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Thinning semiarid forests amplifies wind erosion comparably to wildfire: Implications for restoration and soil stability

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Abstract

Semiarid forests across the western USA and elsewhere are being thinned to reduce risk from fire, restore previous ecological conditions, and/or salvage trees from recently burned areas. Prescriptions and monitoring for thinning generally focus on biotic characteristics of vegetation, like tree density, rather than abiotic characteristics of soils and their loss, which are usually only considered in association with water erosion. Recent studies indicate that sediment transport by wind in forests is substantial and can exceed water transport, yet forest wind erosion responses to tree thinning and/or burning are unknown. We measured wind-driven horizontal dust flux, a metric related to wind erosion, with respect to presence/absence of fire and/or thinning in a ponderosa pine (*Pinus ponderosa*) forest in northern New Mexico, USA. Passive dust collectors at several sampling heights documented elevated dust fluxes at sites that were burned and/or thinned. Unexpectedly, thinned sites had erosion rates as large as burned sites, documenting significant restoration impacts on soil stability. Thinning and fire impacts on dust flux were correlated with remaining tree or ground cover. The results highlight that dust fluxes provide a readily measurable metric of soil stability that should be integrated into prescription and monitoring plans for forest restoration and thinning.

Keywords: Environmental disturbance; Forest fire; Forest thinning; Restoration; Soil stability; Wind erosion

1. Introduction

Semiarid forests are globally extensive (Brekle, 2002) and provide many important ecosystems goods and services (Millennium Ecosystem Assessment, 2005). Of particular note are ponderosa pine (*Pinus ponderosa* Laws.) forests, which occupy many millions of hectares in the mountains of the southwestern USA, because the structure of these semiarid forests has changed dramatically since Euro-American settlement.

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Tree densities have increased by a factor of 10 or more resulting in dramatically altered ecosystem function and higher risk of severe wildfires (Covington et al., 1997; Friederici, 2003). Strategies for reducing risk of severe fires and for restoring previous ecosystem function have mostly involved thinning tree densities to presettlement levels (Covington, 2003). More recently, forest thinning strategies have been expanded to include additional procedures designed to restore ecological integrity such as promoting high biological diversity and including heterogeneity in landscape forest patterns and tree age (Allen et al., 2002; Lynch et al., 2000; Mast, 2003).

Notably, these goals and the metrics used for evaluating success following forest thinning are largely focused on biotic characteristics and properties of forests, with less emphasis on abiotic factors such as soil quality[2] that can significantly impact biotic properties (Fisher and Binkley, 2000; Kimmins, 2003; Wardle et al., 2004). Thinning procedures can impact soil quality through numerous mechanisms (Selmants et al., 2003) including the potential for increased soil erosion, which has been shown to adversely impact soil quality if excessive (Kaiser, 2004; Lal et al., 2003; Pimental, 2000; Toy et al., 2002). In contrast, high soil quality is generally associated with high soil stability and low rates of soil erosion (Baker and Jemison, 1991; Lal et al., 1996; Tongway and Ludwig, 1997). The removal of tree canopy and ground cover during thinning can potentially increase soil erosion as has been documented following fires (Johansen et al., 2001; Vermeire et al., 2005; Whicker et al., 2002, 2006b; Zobeck et al., 1989).

Potential increases in soil erosion following forest disturbance could be driven by either water or by wind. Numerous studies have demonstrated increased post-fire water-erosion in a variety of ecosystems including semiarid forests (Freedman, 1995; Johansen et al., 2001; Rodgers, 1996) and water erosion is a major consideration for forest management (Aber et al., 2000; LANL, 2001). Fewer studies have evaluated wind erosion rates in semiarid forests, yet recent studies have provided initial estimates and conclude that wind-driven erosion and transport rates can exceed water-driven erosion rates in some undisturbed semiarid systems (Breshears et al., 2003). These and other studies in arid and semiarid ecosystems suggest that the importance of wind erosion and dust transport may be under appreciated, particularly for disturbed systems. Despite the potential importance for soil quality and ecosystem function, there are few studies quantifying the impact of forest wildfire on wind erosion and none that evaluate wind erosion in response to tree thinning or how the two compare.

Potential incorporation of the consequences of wind erosion into restoration plans will require comparable measurements of wind erosion rates for undisturbed, burned, and thinned forests. Rate estimates will also be required for sites undergoing forest restoration with various intensities of fire and thinning, especially where heavy machinery is used. We address this issue by evaluating wind erosion in areas within a semiarid ponderosa pine forest in northern New Mexico that were burned or not burned and then subsequently thinned or not thinned. We build upon a previous study showing increased wind erosion in burned forests (Whicker et al., 2006b) and discuss the implications of our findings, which document that thinning activities, at least initially, increased wind erosion rates to levels comparable to those following burning.

2. Methods

2.1. Study sites

The study sites are located in the Jemez Mountains in north central New Mexico, USA (at general area coordinates of 35°52′ N; 106°19′ W), at an elevation of about 2300 m, and are within the boundaries of the western edge of Los Alamos National Laboratory (LANL) (Fig. 1). Average annual precipitation in the area is about 500 mm (Bowen, 1990). Woody vegetation is comprised mostly of ponderosa pine (*Pinus ponderosa* Laws. Var. Scopulorum Engelm.; nomenclature follows Martin and Hutchins, 1980) and a sparse understory

[2]The generic term ''soil quality'' is used in this paper as stated by the United States Department of Agriculture ''as the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation'' (http://soils.usda.gov/sqi/concepts/concepts.html) [verified 9/06].

