is located within the Pajarito watershed near Los Alamos, NM, in the southwestern U.S. The watershed has a semi-arid, temperate mountain climate, with an average annual precipitation of about 50 cm, with the major portion occurring in July and August (Bowen, 1990). Study plots have loam soils, consisting of about 40% sand, 47% silt, and 13% clay (Johansen et al, 2001a). Site vegetation is dominated by mature ponderosa pine (*Pinus ponderosa*), gambel oak (*Quercus gambelii*), and a variety of forbs and grasses.

This study site may be unique because it is located within the Los Alamos National Laboratory where a baseline of environmental radiation data existed before May 2000, when the Cerro Grande wildfire burned the study site. Study plots were established 87 days after the fire at burned and unburned locations approximately 150 m apart. On the burned plots, all grasses and forbs were consumed, most Gambel oak was burned with some residual stalks remaining, and the ponderosa pine trees were burned to their full height of about 15 m. All needles and small limbs (approximately < 1 cm thick) were completely burned. In addition, the duff layer that had existed on the ground surface before the fire (ranging from 0 to 4 cm depth) was ashed including organic soil decomposition layers. Prior to the Cerro Grande fire, the study site is thought to have been unburned since the 1940's and thus available to accumulate fallout ^{137}Cs since the beginning of atmospheric testing of nuclear weapons.

The hydrology in ponderosa pine forest within the Pajarito watershed has been extensively characterized with respect to surface water runoff and subsurface interflow

(Wilcox et al., 1997; Newman et al., 1998; Wilcox and Breshears, 1998), soil water dynamics (Brandes and Wilcox, 2000), and infiltration (Newman et al., 1997). After the Cerro Grande fire, about 52% of the Pajarito watershed area was estimated to have burned to a high severity condition; about 44% was low severity (Interagency BAER, 2000). In total, the Cerro Grande fire burned about 17,400 ha in the Los Alamos area. This study adds to the growing body of work following the Cerro Grande fire regarding post-fire runoff, erosion, and sediment yields, including studies by Beeson et al., 2001; Cannon et al, 2001; Fresquez et al., 2001; Malmon et al., 2002; McLin et al., 2001; and Wilson et al., 2001.

Experimental Design

This study focuses on reconcentration and redistribution of ^{137}Cs, a globally-distributed fallout radionuclide found in plants, soils, and surface water runoff among other media. Post-fire ^{137}Cs measurements were made in soils and in surface water runoff that was generated using the rainfall simulation methods reported by Johansen et al. (2001a). Briefly, two rainfall simulation plots, each 3.03 m by 10.7 m (10 ft by 35 ft), were established in the wildfire-burned area and two additional control plots were established approximately 150 m away in an unburned area. Vegetation canopy cover, ground cover, and surface roughness were characterized on plot surfaces by point frame analysis (Levy and Madden, 1933). 48 soil samples were taken of the topmost five cm of soil and texture analysis was performed using sieving and pipette analysis at the Soils Laboratory, Colorado State University, Fort Collins, CO, U.S.

Rainfall simulation was used to generate runoff in lieu of natural storms to provide for control of rainfall amounts and intensities. A 16-m diameter, rotating-boom rainfall simulator applied rainfall at a rate of 60-mm hr^{-1}, in three separate runs that totaled about 120 mm of applied rain. The three runs consisted of a one-hour run (labeled *Dry* run for its antecedent moisture condition), followed by a twenty-four hour interval of no rain, then second and third rainfall events (*Wet* and *Very wet* runs) separated by a one-half hour period of no rain. A one-hour storm at 60-mm hr^{-1} represents approximately a 100-year recurrence interval at Los Alamos. 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). Rainfall simulations were initiated 80 days following the Cerro Grande fire. One liter unfiltered runoff samples (water and sediment) were collected every 2 - 4 min at the downslope end of each plot in a gutter system that channeled runoff through a calibrated flume where flow measurements were made using a bubble gage flow meter (ISCO, Lincoln, NE). The runoff samples were allowed to settle for more than 24 days, then a 5-ml sample was taken of the water fraction to test for dissolved and colloidal-associated ^{137}Cs. The sample was then dried and weighed, followed by wet sieving of sediment to determine grain size distributions.

^{137}Cs activity in study samples was measured using a high purity germanium gamma ray detector system (EG&G ORTEC, Oak Ridge, TN) at the Department of Radiological Health Sciences, Colorado State University, Fort Collins, CO, U.S. The detector uses a well configuration with a 76.4 mm diameter crystal having a measured efficiency of 95.8% relative to the sodium iodide crystal. Sample count times averaged 26.8 hours. Samples weighed 3.2 grams on average and counting error averaged 3.4 %.

The data collected at rainfall simulation plots for this study are compared to watershed data published by the Los Alamos National Laboratory (LANL, 2001) and the State of New Mexico, Environment Department. Specifically, data from a LANL soil sampling station located between the burned and unburned plot provided a pre-fire baseline of ^{137}Cs activity levels in site soils. A nearby LANL surface water runoff station (LANL station #E240) provided both pre-fire and post-fire runoff and radiological analysis data for the ~300 ha upper Pajarito watershed. ^{137}Cs concentrations in ash were reported by the State of New Mexico from samples taken within 200 m of the study plots, as well as in other Pajarito watershed locations. In addition, twelve background samples were collected of organic soils from ponderosa pine forest areas at: Gila National Forest, NM; Los Alamos, NM; La Veta Pass, CO; and near Fort Collins, CO. These samples were collected from the organic soil layer just above the mineral soil layer in forest areas where no evidence of fire was visible. The samples were collected directly in 5 cm^3 plastic vials for ^{137}Cs activity measurement.

4.4 RESULTS

^{137}Cs increases on the ground surface after fire

The ^{137}Cs concentrations in post-fire soils (0 – 5 cm) on burned study plots averaged about three times greater than in soils on unburned plots and pre-fire soils (Table 4.1); ^{137}Cs concentrations on burned plots averaged 0.044 (±0.002) Bq g^{-1}, compared to pre-fire soils that averaged 0.014 (±0.003) Bq g^{-1} dry weight (Fig. 4.1).

Table 4.1. ^{137}Cs concentrations in soil, ash, and vegetation

	Location	^{137}Cs Concentration		Reference
		Soil, 0-5 cm (Bq g^{-1})		
July, 2000 (post-fire)	Burned plots	0.044	(± 0.002)†	this study (18 samples)
July, 2000 (post-fire)	Unburned plots	0.015	(± 0.001)	this study (18 samples)
1995 – 1999 (pre-fire)	Two Mile Mesa (between study plots)	0.014	(± 0.003)	(LANL, 2001) (5 samples)
		Ash and charred wood, ground surface (Bq g^{-1})		
May-June, 2000, (post-fire)	Pajarito watershed	0.192	(± 0.176)	NMED (11 samples)
		Pre-fire vegetation near study plots (Bq g^{-1})(laboratory-ashed)		
July, 1998 (pre-fire)	Understory plants	0.010	(±0.015)	Gonzales et al., 2000
	Overstory plants	0.014	(±0.020)	
		Organic soil in ponderosa forest (humus) (Bq g^{-1})		
July, 2000	Unburned, at Los Alamos	0.10	(± 0.02)	this study (3 samples)
July 2000 – April 2002	Three unburned locations in New Mexico and Colorado	0.09	(± 0.07)	this study (9 samples)

† Standard deviation on multiple measurements

This post-fire increase was attributed to higher ^{137}Cs concentrations within the ash-dominated uppermost soil layer. Concentrations within just the ash alone averaged 13 times higher, and were up to 40 times higher, than pre-fire soils (ash concentrations averaged 0.192 Bq g^{-1} with a range of 0.007 to 0.570 Bq g^{-1}).

The elevated ^{137}Cs in ash is thought to have been derived from burning of both canopy and ground surface material. However, the greatest portion is expected to have

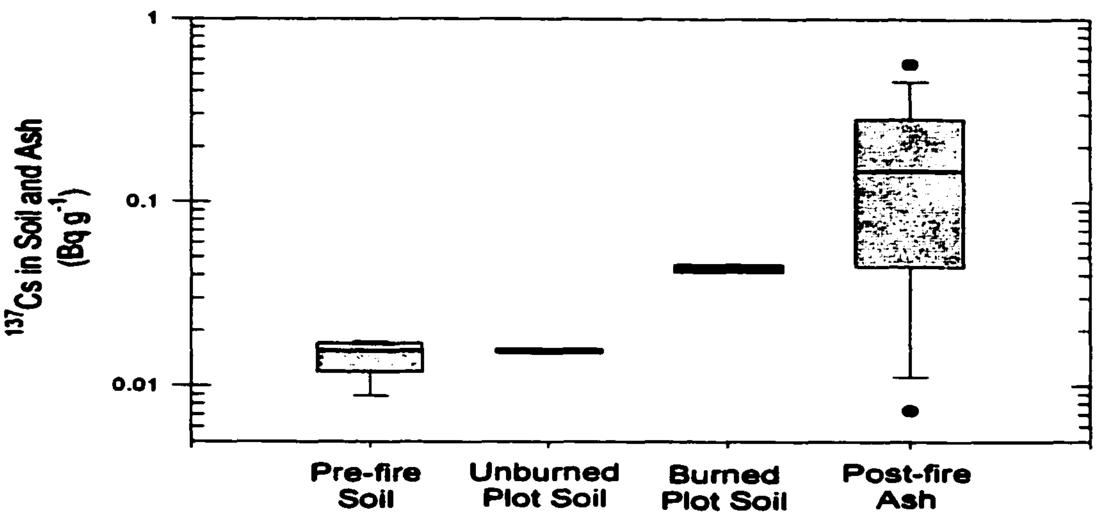

Fig. 4.1. ^{137}Cs concentrations at the ground surface before and after the Cerro Grande wildfire. Data modes are shown within the 25 – 75 percentile boxes. Whisker bars show standard deviation, and outlier data are shown where present.

come from ashing of ground surface litter and organic soil layers, dominated at this site by decaying ponderosa pine needles. ^{137}Cs concentrations in the organic decomposition layer from the closest unburned, undisturbed site at Los Alamos averaged 0.10 (± 0.02) Bq g^{-1}. Similarly, additional organic soil samples from three unburned ponderosa pine sites in New Mexico and Colorado averaged 0.09 (± 0.07) Bq g^{-1}. By comparison, samples of overstory plants from the study site prior to the fire had ^{137}Cs concentrations averaging only 0.01 (± 0.02) Bq g^{-1} (laboratory-ashed) indicating a greater pre-burn ^{137}Cs inventory in the ground surface biomass compared to the above ground biomass (Gonzales et al., 2000).