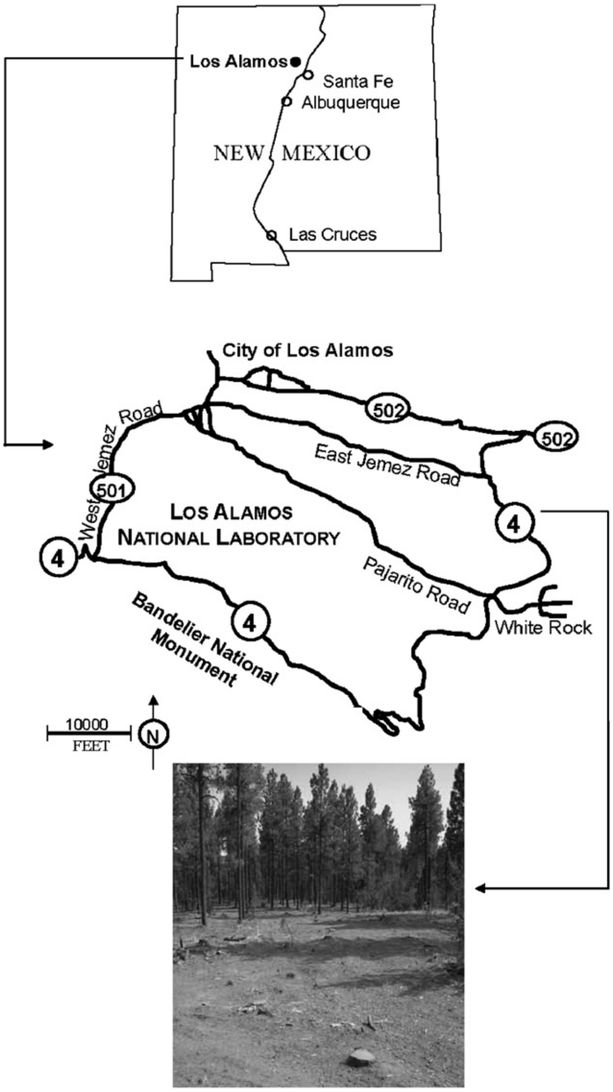

Fig. 1. Map showing the location of the study area. Plots were located in the western portion of Los Alamos National Laboratory and just east of highway 501. Bottom picture is one of the forest plots after thinning.

of gamble oak (*Quercus gambelii* Nutt.). Ground vegetation includes a variety of grasses and forbs (Foxx and Hoard, 1995).

In May 2000, the Cerro Grande fire burned approximately 16,000 ha of mostly ponderosa pine forest, including about 3000 ha of land and some structures and facilities within Los Alamos National Laboratory (LANL, 2000). To reduce the potential for future fire damage to nuclear facilities and to improve the forest "health", approximately 4500 ha of ponderosa pine forests were thinned on LANL property between the years 2000 and 2003 (LANL, 2001). Both unburned forest and moderately burned areas with surviving trees were thinned. Because of the large amount of land needing thinning and short time frames, mechanized techniques were used including the use of track-mounted feller bunchers (manufactured by Timbco), skidders, stroke delimbers, and logging trucks to haul off the lumber (Quam, 2004). The thinning techniques used at LANL were perhaps more mechanized than those which will be employed at some locations, but are likely candidates for large-scale, cost-efficient forest thinning required in vast areas of the southwestern USA.

2.2. Characterization of sampling sites

2.2.1. Sampling locations

Sampling was done in thinned and unthinned areas that were either burned or unburned. The unburned sites had no recent history of fire with ground vegetation of perennial grasses and a cover of pine needles, whereas the burned areas had their ground vegetation and litter cover consumed in the fire. The selected burned sites were moderately burned, which was defined as a fire that completely consumed vegetation and litter cover of the forest floor and produced scorching of pine needles in $\leqslant 75\%$ of the crowns of the trees (Pinder et al., 2004). Many of the scorched needles dropped in the years following the fire to produce a thin litter layer that did not completely cover the bare soil (Pinder et al., 2004). Sampling in these moderately burned areas was done because they represent areas that are likely candidates for post-fire thinning to reduce subsequent fire hazards and for salvaging of dead trees for commercial purposes. Thinning was not performed in severely burned (complete tree burning including the crown) areas so these areas were not included in this study.

Two stands were located in each of four forest categories that included: (1) unburned and unthinned, (2) unburned and thinned, (3) burned and unthinned, and (4) burned and thinned. Two replicate stands were established within each forest category with one stand designated as a ‘‘primary’’ site and the other was designated as a ‘‘secondary’’ site. The replicate sites were separated by less than 1 km. Primary sampling sites contained three dust-flux sampling stations and, due to a limited number of samplers, the secondary sites had only two dust-flux sampling stations. Dust-flux samplers were arranged at 30 m intervals through the stand. Sampling sites were selected based on the following criteria: (1) a relatively homogeneous tree structure, (2) more than 100 m from roadways or disturbed areas to ensure an adequate fetch length within the stand (Baldocchi, 1997), (3) a relatively flat terrain (i.e., slopes less than 10%), and (4) representative soil and vegetation types for the region.

Thinning occurred in the study areas starting with the primary unburned site in the fall of 2001 and continuing with the other sites during the fall of 2002. Thus, dust-flux measurements were made from March 2002 through July 2003 at the primary unburned/unthinned and primary unburned/thinned site. Dust-flux measurements for the rest of the sites were made from March 2003 to July 2003.

2.2.2. Soil characterization

To measure soil texture, samples of the top 2 cm of soil were collected from five random locations in primary sites and three random locations in the secondary sites. For each sampling location, twenty 38 mm diameter soil cores were obtained, composited, air-dried to a constant mass, and a 200 g aliquot was submitted to the Soil, Water, and Plant Testing Laboratory at Colorado State University for analyses of soil texture [% sand, % silt, and % clay (Miller and Gardiner, 2001)] by a modified pipette method (Indorante et al., 1990). Texture was determined without dispersing the samples.