Table 4.2a. Average ^{137}Cs concentrations in runoff from rainfall simulation.

July 2000	^{137}Cs in unfiltered runoff ($Bq\ L^{-1}$)		^{137}Cs in sediment ($Bq\ g^{-1}$)		^{137}Cs yield ($KBq\ ha^{-1}\ mm^{-1}$)	
	Unburned	Burned	Unburned	Burned	Unburned	Burned
Dry run	0.08 (±0.04)†	1.48 (±0.60)	0.032 (±0.012)	0.087 (±0.014)	0.55 (±0.09)	7.03 (±6.50)
Wet run	0.05 (±0.01)	1.20 (±0.36)	0.034 (±0.005)	0.064 (±0.006)	0.41 (±0.01)	5.65 (±3.71)
Very wet run	0.06 (±0.03)	1.03 (±0.19)	0.039 (±0.002)	0.060 (±0.004)	0.70 (±0.02)	5.20 (±3.45)

† Standard deviation on multiple measurements

Table 4.2b. ^{137}Cs concentrations in runoff from the Pajarito watershed.

	^{137}Cs in unfiltered runoff ($Bq\ L^{-1}$)	Uncertainty	Reference
1995-1998, Prefire surface water and runoff samples	0.009	(± 0.07)†	LANL 2001 (4 samples)
June 28, 2000 (first runoff event)	4.03	-	LANL, 2001 (1 sample)
Sept. 9, 200 (second event)	1.17	(±0.22)‡	LANL, 2001 (1 sample)
Oct. 23, 2000 (third event)	0.07	(±0.10)‡	LANL 2001 (1 sample)
Feb.- May 2000	0.09	(±0.11)†	LANL, 2002 (6samples)

† Standard deviation on multiple measurements.
‡ Counting uncertainty.

^{137}Cs increases in surface water runoff after fire

Post-fire concentrations of ^{137}Cs in unfiltered runoff from burned plots subjected to rainfall simulation were elevated on average 20 times greater than unburned plots (burned plot average of 1.22 (±0.44) Bq L^{-1}, unburned of 0.06 (±0.03) Bq L^{-1}) (Table 4.2a). Post-fire concentrations of ^{137}Cs in runoff from the Pajarito watershed were elevated even higher, reaching 4.0 of Bq L^{-1}, an increase of more than two orders of magnitude over pre-fire concentrations (Table 4.2b).

Total yields of ^{137}Cs in runoff leaving plots during rainfall simulation (Bq transported per ha, per mm rainfall) were about one order of magnitude greater from burned plots (5.96 KBq mm^{-1} ha^{-1}), than yields from unburned plots (0.55 KBq ha^{-1} mm^{-1}). The total yields from burned plots ranged from 2.4 to 11.6 KBq ha^{-1} mm^{-1}.

These large increases in ^{137}Cs transport on burned plots were associated with increased sediment yields. ^{137}Cs concentrations in unfiltered water were strongly correlated with amount of suspended solids present ($r = 0.95$) (Fig. 4.2). Total suspended solid measurements after the fire reached as high as 27 g L^{-1} for burned plots and up to 35 g L^{-1} in the runoff from the burned watershed, compared to an average of 2.3 g L^{-1} for unburned plots. Consistent with this, samples of unfiltered runoff from burned plots had much more ^{137}Cs sorbed to particulates (up to 2.3 Bq in the particulates suspended in one liter of runoff water), than in the separated water fraction (average of 0.1 of Bq L^{-1}).

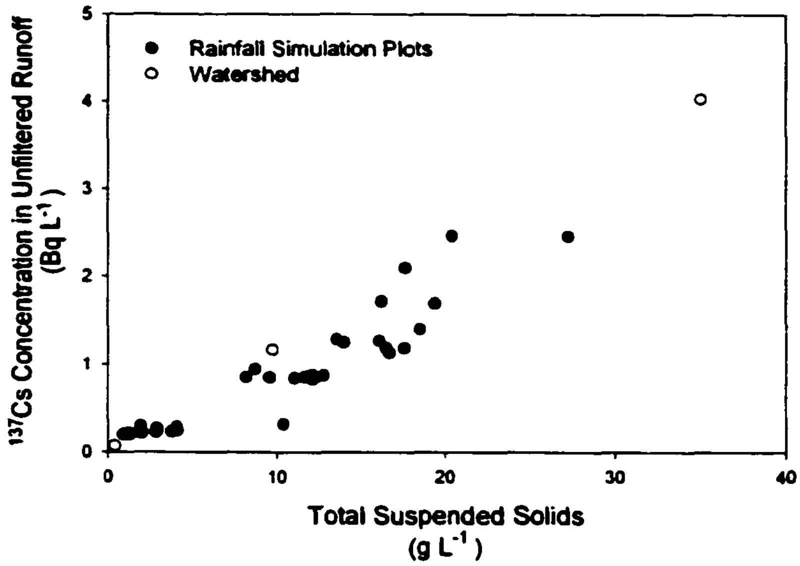

Fig. 4.2. Concentrations of ^{137}Cs and total suspended solids in runoff.

Duration of elevated ^{137}Cs in runoff

In runoff from study plots, the highest ^{137}Cs concentration was found in the initial runoff, 2.46 of Bq L^{-1}, with subsequent concentrations lowered by half at the end of the rainfall simulation (Fig. 4.3). This decrease by half of the initial concentration occurred during application of about 120 mm of simulated rainfall, or about one-fourth of the average annual rainfall for Los Alamos. Similarly, for runoff from the Pajarito watershed, the highest concentration observed (4.0 Bq L^{-1}) was in a sample taken from the first substantial runoff event following the fire (49 days after burning). A sample from the second runoff event had a lower concentration of 1.2 Bq L^{-1}. By the third watershed runoff event, 166 days following the fire, the measured concentration was reduced to 0.07 Bq L^{-1}, which is comparable to the pre-fire average of 0.03 Bq L^{-1}.

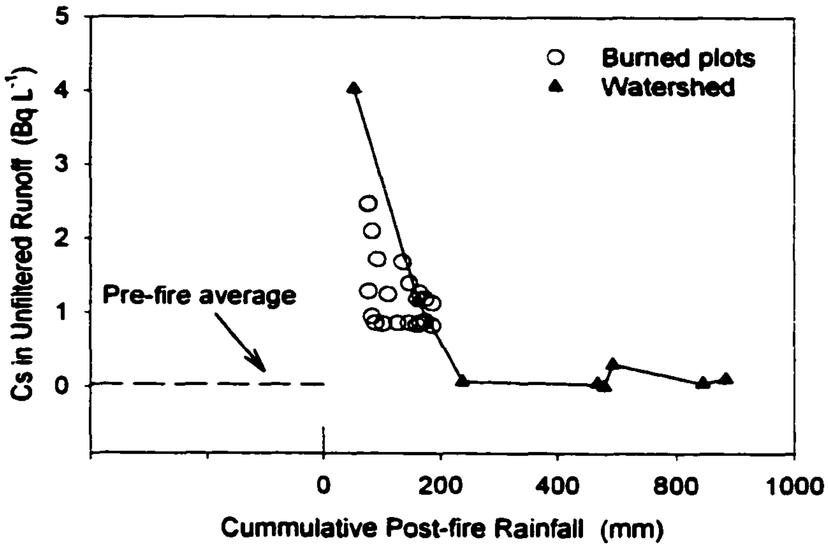

Fig. 4.3. Post-fire concentrations of ^{137}Cs in unfiltered runoff during cumulative rainfall

During this 166-day interval, about 240 mm of rain fell on the watershed, including three storm events large enough to generate runoff sufficient for sampling. This amplified transport of ^{137}Cs, and subsequent reduction, coincided with removal of ash from the post-fire ground surface. For the Pajarito watershed, it appeared that most ash was removed with about 24 cm of rainfall based on inspection of the burned areas and the return of runoff concentrations to near pre-fire levels.

Enrichment effects

Enrichment of ^{137}Cs concentrations in runoff sediments occurred during surface water erosion on burned and unburned study plots. Enrichment ratios comparing ^{137}Cs concentrations in runoff sediments to the concentrations in their parent soils ranged from 1.4 to 2.9. The average enrichment ratio for unburned plots was 2.3 (± 0.6) and for burned plots was 1.6 (± 0.4) (Table 4.3). The measured enrichment effect appears to be primarily associated with preferential entrainment of smaller sized particles during

Table 4.3. Enrichment ratios of ^{137}Cs in runoff sediments from rainfall simulation.

	Location	Enrichment ratio	Increase in ≤50 μm size fraction	Reference
July, 2000	Unburned plots	2.29 (± 0.60)†	18.5 %	this study (12 samples)
July, 2000	Burned plots	1.62 (±0.39)	7.8 %	this study (12 samples)

† Standard deviation on multiple measurements.

surface water erosion. These smaller sized particles (≤ 50 μm) made up greater weight percentages in runoff sediments than in parent soils (18.5 % greater for unburned plots and 7.8% greater for burned plots). The smaller sized particles also have greater affinity with the ^{137}Cs cation, resulting in higher per weight concentrations of ^{137}Cs in runoff sediments compared to parent soils.

Comparison of post-fire ^{137}Cs transport and enrichment from different ecosystems

Combining results from this study in forest, with results from previous companion studies in grassland and shrubland, provides a comparison of ^{137}Cs transport and enrichment processes. The main similarities between these rainfall simulation studies were rainfall amount and intensity, rainfall duration, and size and shape of study plots. The main differences were vegetation types (grassland to forest), type of burning, and soils textures. An intense wildfire burned the ponderosa pine forest site, while controlled fire was used at the grassland and shrubland sites. Some variation existed in slopes (from 4.5% to 9%) of the various study plots (Johansen et al., 2001b).

Combined results show amplified ^{137}Cs transport following burning in ponderosa pine forest where the intense wildfire consumed up to 80% of the ground cover (Fig. 4.4). ^{137}Cs transport was consistently lower from the unburned plots in all vegetation types,

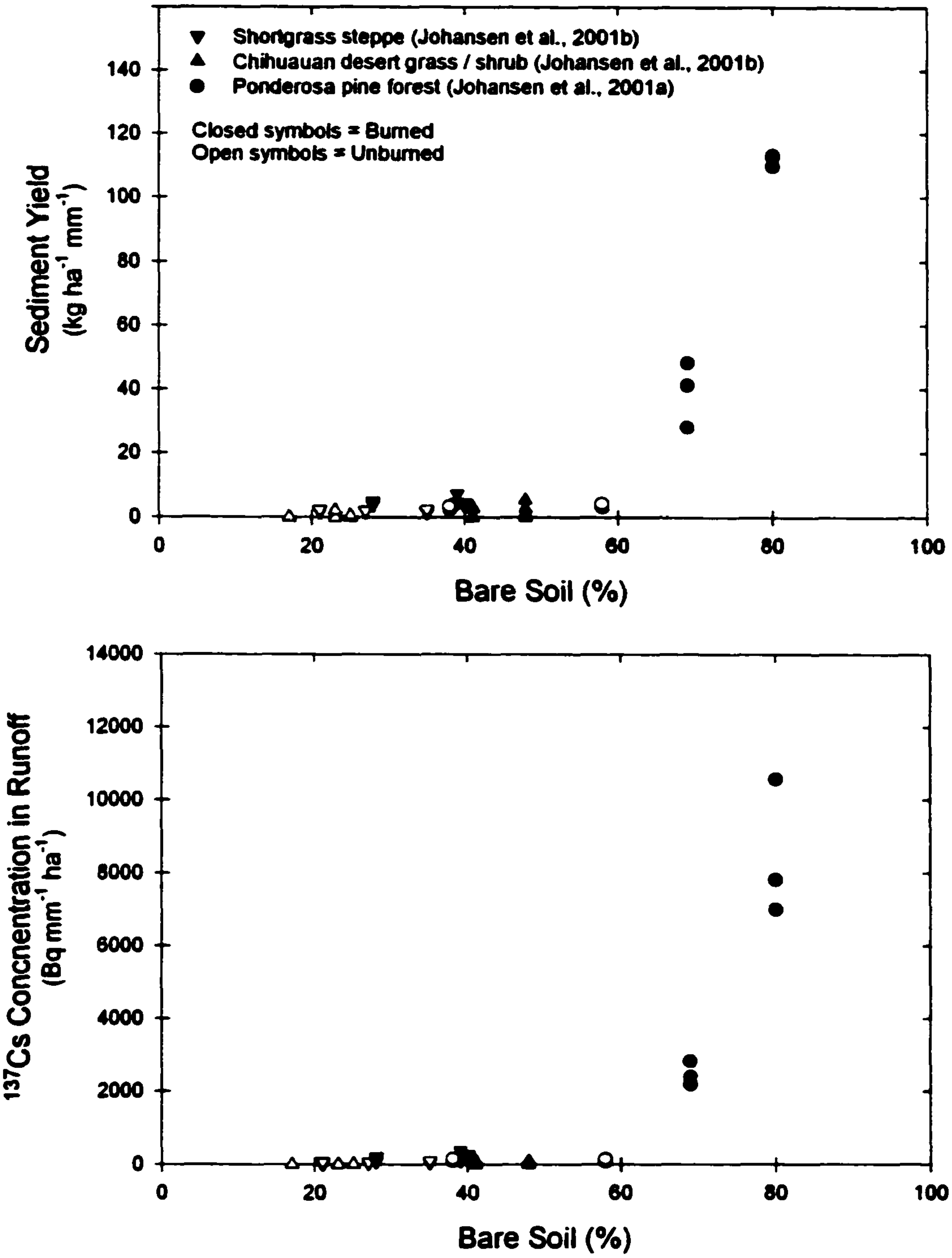

Fig. 4.4. Sediment yields and ^{137}Cs yields from rainfall simulations on burned and unburned plots over a range of vegetation types.

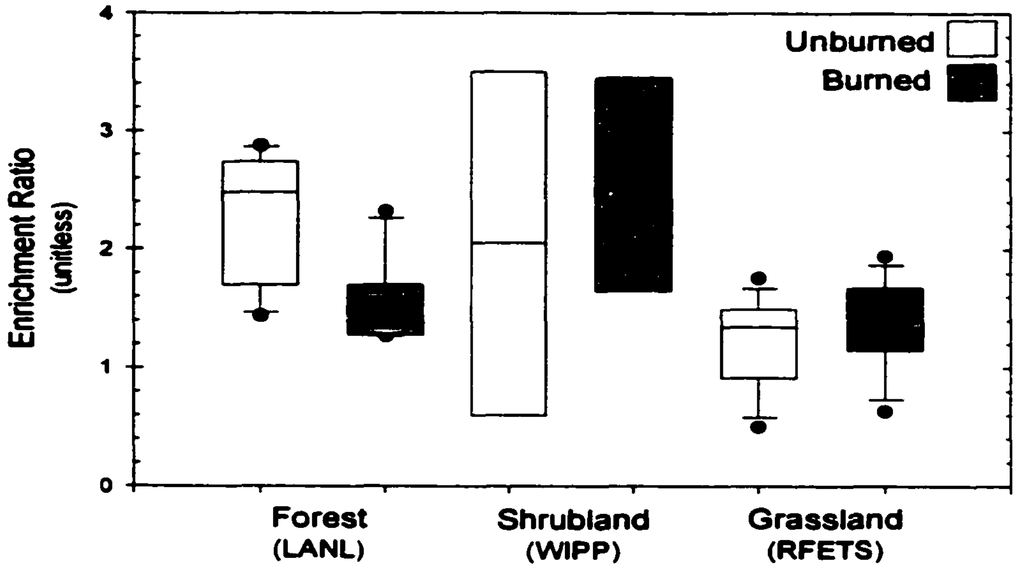

Fig. 4.5. Enrichment ratios in runoff sediments from rainfall simulation in a range of ecosystem types. Data modes are shown within the 25 – 75 percentile boxes. Whisker bars show standard deviation, and outlier data are shown where present.

and from the burned plots in grassland and shrubland where smaller percentages of vegetation was removal by fire. Further, ^{137}Cs transport relative to the percentage of bare soil on a plot surface is not linear, but appears to have little change as percent bare soil varies between 0% and about 60-70%, and sharp increases in ^{137}Cs transport when the amount of bare soil is greater than 60-70%. Correlation between ^{137}Cs yields and sediment yields was high across all sites ($r = 0.96$) and enrichment of runoff sediments occurred at all sites for both burned and unburned plots (Fig. 4.5). Average enrichment ratios ranged from 1.2 to 2.6 at all sites in grassland, shrubland, and forest. This enrichment of ^{137}Cs at all sites was primarily associated with preferential entrainment of fine particles that have a greater affinity for ^{137}Cs than coarser particles.

4.5 Discussion and Conclusions

A more complete description of the post-fire fate of fallout radionuclides emerges when our data are synthesized with results from previous studies. After fire, the ^{137}Cs from the burned biomass that is not lost to the atmosphere is concentrated in ash deposits on the ground surface (Paliouris et al., 1995; Amiro et al., 1996). In addition, burning of vegetation by fire exposes mineral soil having its own inventory of sorbed ^{137}Cs. The ^{137}Cs in ash and newly exposed soil is highly susceptible to resuspension to the air by wind (Whicker et al., 2001), and subject to erosion and transport by water processes (Johansen et al., 2001b) (Fig. 4.6). As a result, runoff from the burned areas contains

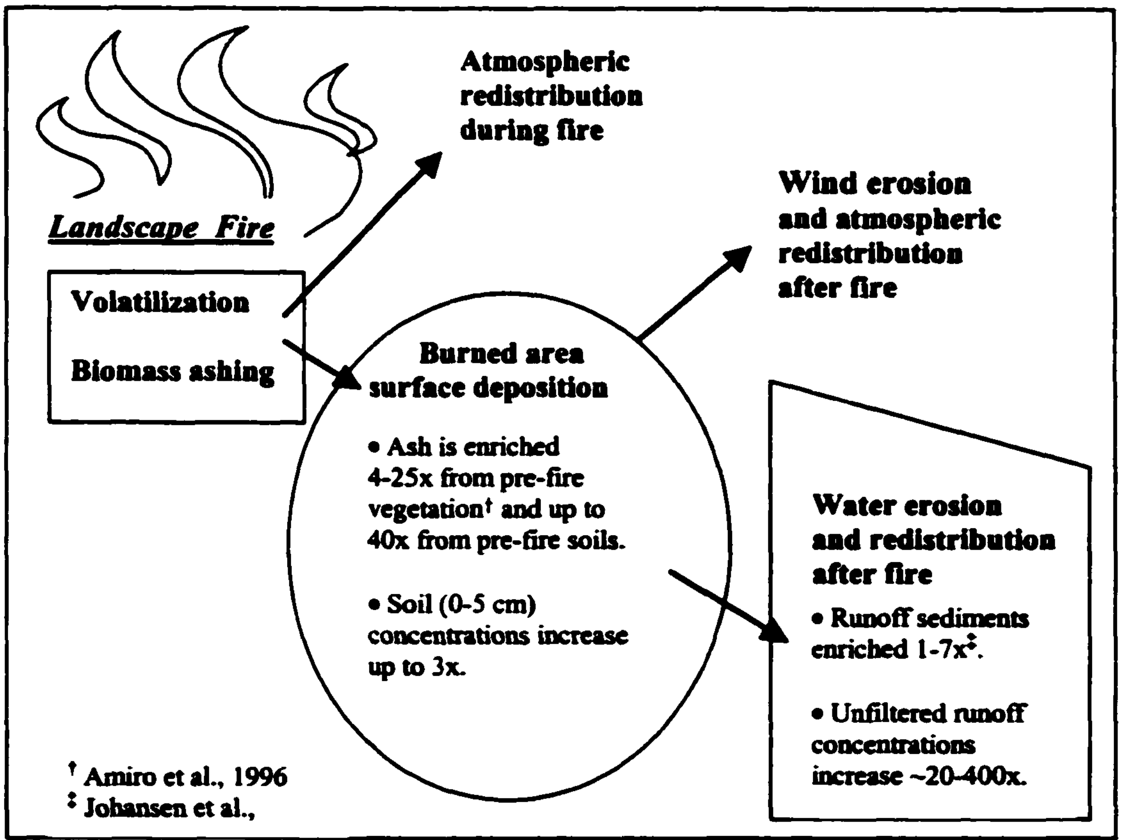

Fig. 4.6. ^{137}Cs Reconcentration processes and transport pathways following fire

elevated ^{137}Cs concentrations, associated primarily with sediments (maximums of 2.5 Bq L^{-1} from burned study plots and 4.0 Bq L^{-1} from the study watershed). Concentration increases on the 32.5 m^2 plots were an order of magnitude times greater than unburned plots, and increases in the 300 ha watershed scale were two orders of magnitude greater than pre-fire levels suggesting a greater concentration effect as landscape scale increases.