2.2.3. Characterization of tree density and canopy cover

Tree densities at unthinned sites were measured in March of 2002 in five 100-m^2 plots within the primary sites and in four 100-m^2 plots within the secondary sites, and measured at thinned sites in September of 2002 in 10 and seven 100-m^2 plots within the primary and secondary sites, respectively. All plot locations were randomly selected, and more plots for tree density measurements were used in the thinned sites because of the reduced tree density. In unthinned plots, the number of live trees and trees killed by scorching in the fire were recorded and used to estimate densities. In the thinned plots, the number of trees and freshly cut tree stumps within a 100-m^2 area were determined and used to estimate density before thinning or before fire and thinning. Heights of standing trees were measured using a forester's altimeter (Avery and Burhart, 1994). The percent canopy cover of live trees was estimated from measurements made using a spherical densitometer (Lemmon, 1956). Tree canopy measurements were made in 15–22 locations distributed through a 60 m $\times$ 120 m area surrounding the dust-flux samplers in primary sites and in 12 locations distributed through a 60 m $\times$ 90 m area in secondary sites.

2.2.4. Ground cover

Ground cover was measured using 3.2 Mega pixel digital images of 1-m^2 plots at randomly selected locations, with 10 plots at primary sites and seven plots in the smaller secondary sites for a total of 17 plots areas within each of the four burn/thin forest categories. The ground cover images included a 1-m^2 frame, and the percent cover for each of the vegetation/cover plots was determined by visual examination of the image under a 2 power magnification and separately by estimating cover for 25 subgrids within the 1 m^2 sampling frame (Pinder et al., 2004). Vegetation and litter cover were separately evaluated, and a measure of the total cover was estimated from 81 points dispersed throughout the plot (Pinder et al., 2004). Images for ground cover analyses were obtained before seasonal growth (early spring), at mid growth (summer), and at the end of the growing season (fall).

2.3. Measurement methods for horizontal dust flux

Wind erosion rates were evaluated through measurement of horizontal dust flux (HDF), which is a direct measure of transported soils (Breshears et al., 2003; Stout and Zobeck, 1996) and is significantly correlated with the vertical flux of suspended soil material associated with erosion (Gillette et al., 1997a; Whicker et al., 2006a, b). The HDF was measured with Big Spring Number Eight (BSNE) dust samplers (Fryrear, 1986; Zobeck et al., 2003). These samplers have a tail-fin and self-orient a small opening (10 cm^2) into the wind through which wind-driven dust particles enter. Once inside the sampler, the dust particles decelerate and deposit into a collection pan. The BSNE samplers have been shown to have good sampling efficiency for saltating particles, e.g., sizes $>50\,\mu m$, which make up a large fraction of the soil in the study area (Nyhan et al., 1978) and have been used extensively for field measurement of HDF (Fryrear, 1986; Goossens and Offer, 2000; Shao et al., 1993; Zobeck et al., 2003). For the purposes of this study, "dust" is defined as all of the airborne particulates collected by the sampler without regard to the composition (e.g., mineral or organic) or particle size considerations (Hinds, 1982; Pye, 1987), but others have used the term "dust" more narrowly to refer to the smaller fine fraction of soil suspended particles and exclude larger silt and sand particles (Goossens and Riksen, 2004).

To assess horizontal flux with height above the surface, BSNE samplers were positioned at sampling heights of 25, 50, and 100 cm. The collected dust was removed from the sampler by washing with distilled water, transferred into glass vials, dried at 50 °C to a constant mass, and measured to the nearest mg. The HDF was calculated by dividing the mass of the dust collected by the area of the sampler opening (10 cm^2) and the duration of the sampling period. The units of HDF were $g\,m^{-2}\,d^{-1}$. Thus, HDF represents the wind-driven mass flux of dust flowing horizontally along the earth surface at each sampling height. Lack of time-resolved measurements of particle size and wind velocity precluded HDF corrections for particle collection efficiency, so a collection efficiency of 100% was assumed potentially biasing the HDF measurements low. Dust was collected for sampling intervals that ranged from weekly during the windy spring months of March through May up to several weeks during less windy periods.

2.4. Statistical analyses

Statistical comparisons of HDF among burned and unburned and thinned and unthinned areas were performed using a mixed-model Analysis of Variance (ANOVA) where thinning, burning, and their interactions were fixed-effects and sampling periods, sampling locations, and BSNE positions within locations were random-effects (Milliken and Johnson, 1984). Statistical computations were performed using the Type 3 option of PROC MIXED of the SAS System (Littell et al., 1996). Hypotheses of specific effects, such as (1) the effects of thinning and (2) changes in the effects of thinning among years, were tested using F ratios computed for linear contrasts of means (LCM) (Littell et al., 1996). Due to the differential effects of precipitation and saltating mass flux with sampling height (Whicker et al., 2006b), separate analyses were performed for the 0.25, 0.5 and 1.0 m elevations. Because rainsplash of soil particles can be significant up to more than 0.5 m (Finkle, 1986) and contribute to the materials collected by the samplers (Whicker et al., 2006b), data analyses were performed for all sampling periods and separately for sampling periods with no measurable precipitation, which were termed dry periods. Dry periods were defined based on meteorological measurements made at a

LANL weather station (Rishel et al., 2003) located within a few km of all the study plots. Similar statistical procedures were used to test for differences in tree densities, canopy cover, and ground cover among forest sites. Results from statistical tests are reported for the *F*-ratio (F), degrees of freedom (df), and the level of significance (*P*).