Because the radionuclide-enriched ash deposits that occur following fire are highly susceptible to erosion, the occurrence of elevated radionuclide levels following fire is expected to be relatively short lived. In the study watershed, most of the concentrated ^{137}Cs in ash was transported away from its original deposition area after about 24 cm of rainfall that occurred over the 166 days following the fire. Corresponding reductions of concentrations in runoff occurred during that same time period. When longer time-frames are considered, this rapid redistribution of ^{137}Cs will appear as a sharp pulse of elevated concentrations leaving the burned area.

The rate of ^{137}Cs loss caused by water erosion depends highly on the amount of ground cover removed by fire (Fig. 4.5). The relationship between post-fire ^{137}Cs transport rates and removal of groundcover by fire appears to have a threshold at about 60-70% with little transport occurring as percentages of bare soil vary between 0% and 60-70% and sharp increases for surfaces having greater percentages of bare soil. Total ^{137}Cs yields in runoff following intense fire in forest can be as high as 5 - 12 KBq ha^{-1} mm^{-1} compared to ^{137}Cs yields in runoff from unburned areas that averages about one order of magnitude lower.

Part of the elevated levels ^{137}Cs in runoff appears to be a result enrichment of ^{137}Cs in sediments during surface water erosion that further concentrates the ^{137}Cs.

Sediments leaving study plots had higher proportion of fine-grained soil particles than their parent plots, and corresponding higher amounts of ^{137}Cs. Enrichment by water erosion on plots in this study, and in companion studies, ranged from 1.2 – 2.6. This range is consistent with other enrichment ratio measurements of 0.4 – 7.1 made in studies that examined enrichment of sediments for a broad range of radionuclides and nutrients, on varying landscape scales (Johansen et al., 2001b), and from 0.4 – 4.9 specific to ^{137}Cs in small watersheds (Weigand et al., 1998). Our study specifically examined the question of whether or not soil heating during fire would alter enrichment ratios in subsequent runoff. In repeated tests on burned and unburned plots in grassland, shrubland, and forest, no significant difference in enrichment ratios was measured in sediments compared to their respective burned and unburned parent plots. Enrichment occurred on both types of plots, and ^{137}Cs yields were much higher on burned plots because of the associated greater sediment and ash yields, however, the ratio of enrichment between runoff sediments and parent soils was about the same for burned and unburned surfaces.

Considering the various transport mechanisms discussed above, the temperature at which fire burns is thought to be a key underlying factor in determining the fate of post-fire ^{137}Cs. Higher combustion temperatures lead to combustion of a greater portion of the organic soil layers that contain elevated concentrations of fallout radionuclides. ^{137}Cs concentrations in organic ponderosa pine forest soils measured in this study (about 0.1 Bq g^{-1}) are about an order of magnitude greater than typical soil levels in the southwestern U.S. (about 0.01 Bq g^{-1}) (Fresquez et al., 1996). It is these same forest areas that are susceptible to fire, and perhaps increasingly at risk of high-intensity fire from increased build up of forest fuels. In addition to ashing organic soils where most fallout radionuclides reside, hot fires increase ^{137}Cs loss to the atmosphere (Amiro et al., 1996), and lead to accelerated surface water transport brought on by the greater loss of ground cover. At lower burn severities, less of the ^{137}Cs inventory is ashed and less is lost to the atmosphere. In low intensity fires, ^{137}Cs in ash and soil is less susceptible to surface water erosion when enough ground cover remains to retard erosion (Fig. 4.5). Our results suggest approximately 30-40% ground cover is needed to prevent highly accelerated erosion on slopes of less than ten percent.

While this study focuses on concentration and transport of ^{137}Cs, our conclusions are expected to generally apply to other particle-reactive fallout radionuclides, metals, and nutrients. Hulse et al., (1998), showed that ^{137}Cs behavior in soils was well correlated to that of ^{241}Am, and $^{239,240}Pu$. Further, following the Cerro Grande fire at Los Alamos, NM, data gathered by the Los Alamos National Laboratory indicated order of magnitude concentration increases in ^{90}Sr, ^{238}Pu, and $^{239/240}Pu$, barium, manganese, and calcium (among others) in runoff samples collected in burned watersheds (Gallaher et al., 2002). Samples for these measurements were taken in watercourses upstream of the Los Alamos National Laboratory areas, away from laboratory waste areas.

In conclusion, this study documents order of magnitude increases of fallout ^{137}Cs in ash, soils, and surface water runoff following a major fire in a southwestern U.S. ponderosa pine forest. These increases are a result of a series of reconcentration and redistribution processes including concentration increases during ashing of biomass, amplified transport of ^{137}Cs by surface water runoff, and enrichment of ^{137}Cs during surface water transport due to preferential entrainment of small sediment particles. ^{137}Cs redistribution rates increase sharply when fire removes more than 60-70% groundcover.

Study results provide new data on contaminant reconcentration and redistribution following wildfire in forests, and systematic comparisons between forests, grasslands, and shrublands. This new information is expected to have importance for a wide range of interests including ecosystem dynamics studies, geophysical studies, fire management, and risk assessments where periodic fire events may dominate contaminant redistribution rates.

Chapter 5. Comprehensive Discussion and Conclusions

This discussion summarizes the comprehensive findings and discussion points of the entire study conducted on sites ranging from grassland, shrubland, and forest.

^{137}Cs increases on the ground surface after fire

On landscapes ranging from grassland to forest, ^{137}Cs concentrations in soils (0 – 5 cm) increase after landscape fire compared to pre-fire conditions. In grassland (RFETS) and shrubland (WIPP), burned study plots averaged 46% to 55% greater soil concentrations than unburned plots. In ponderosa forest (LANL), post-fire soils had ^{137}Cs concentrations much greater (average of 3x times higher) than soils of unburned plots (burned plot ^{137}Cs concentrations at the three study sites averaged respectively 0.006 (±0.003), 0.023 (±0.015) and 0.044 (±0.002) Bq g^{-1}, compared to respective unburned plot averages of 0.004 (±0.001), 0.015 (±0.013), and 0.015 (±0.001)).

The larger increases in post-fire soils in ponderosa pine (LANL) are a result of greater amounts of deposited ash, compared to the very little ash that was observed after fire in grassland and shrubland (RFETS and WIPP). ^{137}Cs concentrations within just the ash alone at LANL averaged 0.192 Bq g^{-1} with a range of 0.007 to 0.570 Bq g^{-1}. These levels are up to 40 times higher than those measured in pre-fire soils.

The elevated ^{137}Cs in ash is thought to have been derived from burning of both canopy and ground surface material. However the greatest portion is expected to have come from ashing of ground surface litter and organic soils. Decomposing organic matter in forest soils can be found in up to six components ranging from a top layer of recognizable plant matter, to decayed wood and charcoal just above the mineral horizon (DeBano et al, 1998). In this study, ^{137}Cs concentrations in unburned organic soils beneath ponderosa pine at LANL averaged 0.10 (± 0.02) Bq g^{-1}, and similarly averaged 0.09 (± 0.07) Bq g^{-1} at three additional western U.S. ponderosa pine locations. By comparison, samples of overstory plants from LANL prior to the fire had much lower ^{137}Cs concentrations, averaging only 0.01 Bq g^{-1} (laboratory-ashed) (Gonzales et al., 2000). This suggests a much a greater pre-burn ^{137}Cs inventory at the ground surface, primarily in organic soils, compared to the above ground biomass.

Until the occurrence of a disturbance such as high-intensity fire, the organic soil layer (duff) at ponderosa pine forest sites appears to be a relatively stable reservoir for elevated fallout radionuclide concentrations. In contrast, the concentrations of ^{137}Cs in unburned soil at the grassland and shrubland sites averaged an order of magnitude lower (0.01 (± 0.003) Bq g^{-1}) consistent with referenced values for soils in the western U.S. (Fresquez et al., 1996) and a previous study by Ulsh et al., (2000). These forest areas containing elevated ^{137}Cs concentrations are subject to increasing risk of catastrophic, intense fires today (Chapter 2) along with the consequent potential for such fires to greatly increase runoff and erosion processes and hence the transport of concentrated fallout radionuclides.

^{137}Cs increases in surface water runoff after fire

Concentrations of ^{137}Cs in unfiltered runoff after fire were elevated up to ~20 times higher at burned rainfall simulation plots in ponderosa pine (averaging 1.2 Bq L^{-1}) compared to unburned plots. In the surrounding watershed, post-fire ^{137}Cs concentrations in runoff were more than two orders of magnitude greater than pre-fire measurements. In terms of total yields of ^{137}Cs in runoff leaving plots (Bq transported per ha, per mm rainfall), the yields in ponderosa forest were more than one order of magnitude greater from burned plots (5.96 KBq mm^{-1} ha^{-1}), than yields from unburned plots (0.15 KBq ha^{-1} mm^{-1}). In shrubland and grassland, increases in total yields from burned plots were about 22 and 4 times greater, respectively.

These large increases in ^{137}Cs transport on burned plots were strongly correlated with increased sediment yields across all sites ($r = 0.96$). Total suspended solid measurements after the fire at LANL reached 27 g L^{-1} for burned plots at LANL and up to 35 g L^{-1} in the runoff from the surrounding burned watershed, compared to an average of 2.3 g L^{-1} for unburned plots.

Duration of elevated ^{137}Cs in runoff.

In ponderosa pine forest where substantial ash deposits remained after burning, the highest post-fire concentrations of ^{137}Cs were found in the initial runoff (2.46 Bq L^{-1} from plots, and 4.0 Bq L^{-1} from the study watershed), with subsequent concentrations lowering as rainfall and runoff removed ash. On the rainfall simulation plots, the concentrations decreased by half during application of about 120 mm of rainfall. Similarly, for runoff from the watershed, ^{137}Cs concentrations in unfiltered runoff returned to near their pre-fire levels during the 166 day interval following the fire when about 240 mm of rain fell on the watershed. When longer time-frames are considered, this rapid redistribution of ^{137}Cs appears as a sharp pulse of elevated concentrations leaving the burned area.

Enrichment effects

Enrichment of ^{137}Cs concentrations in runoff sediments occurred during surface water erosion on burned and unburned study plots. Enrichment ratios comparing ^{137}Cs concentrations in runoff sediments to the concentrations in their parent soils ranged from 1.4 to 2.9. The average enrichment ratio for unburned plots was 2.3 (± 0.6) and for burned plots was 1.6 (± 0.4). The enrichment effect appears to be associated with preferential entrainment of smaller sized particles during surface water erosion. These smaller sized particles (≤ 50 μm) made up greater weight percentages in runoff sediments than in parent soils (average of 18.5% greater for unburned plots and 7.8% greater for burned plots in forest). The smaller size particles also have greater associated concentrations of the ^{137}Cs cation (Lane and Hakonson, 1982), resulting in higher, per weight, concentrations of ^{137}Cs in runoff sediments compared to parent soils.

In repeated tests on plots in grassland, shrubland, and forest, burning of vegetation for all fire types had no consistent effect on enrichment ratios (i.e., enrichment ratios in runoff from burned and unburned plots were not significantly different). Enrichment occurred on both types of plots, and although ^{137}Cs yields were much higher on burned plots because of the associated greater sediment and ash yields, the ratio of enrichment between runoff sediments and parent soils was about the same for burned and unburned surfaces.

Post-fire ^{137}Cs transport and ground cover loss

Integrating results from grassland, shrubland, and forest study sites allowed comparison of ^{137}Cs transport across a range of ecosystems types. Main differences between these sites were vegetation types (grassland to forest) and type of burning. At the ponderosa pine forest site, an intense wildfire burned, while controlled fire was used at the grassland and shrubland sites.

Results show amplified ^{137}Cs transport following burning in ponderosa pine forest where the intense wildfire consumed up to 80% of the ground cover. ^{137}Cs transport was consistently lower from the unburned plots in ponderosa pine forest, and all other plots where vegetation cover was less. Further, ^{137}Cs transport relative to the percentage of bare soil on a plot surface is not linear, but appears to have little change as percent bare soil varies between 0% and about 60-70%. Sharp increases in ^{137}Cs transport occur when the amount of bare soil is greater than 60-70%. This relationship between ^{137}Cs transport and groundcover matches closely the relationship between sediment transport and groundcover (Chapter 2).

Other implications

While this study focuses on concentration and transport of ^{137}Cs, our conclusions are expected to have relevance to reconcentration and redistribution of other particle-reactive fallout radionuclides, metals, and nutrients. Following the Cerro Grande fire at Los Alamos, NM, data gathered by the Los Alamos National Laboratory indicated order of magnitude concentration increases in ^{90}Sr, ^{238}Pu, and $^{239/240}Pu$, barium, manganese, and calcium (among others) in runoff samples collected in burned watersheds (Gallaher et al., 2002). These measurements were taken upstream of the Los Alamos National

Laboratory areas, away from laboratory waste areas. The results reported in this study are believed to be representative of conditions that can occur after fire that can typically be found elsewhere in the U.S. and the world.

In summary, study results provide evidence of order of magnitude reconcentration and rapid redistribution of fallout radionuclides following fire from systematic studies in grassland, shrubland, and forest. These new results have importance for a wide range of interests including ecosystem dynamics studies, geophysical studies, fire management, and risk assessments where periodic fire events may dominate redistribution rates.

REFERENCES

Allen, C.D. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. PhD. diss., Univ. of California Berkley.

Allen, C.D., and D.D. Breshears. 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proc. Natl. Academy of Sci. USA 95:14839-14842.

Amiro, B.D., S.C. Sheppard, F.L. Johnston, W.G. Evenden, and D.R. Harris. 1996. A burning question: What happens to iodine, cesium, and chlorine in biomass fires? Sci. Total Environ. 187(2):93-103.

Auclair, A.N.D. 1977. Factors affecting tissue nutrient concentrations in a Carex meadow. Oecologia (Berl.) 28:233-246.

Baird, M., D. Zabowski, and L. Everett. 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. Plant and Soil 209(2):233-243.

Beeson, P.C., S.N. Martens, and D.D. Breshears. 2001. Simulating overland flow following wildfire: Mapping vulnerability to landscape disturbance. Hydrol. Process. 15:2917-2930.

Bowen, B.M. 1990. Los Alamos climatology. Los Alamos Natl. Lab. LA-11735-MS; UC-902. Natl. Tech. Inform. Serv., Springfield, VA.

Brandes, D., and B.P. Wilcox. 2000. Evapotranspiration and soil moisture dynamics on a semiarid ponderosa pine hillslope. J. Am. Water Resour. Assoc. 36: 965-974.

Bryan, R.B. 2000. Soil erodibility and processes of water erosion on hillslope. Geomorphology 32:385-415.

Campbell, R.E., M.B. Baker Jr., P.F. Ffolliott, F.R. Larson, and C.C. Avery. 1978. Wildfire effects on a ponderosa pine ecosystem: An Arizona case study. RM-191. USDA For. Serv. Rocky Mountain Range Experimental Station, Fort Collins, CO.

Cannon, S. H., E. R. Bigio, and E. Mine. 2001. A process for fire-related debris flow initiation, Cerro Grande Fire, New Mexico. Hydrol. Process. 15:3011-3024.

Cannon, S.H., and S.L. Reneau. 2000. Conditions for generation of fire-related debris flows, Capulin Canyon, New Mexico. Earth Surface Processes and Landforms 25:1103-1121.

Chambers, D.P., and P.M. Attiwill. 1994. The ash-bed effect in eucalyptus-regnans forest: Chemical, physical and microbiological changes in soil after heating or partial sterilization. Aust. J. Bot. 42:739-749.

Covington, W.W., R.L. Everett, R. Steele, L.L. Irwin, T.A. Daer, and A.N.D. Auclair. 1994. Historical and anticipated changes in forest ecosystems in the inland west of the United States. J. Sustainable For. 2:13-63.

Davenport, D.W., D.D. Breshears, B.P. Wilcox, and C.D. Allen. 1998. Viewpoint: Sustainability of piñon-juniper ecosystems: A unifying perspective of soil erosion thresholds. J. Range Manage. 51:231–240.

DeBano, L.F. 1981. Water repellent soils: A state-of-the-art. Gen. Tech. Rep. PSW-46. USDA For. Serv., Pacific Southwest For. and Range Experimental Station: Berkeley, CA; 21.

DeBano, L.F., D.G. Neary, and P.F. Ffolliott. 1998. Fire's effects on ecosystems. John Wiley & Sons, New York. 1998.

Dietrich, W.E., C.J. Wilson, D.R. Montgomery, and J. McKean. 1993. Analysis of erosion thresholds, channel networks, and landscape morphology using a digital terrain model. J. Geo. 101:259-278.

Emmerich, W.E. 1998. Estimating prescribed burn impacts on surface runoff and water quality in southeastern Arizona. Rangeland Manage. and Water Resourc. Am. Water Res. Assoc. May:149-157

Emmerich, W.E., and J.R. Cox. 1992. Hydrologic characteristics immediately after seasonal burning on introduced and native grasslands. J. Range Manage. 45:476-479.

Emmerich, W.E., and J.R. Cox. 1994. Changes in surface runoff and sediment production after repeated rangeland burns. Soil Sci. Soc. Am. J. 58:199-203.

Essington, E.H., and E.M. Romney. 1986. Mobilization of ^{137}Cs during rainfall simulation studies at the Nevada Test Site. p. 35-38. *In* L.J. Lane (ed.) Erosion on rangelands: Emerging technology and data base. Proc. of Rainfall Simulation Workshop, Tucson, AZ. 14-15 Jan. 1985. Soc. Range Manage., Denver, CO.

Flanagan, D.C., and M.A. Nearing. 1990. Sediment enrichment in the WEPP model. ASAE Paper 90-2079. Am. Soc. Agric. Eng., St. Joseph, MI.

Foster, G.R., anc T.E. Hakonson. 1987. Erosional losses of fallout plutonium. *In* Environmental Research on Actinide Elements. USDOE CONF-841142(DE86008713) August 1987. Natl. Tech. Inform. Serv., Springfield, VA.

Foxx, T.S. 1984. La Mesa Fire Symposium. Los Alamos New Mexico, October 6-7, 1981. Los Alamos Natl. Lab. Rep. LA-9236-NERP. Natl. Tech. Inform. Serv., Springfield, VA.

Frasier, G.W., M. Weltz, and L. Weltz. 1998. Rainfall simulator runoff hydrograph analysis. J. Range Manage 51:531-535.

Fresquez, P.R., M.A. Mullen, J.K. Ferenbaugh, and R.A. Perona. 1996. Radionuclides and radioactivity in soils within and around Los Alamos National Laboratory, 1972 through 1994: Concentrations, trends, and dose comparisons. Los Alamos Natl. Lab. Rep. LA-13149-MS. Los Alamos, NM.

Fresquez, P.R., W.R. Velasquez, L. Naranjo, Jr. 2001. The effects of the Cerro Grande fire (smoke and fallout ash) on possible contaminants in soils and crops downwind of Los Alamos National Laboratory. Los Alamos Natl. Lab. Rep. LA-13842-MS. Los Alamos, NM.