3. Results

3.1. Impact of thinning on tree densities, tree heights and canopy cover

Thinned and unthinned tree densities show significant variability in tree densities among and within tree stands (Table 1). In areas that were thinned, the decrease in tree densities ranged from 32% to 86% (Table 1). Approximately 35% of the trees on the burn plots before the thinning had died from either the direct effects of the fire or the additional subsequent effects of drought and bark beetle infestations. Trees that apparently succumbed to these subsequent effects were included in the data in Table 1 to isolate the effects of thinning on tree density. Although thinning in the burned areas reduced tree densities, tree densities on thinned sites in the burn areas were similar to those for the burned + unthinned sites. This lack of pronounced difference reflects significant variation among sites in tree densities, and variation among sites in tree mortalities due to fire severity (Pinder et al., 2004).

Spherical densitometer measurements of tree canopy cover (Table 1) largely reflected the patterns of tree densities. Individual point measurements of canopy cover ranged from 0% to 96%, and means for forest sites ranged from 21.7% to 77.9%. Individual point measurements of 0% only occurred on burned and/or thinned sites. Measurements within sites were often variable as indicated by the large ratios of standard deviations to means which were usually >0.5. The similarities among treatment sites in tree densities and the variability in

Table 1
Results from measurements of tree densities and canopy cover in the various burn types

Burn type	Plot type	*N* plots	Unthinned	Post-thin	Unthinned minimum	Thined minimum	Unthinned maximum	Thinned maximum
			Mean ± Std	Mean ± Std				
Tree densities (trees per ha)								
Unburned-unthinned	Primary	5	1280 ± 455		700		1900	
	Secondary	4	50 ± 58		0		100	
Unburned-thinned	Primary	10	410 ± 623	110 ± 88	0	0	2100	300
	Secondary	7	1400 ± 719	200 ± 129	200	0	2300	400
Burned-unthinned	Primary	5	400 ± 255		200		800	
	Secondary	4	350 ± 129		200		500	
Burned-thinned	Primary	10	980 ± 394	410 ± 288	400	0	1600	900
	Secondary	7	757 ± 435	243 ± 140	100	100	1400	500
Canopy cover (%)								
Unburned-unthinned	Primary	15	77.9 ± 14.1		56		96	
	Secondary	12	46.0 ± 28.4		8		96	
Unburned-thinned	Primary	22		39.5 ± 21.6		0		76
	Secondary	14		33.1 ± 19.2		0		72
Burned-unthinned	Primary	15	36.2 ± 29.6		0		88	
	Secondary	12	21.7 ± 17.1		0		48	
Burned-thinned	Primary	22		21.2 ± 17.6		0		56
	Secondary	14		29.1 ± 15.4		0		56

individual densitometer measurements within treatment sites complicated the comparison among burn and thinning treatments. Although there were significant differences in canopy cover measurements among sites within combinations of burning and thinning ($F = 5.17$; df $= 4$, 116; $P < 0.01$), there were no statistically significant effects ($P \geqslant 0.05$) of burning, thinning, or the interaction of burning and thinning.

Besides reducing densities, thinning altered the distributions of tree heights. Because smaller trees were removed, the height distributions on thinned areas were skewed more toward taller trees in both burned and unburned areas. Few trees $\leqslant$8-m tall were present on the thinned sites, but $>20\%$ of the trees on the unthinned sites were $\leqslant$8-m tall.

3.2. Impact of thinning on forest ground cover

Thinning operations reduced the litter cover and exposed approximately 20% bare soil in both unburned (LCM $F = 21.1$; df $= 1$, 83; $P < 0.01$) and burned (LCM $F = 10.3$; df $= 1$, 83; $P < 0.01$) areas that were thinned (Table 2). Although the litter cover in the burned areas was less than that in the unburned and was composed of a thinner layer of recently deposited pine needles (Pinder et al., 2004), there was no significant difference in the level of exposed soils between thinned sites in unburned and burned areas (LCM $F = 0.3$; df $= 1$, 83; $P \geqslant 0.10$). There is a potentially important difference between the effects of thinning and fire on the percent reduction in litter cover. Whereas fire generally removed the litter cover until scorched needles reestablished a relatively thin layer of litter over the next months, thinning exposed patches of bare soil due more to the redistribution of litter more than its removal. Also, thinning may have increased the mass of litter through the addition of wood chips and slash.

We found no measurable effect of thinning on vegetation cover possibly because of large variations in vegetation cover among sites within the same thin/burn category as discussed in Pinder et al. (2004). The results for total cover (Table 2), which includes both litter and vegetation, were similar to those for litter cover alone with significant decreases in total cover for thinned plots on (1) unburned (LCM $F = 12.7$; df $= 1$, 83; $P < 0.01$) and (2) burn areas (LCM $F = 16.8$; df $= 1$, 83; $P < 0.01$). These results indicate that any responses of vegetation cover were not sufficient to moderate the reductions in litter cover. Although an increase in total cover during the second year would suggest recovery, there was no measurable change in total cover during the second year on the primary unburned + thinned site (LCM $F = 1.59$; df $= 1$, 83; $P \geqslant 0.10$).

3.3. Characterization of soils in study sites

The textures of undispersed surface soil in the plots were characterized as silt loam soils with mean and one standard deviation values of $40 \pm 8\%$ sand, $47 \pm 7\%$ silt and $12 \pm 3\%$ clay and were not statistically different among the thinned and unthinned plots. Therefore, any difference in soil erosion between thinned and unthinned sites is not likely to be explained by differences in soil texture.