Gallaher, B., R. Koch, and K. Mullen. 2002. Quality of storm runoff at Los Alamos National Laboratory in 2000 with emphasis on the Cerro Grande fire. Los Alamos Natl. Lab. Rep. LA-13926. Los Alamos, NM.

Gonzales, G.J., P.R. Fresquez, M.A. Mullen, and L. Naranjo, Jr. 2000. Radionuclide concentrations in vegetation at the Los Alamos National Laboratory in 1998. Los Alamos Natl. Lab. Rep. LA-13704-PR. Los Alamos, NM.

Graf, W.H. 1971. Hydraulics of sediment transport. McGraw-Hill Book Co., New York.

Hakonson, T.E. 1999. The effects of pocket gopher burrowing on water balance and erosion from landfill covers. J. Environ. Qual. 28:659-665.

Hakonson, T.E., and L. J. Lane. 1993. The role of physical process in the transport of man-made radionuclides in arid ecosystems. p.101-176. *In* R.M. Harrison (ed.) Biogeochemical pathways of artificial radionuclides. John Wiley & Sons, New York.

Hakonson, T.E., L.J. Lane, G.R. Foster, and J.W. Nyhan. 1986. An overview of Los Alamos research on soil and water processes in arid and semi-arid ecosystems. p. 7-10. *In* L.J. Lane (ed.) Erosion on rangelands: Emerging technology and data base. Proc. of Rainfall Simulation Workshop, Tucson, AZ, 14-15 Jan. 1985. Soc. Range Manage., Denver, CO.

Hester, J.W., T.L. Thurow, and C.A. Taylor, Jr. 1997. Hydrologic characteristics of vegetation types as affected by prescribed burning. J. Range Manage. 50:199-204.

Hulse, S.E., S.A. Ibrahim, F.W. Whicker, and P.L. Chapman. 1998. Comparison of ^{241}Am, ^{239,240}Pu, and ^{137}Cs concentrations in soil around Rocky Flats. Health Physics. 76:275-286.

Interagency BAER. 2000. Burned Area Emergency Rehabilitation plan, Cerro Grande Fire, Los Alamos, New Mexico, May – June, 2000. USDA, For. Serv., Santa Fe Natl. For., Santa Fe, NM.

IPCC, 2001. Climate Change 2001: Synthesis Report. A Contribution of Working Groups I, II, and II to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Watson, R. T., and the Core Writing Team (eds.). Cambridge University Press, NY.

Johansen, M.P., T.E. Hakonson, and D.D. Breshears. 2001a. Post-fire runoff and erosion from rainfall simulation: Contrasting forests with shrublands and grasslands. Hydrol. Process. 15:2953-2965.

Johansen, M.P., T.E. Hakonson, F. W. Whicker, J. R. Simanton, and J.J. Stone. 2001b. Hydrologic response and radionuclide transport following fire at semiarid sites. J. Environ. Qual. 30:2010-2017.

Kashparov, V.A., A.M. Lundin, V.P. Kadygrib, V.P. Prostak, S.E. Levtchuk, V.I. Yoschenko, V.A. Kashpur, and N.M. Talerko. 2000. Forest fires in the territory contaminated as a result of the Chernobyl accident: Radioactive aerosol resuspension and exposure of fire fighters. J. Environ. Radioact. 51(3):281-298.

Kauffman, J.B., R.L. Sanford, Jr., D.L. Cummings, I.H. Salcedo, and E.V.S.B. Sampaia. 1993. Biomass and nutrient dynamics associated with slash fires in neotropical dry forests. Ecology 74:140-151.

Kendall, C., and J.J. McDonnell. 1998. Isotope tracers in catchment hydrology. Elsevier Science B.V., Amsterdam, The Netherlands.

Knight, R.W., W.H. Blackburn, and C.J. Scifres. 1983. Infiltration rates and sediment production following herbicide/fire brush treatments. J. Range Manage. 36:154-157.

Lane, L.J., and T.E. Hakonson. 1982. Influence of particle sorting on transport of sediment associated contaminants. p. 543-557. *In* Proc. of Waste Manage. 1982 Symp. Univ. of Arizona Press, Tucson.

Lane, L.J., T.E. Hakonson, G.R. Foster. 1986. Watershed erosion and sediment yield affecting contaminant transport. *In* Proc. of the USDOE Symp. for Environ. Res. on Actinide Elements, Hilton Head, SC. November 1983. CONF- 841142. U.S. Dep. of Energy, Washington, DC.

Lane, L.J., M. Hernandez, M. Nichols. 1997. Processes controlling sediment yield from watersheds as function of spatial scale. Environ. Modelling Software 12:355-369.

Levy, E.B., and E.A. Madden. 1933. The point method of pasture analysis. N.Z. J. Agric. 46:267-279.

Los Alamos National Laboratory. 2001. Environmental surveillance at Los Alamos during 2002. Los Alamos Natl. Lab. Rep. LA-13861-ENV. Los Alamos, NM

Malmon, D.V., T. Dunne, and S.L. Reneau. 2002. Predicting the fate of sediment and pollutants in river floodplains. Environ. Sc. and Tech. 36:2026-2032.

Marcos, E., R. Tarrega, and E. Luis-Calabuig. 2000. Comparative analysis of runoff and sediment yield with a rainfall simulator after experimental fire. Arid Soil Res. Rehab. 14:293-307.

Massey, H.F., and M.L. Jackson. 1952. Selective erosion of soil fertility constituents. Proc. SSSA 16:353-356.

Mast, J.N., P.Z. Fulé, M.M. Moore, W.W. Covington, and A.E.M. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecol. Applic. 9:228-239.

McHenry, J.R., and J.C. Ritchie. 1977. Physical and chemical parameters affecting transport of ^{137}Cs in arid watersheds. Water Resour. Res. 13:923-926.

McLin, S.G., E.P. Springer, and L.J. Lane. 2001. Predicting floodplain boundary changes following the Cerro Grande wildfire. Hydrol. Process. 15:2967-2980.

Menzel, R.G. 1980. Enrichment ratios for water quality modeling. p. 486-492. *In* W.G. Knisel (ed.) CREAMS: A field scale model for chemicals, runoff, and erosion from agricultural management systems. Conserv. Res. Rep. 26, III. USDA, Washington DC.

Moore, M.M., W.W. Covington, and P.Z. Fulé. 1999. Reference conditions and ecological restoration: A Southwestern ponderosa pine perspective. Ecol. Applic. 9:1266-1277.

National Research Council. 2000. National Research Council report; Long-term institutional management of U.S. Department of Energy legacy waste sites. Natl. Academy Press, Washington DC.

Newman, B.D., A.R. Campbell, and B.P. Wilcox . 1997. Tracer-based studies of soil water movement in semi-arid forest of New Mexico. J. of Hydrol. 196:251-270.

Newman, B.D., A.R. Campbell, and B.P. Wilcox . 1998. Lateral subsurface flow pathways in a semiarid ponderosa pine hillslope. Water Resour. Res. 34:3485-3496.

Nyhan, J.W., T.E. Hakonson, B.J. Drennon. 1990. A water balance study of two landfill cover designs for semiarid regions. J. Environ. Qual. 19:281-288.

Paliouris, G., H.W. Taylor, R.W. Wein, J. Svoboda, and B. Mierzynski. 1995. Fire as an agent in redistributing fallout Cs-137 in the Canadian boreal forest. Sci.Total Environ. 161:153-166.

Parra, J.G., V.C. Rivero, and T.I. Lopez. 1996. Forms of Mn in soils affected by a forest fire. Sci. Total Environ. 181:231-236.

Peles, J.D., A.L. Bryan, Jr., C.T. Garten, Jr., D.O. Ribble, and M.H. Smith. 2000. Ecological half-life of ^{137}Cs in fish from a stream contaminated by nuclear reactor effluents. Sci. Total Environ. 263:255-262.

Piñol, J., J. Terradas, and F. Lloret. 1998. Climate warming, wildfire hazard, and wildfire occurrence in coastal eastern Spain. Climatic Change 38: 345-357

Quinton, J.N., J.A. Catt, T.M. Hess. 2001. The selective removal of phosphorus from soil: Is event size important? J. Environ. Qual. 30:538-545.

Reid, K.D., B.P. Wilcox, D.D. Breshears, and L. MacDonald. 1999. Runoff and erosion in a piñon-juniper woodland: Influence of vegetation patches. Soil Sci. Soc. of Am. J. 63:1869-1879.

Renard, K.G. 1985. Rainfall simulators and USDA erosion research: History, perspective, and future. *In* L. J. Lane (ed.) Erosion on rangelands: Emerging technology and data base. Proc. of the Rainfall Simulator Workshop, Tucson, AZ, 14-15 Jan. 1985. Soc. Range Manage., Denver, CO.

Robichaud, P.R. 2000. Fire effects on infiltration rates after prescribed fire in northern Rocky Mountain forests, USA. J. Hydrol. 231:220-229.

Robichaud, P.R., R.T. Graham, and R.D. Hungerford. 1994. Onsite sediment production and nutrient losses from a low severity burn in the interior northwest. Proc. Interior Cedar-Hemlock-White Pine Forests: Ecology and Manage., Spokane, WA, 2-4 Mar. 1993. Dep. Natural Res. Sci., Washington State Univ., Pullman, WA.

Robichaud, P.R., and T.A. Waldrop. 1994. A comparison of surface runoff and sediment yields from low- and high-severity site preparation burns. Water Res. Bullet. 30:27-34.

Roundy, B.A., W.H. Blackburn, and R.E. Eckert, Jr. 1978. Influence of prescribed burning on infiltration and sediment production in the pinyon-juniper woodland, Nevada. J. Range Manage. 31:250-253.

Sackett, S.S., and S.M. Haase. 1999. Two case histories using prescribed fire to restore ponderosa pine ecosystems in northern Arizona. Proc. of the 20th Tall Timbers Fire Ecology Conference: Fire in Ecosystem Manage., May, 1999.

Seyfried, M.S., and B.P. Wilcox. 1995. Scale and the nature of spatial variability: Field examples having implications for hydrological modeling. Water Res. Res. 31:173-184.

Shahlaee, A.K., W.L. Nutter, E.R. Burroughs, Jr., and L.A. Morris. 1991. Runoff and sediment production from burned forest sites in the Georgia Piedmont. Water Resourc. Bullet. 27:485-493.

Shakesby, R.A., S.H. Doerr, R.P.D. Walsh. 2000. The erosional impact of soil hydrophobicity: Current problems and future research directions. J. Hydrol. 231:178-191.

Simanton, J.R., and W.E. Emmerich. 1994. Temporal variability in rangeland erosion processes. *In* Variability of Rangeland Water Erosion Processes; SSSA Special Publ. 38: 51-65.

Simanton, J.R., C.W. Johnson, J.W. Nyhan, and E.M. Romney, 1986. Rainfall simulation on rangeland erosion plots. p. 11-17. *In* L. J. Lane (ed.) Proc. of the Rainfall Simulator Workshop, Tucson, AZ, 14-15 Jan. 1986. Soc. Range Manage., Denver, CO.

Simanton, J.R., G.D. Wingate, and M. Weltz. 1990. Runoff and sediment from a burned sagebrush community. p.180-185. *In* Effects of fire management on southwestern resources, proceedings of the symposium. For. Serv. General Tech. Rep. RM-191. USDA, Fort Collins, CO.

Soto, B., and F. Diaz-Fierros. 1998. Runoff and soil erosion from areas of burnt scrub: comparison of experimental result with those predicted by the WEPP model. Catena 31:257-270.

Stauffer, D. 1985. Introduction to percolation theory. Taylor and Francis, London.

Swanson, N.P. 1965. Rotating-boom rainfall simulator. Trans. ASAE 8:71-72.

Swetnam, T.W., C.D. Allen, and J.L. Betancourt. 1999. Applied historical ecology: Using the past to manage the future. Ecol. Appl. 9:1189-1206.

Swetnam, T.W., and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. p. 11-32. *In* C.D. Allen (ed.) Fire Effects in Southwestern Forests. Proc. of the Second La Mesa Fire Symposium. USDA For. Serv. General Tech. Rep. RM-GTR-286. USDA For. Serv. Rocky Mountain Range Experimental Station: Fort Collins, CO.

Swetnam, T.W., and J.L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climate variability in the American Southwest. J. Climate 11:3128-3147.

Thomas, A.D., R.P.D. Walsh, and R.A. Shakesby. 1999. Nutrient losses in eroded sediment after fire in eucalyptus and pine forest in the wet Mediterraniean environment of northern Portugal. Catena 36(4):283-302.

Touchan, R., C.D. Allen, and T.W. Swetnam. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, Northern New Mexico. p. 33-46. *In* C.D. Allen (ed.) Fire effects in southwestern forests. Proc. of the Second La Mesa Fire Symposium. USDA For. Serv. General Tech. Rep. RM-GTR-286. USDA For. Serv. Rocky Mountain Range Experimental Station: Fort Collins, CO.

Tysdal, L. 2000. Erosion, sediment yield, and actinide migration near the 903 pad at the Rocky Flats Environmental Technology Site, Golden, Colorado. M.S. thesis. Colorado State Univ., Fort Collins.

Ulsh, B., S. Rademacher, F.W. Whicker. 2000. Variations of ^{137}Cs depositions and soil concentrations between alpine and montane soils in northern Colorado. J. Environ. Activity 47:57-70.

U.S. Department of Energy. 1977. Waste Isolation Pilot Plant disposal phase final supplemental environmental impact statement. Volume 1. U.S. Department of Energy, Washington D.C.

Watters, R.L., T.E. Hakonson, and L.J. Lane. 1983. The behavior of actinides in the environment. Radiochemical Acta 32:89-103.

Weigand, S., W. Schimmack, and K Auerswald. 1998. The enrichment of Cs-137 in the soil loss from small agricultural watersheds. Zietschrift fur pflanzenernahrung und bodenkunde 161:479-484.

Weltz, M.A., M.R. Kidwell, and H.D. Fox. 1998. Influence of abiotic and biotic factors in measuring and modeling soil erosion on rangelands. J. Range Manage. 51:482-495.

Whicker, F.W., and V. Schultz. 1982. Radioecology: Nuclear energy and the environment. Vol. 1. CRC Press, Boca Raton, FL.

Whicker, F.W., G. Shaw, G. Voigt, and E. Holm. 1999. Radioactive contamination: State of the science and its application to predictive models. Environ. Pollut. 100:133-149.

Whicker, J.J., D.D. Breshears, P.T. Wasiolek, T.B. Kirchner, R.A. Tavani, D.A. Schoep, and J.C. Rodgers. 2002. Temporal and spatial variation of episodic wind erosion in unburned and burned semiarid shrubland. J. Environ. Qual. 31:599-612.

White, W.D. 1996. Geomorphic response to six headwater basins fifteen years after the La Mesa Fire. *In* C.D. Allen (ed.) Fire Effects in Southwestern Forests. Proc. Second La Mesa Fire Symposium. For. Serv. General Tech. Rep. RM-GTR-286. USDA For. Serv. Rocky Mountain Range Experimental Station: Fort Collins, CO; 95-113.

White, W.D., and Wells SG. 1984. Geomorphic effects of La Mesa Fire. p. 73-90. *In* T.S. Foxx (ed.) La Mesa Fire Symposium. LA-9236-NERP. Natl. Tech. Inform. Serv., Springfield, VA.

Wilcox, B.P., and D.D. Breshears. 1998. Interflow in semiarid environments: An overlooked process in risk assessment. Human and Ecol. Risk Assess. 3:187-203.

Wilcox, B.P., B.D. Newman, C.D. Allen, K.D. Reid, D. Brandes, J. Pitlick, and D.W. Davenport. 1996. Runoff and erosion on the Pajarito Plateau: Observations from the field. New Mexico Geol. Soc. Guidebook, 47th Field Conf. Jemez Mountains Region. New Mexico Geol. Soc., Socorro.

Wilcox, B.P., B.D. Newman, D. Brandes, D.W. Davenport, and K. Reid. 1997. Runoff from a semiarid ponderosa pine hillslope site in New Mexico. Water Resour. Res. 33:2301-2314.

Wilcox, B.P., K.M. Wood, and J.M. Tromble. 1988. Factors influencing infiltrability of semiarid mountain slopes. J. Range Manage. 41:197-207.

Wilson, C.J. 1999. Effects of logging and fire on runoff and erosion on highly erodible granitic soils in Tasmania. Water Resour. Res. 35:3531-3546.

Wilson C.J., J.W. Carey, P.C. Beeson, M.O. Gard, and L.J. Lane. 2001. A GIS-based hillslope erosion and sediment deliver model and its application in the Cerro Grande burn area. Hydrol. Process. 15:2995-3010.

Appendix

Table A.1 Surface characteristics of study plots at WIPP, RFETS, and LANL

		slope	rough-ness	ground cover (% area)						canopy cover (% area)				
				bare soil	gravel	rock >20mm	non-persis. litter	persis. litter	basal veg.	forbs	grass	shrub	Stand. dead	none
		(%)	(cm^2)	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)
WIPP														
Plot 2	Unb	7.1	2.2	17	4	0	36	0	43	11	71	3	0	16
Plot 3	Unb	5.6	1.6	25	13	0	41	0	21	12	64	0	1	23
Plot 5	Unb	6.7	2.3	23	11	1	23	0	42	14	73	0	1	12
Plot 1	burn	6.0	2.5	48	12	0	28	0	12	0	0	0	0	0
Plot 4	burn	5.8	2.2	48	6	1	36	0	9	0	0	0	0	0
Plot 6	burn	6.2	2.0	41	5	0	43	0	11	0	0	0	0	0
RFETS														
Plot 2	unb	8.4	1.1	35	2	0	3	31	29	24	34	7	4	31
Plot 4	unb	8.8	1.2	21	2	3	4	30	38	23	44	4	6	23
Plot 6	unb	9.5	1.2	27	2	1	2	38	30	27	40	5	1	27
Plot 1	burn	9.7	2.4	39	2	3	22	16	18	0	0	0	0	0
Plot 3	burn	9.0	2.3	28	3	0	17	18	34	0	0	0	0	0
Plot 5	burn	8.9	1.0	40	3	0	16	14	27	0	0	0	0	0
LANL														
Plot 1	unb	4.0	1.4	38	0	0	18	17	27	12	8	0	9	71
Plot 2	unb	4.8	1.1	58	0	0	18	8	16	14	12	0	15	59
Plot 3	burn	7.7	1.8	69	0	2	27	0	2	0	0	0	1	99
Plot 4	burn	6.8	1.3	80	0	1	13	0	6	0	0	0	1	99

Table A.2 Soil characteristics of study plots at WIPP, RFETS, and LANL.

	dry bulk density	CEC	org. matter	sand	silt	clay
	$g\ cm^{-3}$	$meq\ 100g^{-1}$	%	%	%	%
WIPP						
	1.34	8.8	.4	91.1	3.8	5.1
	(±0.2)	(±2.2)	(±0.2)	(±2.7)	(±2.0)	(±1.7)
RFETS						
	1.30	27.5	2.6	34.3	21.2	44.4
	(±0.3)	(±2.6)	(±0.6)	(±5.6)	(±4.6)	(±6.8)
LANL						
unb. plots		10.4	3.0	35	53	12
	(±)	(±2.2)	(±0.4)	(±1.4)	(±1.4)	(±0.0)
burned plots						
		14.5	3.5	45	41	14
	(±)	(±1.7)	(±0.1)	(±1.4)	(±4.2)	(±2.8)

Table A.3 Runoff, sediment yields, and ^{137}Cs yields at WIPP, RFETS, and LANL.