Table 2
Litter cover, vegetation cover and total cover (%) on the ground surface in thinned and unthinned sites in unburned and moderate burn areas of Ponderosa pine forest

Ground cover type	Unburned				Moderate burned			
	Unthinned		Thinned		Unthinned		Thinned	
	n	Mean ± 1 Std	n	Mean ± 1 Std	n	Mean ± 1 Std	n	Mean ± 1 Std
Litter	34	98.3 ± 3.6	17	77.1 ± 15.5	16	79.9 ± 18.0	17	61.6 ± 23.7
Vegetation	34	5.3 ± 5.8	17	11.5 ± 11.9	16	12.3 ± 11.4	17	6.3 ± 10.4
Total	34	98.2 ± 3.9	17	85.6 ± 11.0	16	82.4 ± 13.7	17	64.4 ± 19.2

Data are means of measurements of 9 or 10 samples at primary sites and 7 samples at secondary sites. Samples collected during 2002 for the primary thinned unburned site and corresponding unthinned site and during 2003 for the secondary thinned unburned site, the primary thinned moderate burn site, the secondary thinned moderate burn site and corresponding unthinned sites.

3.4. Effects of thinning in burned and unburned forest on HDF

3.4.1. HDF analysis for all sampling periods

Generally, thinning and burning increased HDF at each of the sampling heights (Fig. 2, Table 3). Further, the impact of thinning on HDF generally decreased with elevation above the ground surface. At 0.25 m, thinning increased HDF for the unburned (LCM on logarithmic transformed data $F = 25.52$; df $= 1, 17$; $P<0.001$) and the burned areas (LCM on logarithmic transformed data $F = 14.78$; df $= 1, 17$; $P<0.001$) with no significant difference in the degree of effect between unburned and burned areas (LCM on logarithmic transformed data $F = 0.16$; df $= 1, 17$; $P>0.10$). The results at 0.50 m were similar to those at 0.25 m with statistically significant thinning effects for unburned (LCM on logarithmic transformed data $F = 6.74$; df $= 1, 17$; $P<0.001$) and burned (LCM on logarithmic transformed data $F = 3.65$; df $= 1, 17$; $P<0.001$) sites with no difference in degree of effect between unburned and burned areas (LCM on logarithmic transformed data $F = 2.80$; df $= 1, 17$; $P\geqslant 0.05$). At 1.00 m there was an increase in HDF on thinned unburned areas (LCM on logarithmic transformed data $F = 7.40$; df $= 1, 17$; $P<0.01$) but not on burned areas (LCM on logarithmic transformed data $F = 0.0$; df $= 1, 17$; $P\geqslant 0.10$).

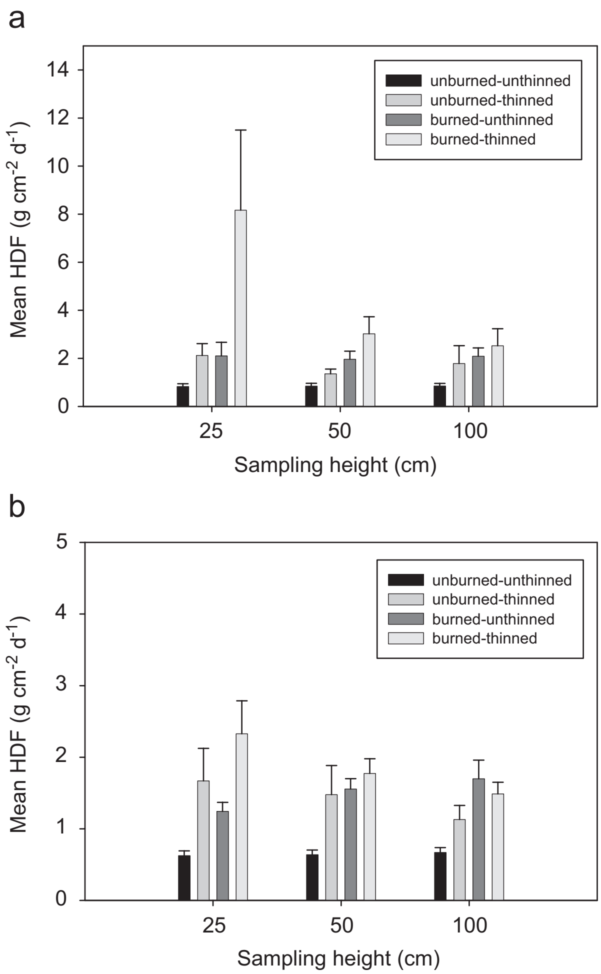

Fig. 2. Horizontal mass flux categorized by thinning/burn category for each sampling height for all sampling periods (a) and for dry sampling periods (b). Note the difference in scale on the *y*-axis between graphs (a) and (b). Error bars represent the standard error of the mean.

Table 3
Summary statistics for measured horizontal mass flux (units of $g\,m^{-2}\,d^{-1}$) within thinning and burned treatments

	Unburned						Burned					
	Primary			Secondary			Primary			Secondary		
	25 cm	50 cm	100 cm	25 cm	50 cm	100 cm	25 cm	50 cm	100 cm	25 cm	50 cm	100 cm
Unthinned												
Median	0.50	0.55	0.50	0.86	0.94	0.94						
	(0.43)	(0.50)	(0.50)	(1.00)	(1.00)	(1.08)						
Minimum	0.14	0.29	0.14	0.29	0.33	0.33						
	(0.14)	(0.28)	(0.14)	(0.50)	(0.57)	(0.71)						
Maximum	2.15	1.91	1.63	1.46	1.46	1.62						
	(0.67)	(0.69)	(0.92)	(1.46)	(1.46)	(1.63)						
Collection period: 3/26/02 *to* 7/5/02												
Median	0.57	0.62	0.71	0.83	0.89	0.94	1.46	1.87	2.13	1.10	1.14	1.29
	(0.47)	(0.44)	(0.50)	(0.73)	(0.79)	(0.73)	(1.31)	(1.69)	(1.88)	(0.92)	(1.00)	(1.11)
Minimum	0.11	0.14	0.08	0.17	0.25	0.25	0.08	0.17	0.71	0.42	0.43	0.57
	(0.11)	(0.29)	(0.14)	(0.29)	(0.29)	(0.43)	(0.71)	(1.00)	(0.71)	(0.43)	(0.57)	(0.57)
Maximum	3.89	2.89	4.33	4.33	3.42	2.89	10.44	7.50	6.67	5.22	3.50	4.17
	(0.83)	(0.71)	(0.83)	(1.29)	(1.00)	(1.14)	(3.00)	(3.67)	(4.56)	(1.29)	(2.00)	(1.86)
Collection period: 3/26/03 *to* 7/5/03							*Collection period*: 3/26/03 *to* 7/5/03					
Thinned												
Median	1.33	1.33	0.94									
	(1.30)	(1.30)	(0.96)									
Minimum	0.50	0.57	0.06									
	(0.50)	(0.57)	(0.06)									
Maximum	4.38	2.14	1.76									
	(1.93)	(2.14)	(1.69)									
Collection period: 3/26/02 *to* 7/5/02 (*year* 1 *post-thin*)												
Median	0.96	1.00	1.00	1.29	1.07	0.85	3.23	2.29	1.77	2.08	1.78	1.75
	(0.75)	(0.86)	(1.00)	(0.88)	(0.87)	(0.76)	(2.18)	(2.00)	(1.49)	(1.67)	(1.50)	(1.52)
Minimum	0.17	0.14	0.17	0.43	0.29	0.29	0.57	0.42	0.42	0.75	0.57	0.57
	(0.43)	(0.14)	(0.29)	(0.29)	(0.29)	(0.29)	(0.71)	(0.57)	(0.57)	(0.75)	(0.57)	(0.57)
Maximum	4.22	3.83	111.80	60.00	6.33	6.78	76.89	15.22	35.00	25.22	7.00	4.33
	(1.71)	(1.33)	(1.33)	(2.29)	(1.17)	(1.00)	(6.71)	(3.00)	(2.57)	(2.67)	(2.71)	(2.00)
Collection period: 3/26/03 *to* 7/5/03 (*year* 2 *post-thin*)							*Collection period*: 3/26/03 *to* 7/5/03 (*year* 1 *post-thin*)					

This table provides summary statistics for all sampling periods and for dry periods (data in parentheses) and is divided into collection periods, which are specified below each data set.

3.4.2. HDF analyses for dry periods

Similar results were obtained using only the data from dry sampling periods (Fig. 2b) as those found using data from all sampling periods (Fig. 2a). The descriptive statistics of HDF for dry sampling periods are presented in Table 3. We found during dry periods that there were significant thinning effects at 0.25 m in burned and unburned plots (LCM on logarithmic transformed data $F = 24.86$; df $= 1, 17$; $P \leqslant 0.001$) and no thinning + burn interactions (LCM on logarithmic transformed data $F = 1.00$; df $= 1, 17$; $P > 0.1$). At 0.5 m, we found significant overall thinning effects, but this difference was found only on the unburned plots (LCM on logarithmic transformed data $F = 3.98$; df $= 1, 17$; $P \leqslant 0.01$). There was also an interaction between the thinning and the fire effect (LCM on logarithmic transformed data $F = 5.00$; df $= 1, 17$; $P \leqslant 0.05$) at the 0.5 m sampling height. Finally, the results at 1 m showed no significant thinning effects or interactions.

3.5. HDF over time in thinned areas

For both the wet and dry periods, we found a decline in HDF at 0.25 m on the primary unburned, thinned site between 2002 and 2003 (LCM on logarithmic transformed data $F = 5.82$; df $= 1, 17$; $P < 0.05$) where the mean HDF declined by 40% from 2002 to 2003 (Fig. 3). There was also a decline in HDF from 2002 to 2003 on the primary thinned, unburned site at 0.50 m (LCM on logarithmic transformed data $F = 12.26$; df $= 1, 17$; $P(<0.001)$ similar to that observed at 0.25 m. There was no statistically significant decline in HDF between years 2002 and 2003 on the primary thinned unburned site at 1 m (LCM on logarithmic transformed data $F = 0.84$; df $= 1, 17$; $P \geqslant 0.10$), potentially because of the lower HDF generally found at this sampling height. Similar results were obtained for dry periods, but their statistical significance was limited by fewer samples. Although this analysis was done using a two-year data set and is limited to two sampling sites (i.e., the primary unburned/unthinned and primary unburned/thinned sites), the combined analysis for all sampling periods and the subset of data for just dry sampling periods, is suggestive of soil stabilization. The decline in HDF was found though the meteorological conditions were similar between 2002 and 2003. Mean daily wind velocities at 12 m above the ground were 2.67 ± 0.98 and $2.86 \pm 1.02\,\mathrm{m\,s^{-1}}$ for 2002 and 2003, respectively. Similarly, the means of the daily gust velocity were $11.69 \pm 4.29\,\mathrm{m\,s^{-1}}$ in 2002 and $11.17 \pm 4.0\,\mathrm{m\,s^{-1}}$ in 2003. Precipitation was slightly less during 2003 (25.25 cm) compared to 2002 (29.74 cm).

3.6. Relationship between HDF and vegetation cover

Combining data on HDF and cover from this study and a companion study on the effects of wildfire severity on dust flux (Whicker et al., 2006b), a relationship between HDF and vegetation cover emerges. HDF at 1-m elevation decreased significantly as the amount of ground and tree cover increased (Fig. 4). These linear relationships, while significant, should not be extrapolated beyond the amount of cover measured in the study because others have shown erosion rates to have thresholds and to be non-linearly related to cover, especially at the lowest levels of ground cover (Bagnold, 1941; Fryrear, 1985; Davenport et al., 1998).