		Antec. Moisture	Time to runoff	Runoff ($mm\ mm^{-1}$)			Sediment ($Kg\ ha^{-1}\ mm^{-1}$)			^{137}Cs Yield ($Bq\ ha^{-1}\ mm^{-1}$)		
		Dry,wet, Very-wet	Dry,wet, Very-wet	Dry	Wet	Very-wet	Dry	Wet	Very-wet	Dry	Wet	Very-wet
WIPP												
Plot 2	Unb	0,6,13	na, na, na	0.00	0.00	0.00	0.0	0.0	0.0	0	0	0
Plot 3	Unb	0,6,14	na, na,9.5	0.00	0.00	0.02	0.0	0.0	0.4	0	0	6
Plot 5	Unb	0,8,14	na,30.0,10.0	0.00	0.00	0.03	0.0	0.0	2.0	0	0	7
Plot 1	burn	0,4,12	na, na,14.1	0.00	0.00	0.04	0.0	0.0	2.7	0	0	23
Plot 4	burn	0,4,13	na,13.5,3.5	0.00	0.02	0.30	0.0	0.9	5.3	0	15	92
Plot 6	burn	0,7,13	na,5.7,5.8	0.00	0.10	0.16	0.0	3.1	2.6	0	84	71
RFETS												
Plot 2	unb	12,25,32	5.0,5.7,1.6	0.24	0.34	0.54	1.5	1.8	2.3	52	64	99
Plot 4	unb	11,29,36	6.0,3.3,1.2	0.30	0.52	0.79	2.1	2.0	2.3	19	54	68
Plot 6	unb	12,32,34	11.8,6.1,1.7	0.32	0.43	0.66	1.8	1.7	1.9	38	39	51
Plot 1	burn	12,28,34	5.0,6.8,1.9	0.29	0.29	0.55	4.6	4.5	7.2	113	195	373
Plot 3	burn	12,30,40	6.0,6.0,2.2	0.39	0.61	0.67	4.9	4.0	4.5	82	203	190
Plot 5	burn	15,30,37	11.5,6.0,1.7	0.42	0.56	0.76	3.4	4.5	3.5	124	209	265
LANL												
Plot 1	unb	4,14,19	5.0,5.0,5.0	0.19	0.20	0.31	2.9	2.3	3.4	165	87	171
Plot 2	unb	5,13,18	5.0,4.5,4.4	0.21	0.18	0.31	3.8	3.0	4.2	199	105	195
Plot 3	burn	5,14,18	3.0,2.5,1.0	0.25	0.35	0.32	28.2	48.4	41.4	2432	3028	2738
Plot 4	burn	5,15,19	2.5,1.5,1.0	0.58	0.58	0.64	109.9	112.8	113.3	11633	8275	7671

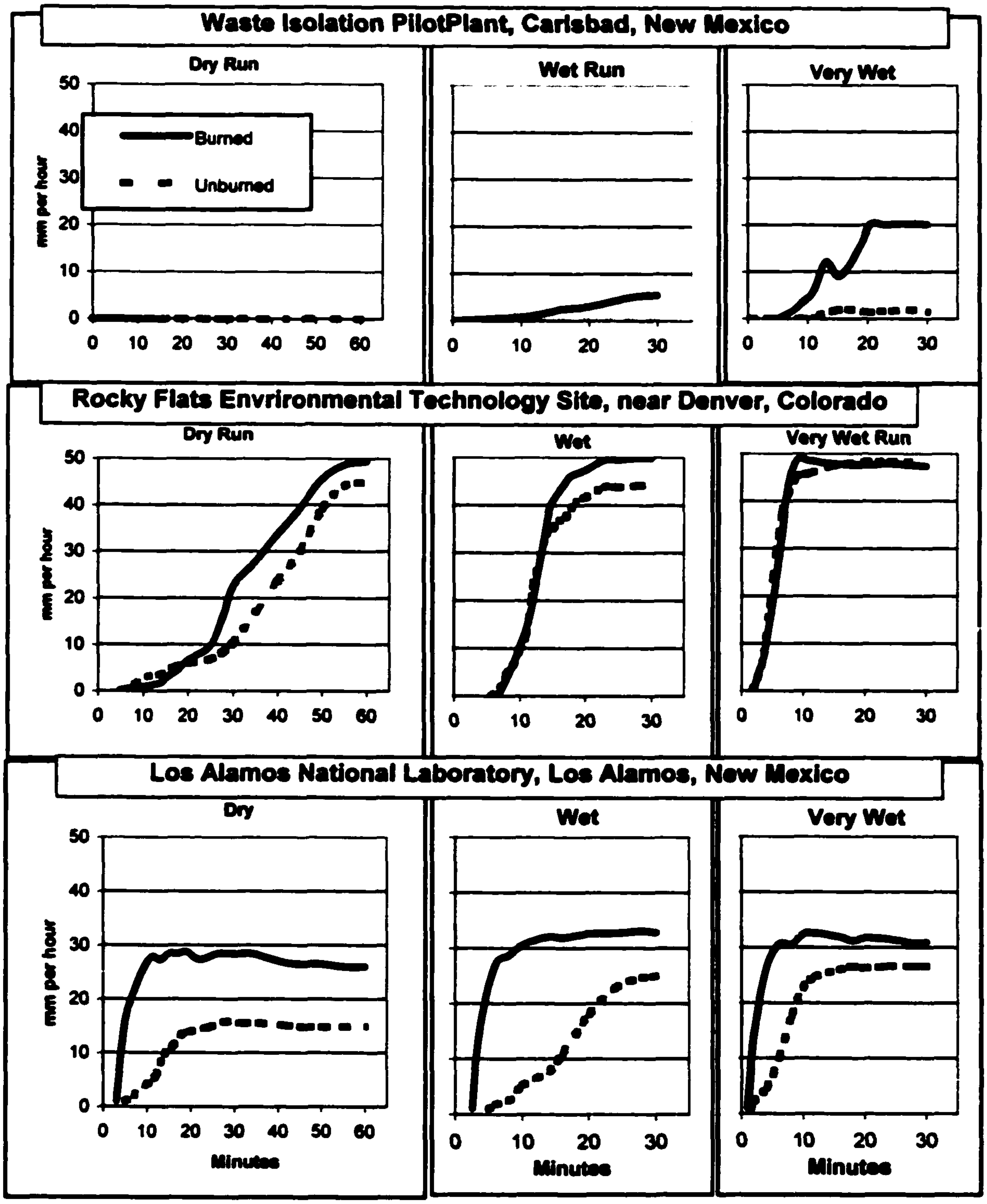

Fig. A.1. Average runoff hydrographs at WIPP (3 unburned and 3 burned), RFETS (3 unburned and 3 burned), and LANL (2 unburned and 2 burned).

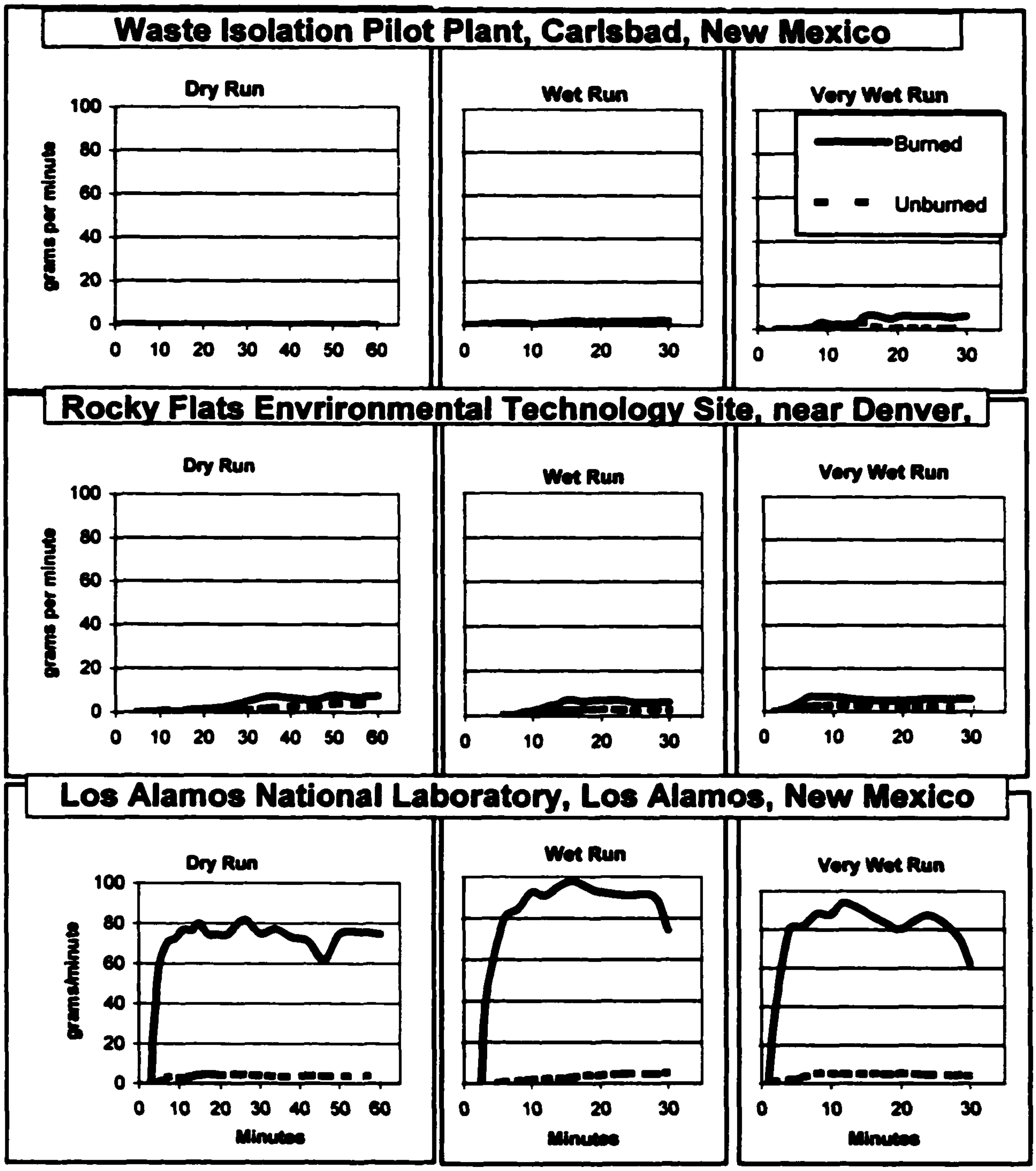

Fig. A.2. Average sediment hydrographs at WIPP (3 unburned and 3 burned), RFETS (3 unburned and 3 burned), and LANL (2 unburned and 2 burned).