4. Discussion and conclusions

Our results quantify differential effects of tree thinning on rates of wind erosion as measured by HDF. Sites that were recently thinned or burned, which consequently had reduced ground cover, had higher rates of wind erosion. This effect was particularly significant at the lower sampling heights, suggesting increased local soil redistribution on a scale of several meters (Stout and Zobeck, 1996). However, because HDF has been shown to be related to vertical dust emissions in forests (Whicker et al., 2006a, b), these results add further support to previous studies showing that wind erosion rates can be substantially increased in disturbed semiarid ecosystems (see also Breshears et al., 2007). Notably, rates of HDF in the first year or so following forest thinning were as great as those in nearby sites that had been recently burned. The increase in HDF in locations that were both burned and thinned is roughly equally to the sum of increases in rates for sites that were burned and for sites that were thinned. This suggests that burning and thinning have primarily an additive rather than a multiplicative effect on wind erosion. The increase in HDF appears to relate to the amount of ground cover at a site, which for our site is also similar to the relationship with tree canopy cover. In the second year following thinning (2003), the results suggested a decrease in rates of wind erosion in thinned sites at the lower two sampling heights, which provided evidence of soil stabilization. Though additional data are needed to better quantify this stabilization, these results in thinned areas may be in contrast to the severely burned forest areas where HDF had not decreased in three years after the fire (Whicker et al., 2006a, b).

Regarding stabilization with time, HDF could be expected to be highest immediately after the forest disturbance before vegetation and litter recovery occurs. Also, Ravi et al. (2006) showed that wind erosion rates were increased following a fire due to fire induced water repellency, which could have increased HDF most during the first months following the fire. We were not able to start measurements in the burned areas until ~ 1 year following the fire and may have missed the biggest increases in wind erosion in the burned areas (Whicker et al., 2006a, b). Thus, though we found significant increases in both burned and thinned sites, our measurements may be underestimating HDF for the first few months in the burned areas.

The quantification and comparison of the HDF among the undisturbed and disturbed forest sites were the main purposes of the study. Using these values, it is valuable to put measured dust transport rates in forests in context relative to other ecosystems. For this, the total amount of dust being transported is generally determined by integration of the HDF up to a height of 1 m, which is the sampling height that contains >99%

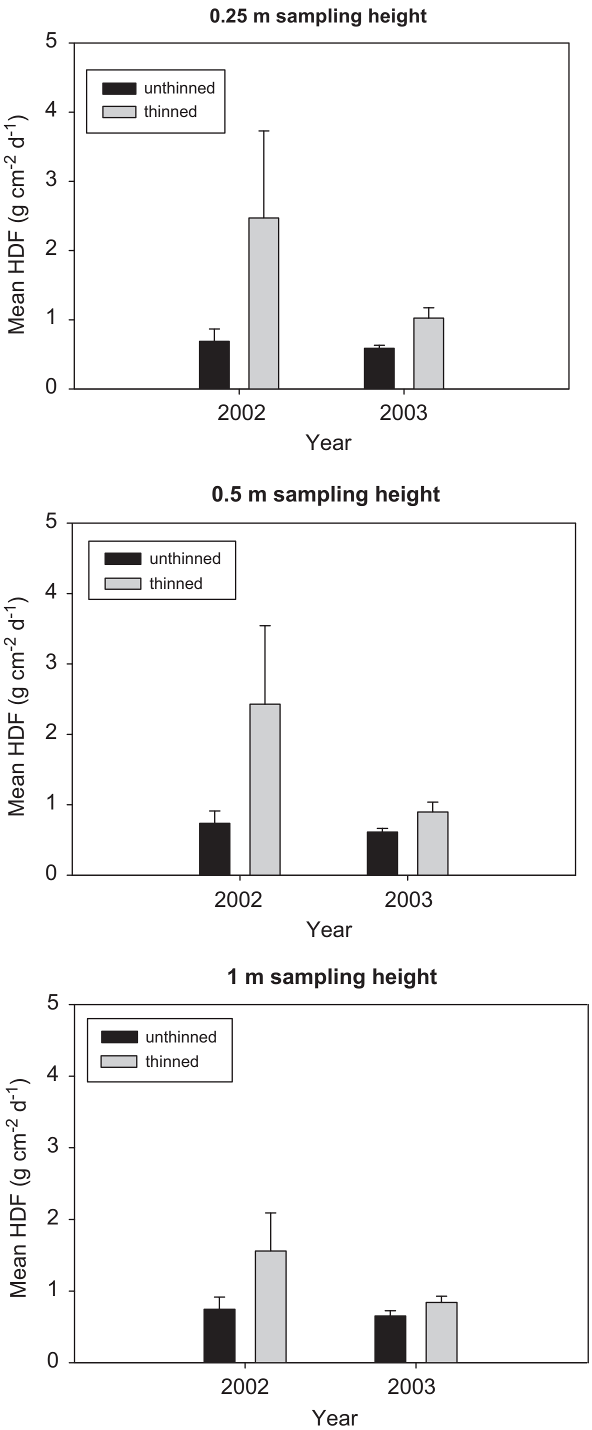

Fig. 3. Mean horizontal dust flux in each year following thinning in the unburned forest showing potential recovery. Error bars represent the standard error of the mean.

of the flux (Stout and Zobeck, 1996). The model developed by Shao and Raupach (1992) was used to describe the relationship between mean horizontal dust flux ($\overline{\mathrm{HDF}}$) with height:

$$\overline{\mathrm{HDF}}(z) = a\mathrm{e}^{-[bz+cz^2]}, \tag{1}$$

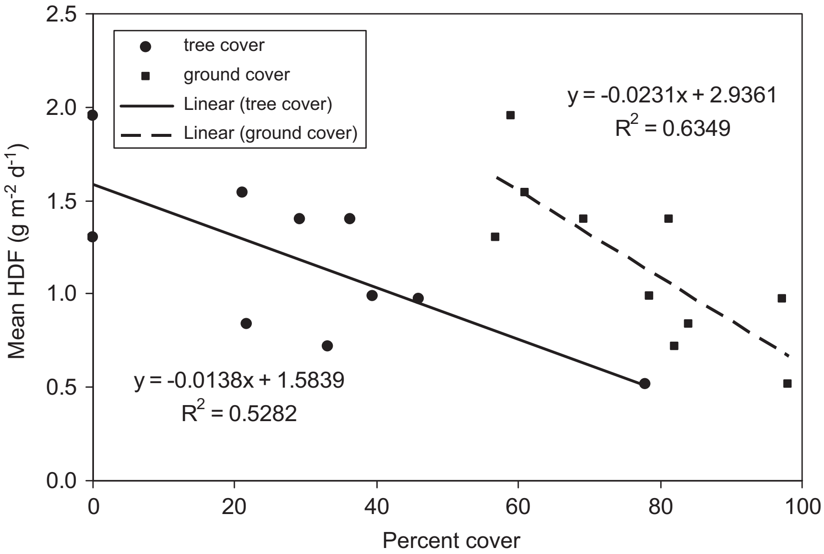

Fig. 4. Mean horizontal dust flux at 1 m as a function of the percent cover of live tree and of ground cover.

where z is height above the ground, and a, b, and c are fitting parameters. This equation was then integrated to get an estimate of the integrated flux:

$$Q = \int_0^{1m} \overline{\mathrm{HDF}}(z)\,\mathrm{d}z. \tag{2}$$

The integration of Eq. (1) was performed for both unburned and burned sites that were thinned (e.g., the most disturbed) and resulted in height-integrated fluxes of 1.43 and 1.86 $\mathrm{g\,m^{-1}\,d^{-1}}$, respectively. For comparison, fluxes in wind-dominated ecosystems such as semiarid shrubland sites or eroding playas are several orders-of-magnitude higher (Gillette et al., 1997b; Gillette and Pitchford, 2004) and the HDF in agricultural fields are much higher than measured in this study (Stout and Zobeck, 1996). The lower dust transport in the disturbed and undisturbed forests in this study could be the result of residual woody and ground cover that would absorb much of the wind momentum (Breshears et al., 2007). However, wind erosion is expected to be more substantial in drastic cases where tree thinning operations remove large amounts tree and ground cover (e.g., clear cutting).

4.1. Considerations for forest ecology

Although our dust-flux measurements were usually $<2\,\mathrm{g\,m^{-2}\,d^{-1}}$, their impacts on soil properties must be based on both the mass and the composition of the dust. Wind erosion affects soil quality by removing and redistributing smaller particles contained in top soil. Soil nutrients and organic matter are often associated with these smaller particles, and wind erosion has been found to preferentially remove the finer fraction of soil that is enriched in materials such as nitrogen, phosphorus, and organic material, all of which can degrade soil quality (Fisher and Binkley, 2000; Goossens, 2004; Larney et al., 1998; Sterk et al., 1996, 2001). Conversely, deposition of aeolian dust into disturbed areas that have been remediated to grassland or tree-covered patches has been shown to increase deposition rates of dust (McGowan and Ledgard, 2005) and over time improve soil quality (Reeder et al., 1998; Shirato et al., 2004). Thus, it seems imprudent to merely dismiss the small fluxes as unimportant. Rather, maintaining small dust fluxes should be considered a metric for maintaining forest quality.

The following tactics could be used to prevent soil and forest degradation following thinning. First, because of the importance of vegetation and litter cover toward reducing wind erosion found in this study (Fig. 4) and other studies (Fryrear, 1985), thinning strategies that leave litter (e.g., pine needles and slash) on the soil surface may significantly reduce wind erosion. This litter could also improve soil quality by conserving nutrients and soil organics (Fisher and Binkley, 2000). Second, the increased HDF found in the thinned areas may be partially explained by significant disruptions to the soil surface by the tracked vehicles (Belnap and Gillette, 1998; Grantham et al., 2001). Techniques that minimize soil disruption and breaking of soil crusts would help conserve soil in the affected areas. For example, removal of trees during times when the soil is frozen or snow covered may result in less disruption to the soil surface and reduce soil loss. Finally, wind erosion is just one impact of thinning that should also be considered. For example, soil compaction can severely degrade soil productivity, lead to increased water erosion, and the effects can last for many years (Selmants et al., 2003).

Questions also remain concerning the impacts of differing thinning technologies. Thinning procedures may vary from simple cutting of mostly small trees with chain saws and with removal of wastes by burning slash piles on up to the removal of large trees using heavy, tracked vehicles and skidding procedures, such as used at LANL. The spatial scale of the thinned areas can also vary in size and pattern. Where heavy equipment is broadly used for cost effective thinning of large areas, such as in our study plots, we expect wind erosion to be greater than at sites where less intrusive techniques were used and/or less area impacted.

Along with those of related studies at larger scales, our results highlight that disturbances to forests and other ecosystems that substantially reduce ground cover and disturb surface soils can increase dust emissions. This increase has the potential to significantly impact human and ecosystem health at spatial scales from local, as shown in this study, to global (Griffin et al., 2001; Whicker et al., 2006a).

In conclusion, our results do not indicate that increases in wind erosion due to thinning should preclude restoration, but we do argue that: (1) the increases are sufficient to warrant consideration and additional study, and (2) should be considered as an important component of restoration, salvage, and monitoring plans (Miller, 2004; Whitford et al., 1998).

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