Changes in Forest Species Composition and Structure After Stand-Replacing Wildfire in Mountains of Southeastern Arizona

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Abstract—*A wildfire in the Chiricahua Mountains of southeastern Arizona apparently altered the long-term structure of the forest. The pre-fire canopy forest, which had not burned for 100 years, was an even mixture of Arizona pines and Rocky Mountain Douglas-firs. A decade later the new forest was numerically dominated by quaking aspen seedlings in clumps separated by persistent gaps. Douglas-firs were positively associated with aspens but Arizona pines were not. The new forest was more open, diverse, and patchy than the one it replaced. Extensive stand-replacing fires in the Mountain West may produce forests with long-term desirable resource properties.*

Introduction

Very large, stand replacement forest fires are becoming increasingly common in the coniferous forests of the Western United States (Zimmerman 2003). Since such fires destroy all of the above ground portions of trees and other vegetation, the forest that regenerates may develop a different structure and species composition than that which existed before the fire. Seed banks and soil properties are examples of variables affected by intense fire that could subsequently alter the course of succession. This is a report on the first decade of regeneration of trees following such a wildfire that occurred in southeastern Arizona in 1994, apparently altering the long-term structure of the forest.

Study Area

The Chiricahua Mountains are located in extreme southeastern Arizona, near the borders of New Mexico and the Republic of Mexico. The range is approximately 65 km long and 32 km wide, with a maximum elevation of 2,975 m. The primary study area contained three species of trees, Arizona pine (*Pinus ponderosa* var. *arizonica*), Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), and quaking aspen (*Populus tremuloides*). The study area was located at elevations of 2,700 to 2,900 m, UTM grid coordinates 3528500N 662100E, immediately north of the Chiricahua Wilderness within the Coronado National Forest in Cochise County. Slope in the sampling area varied from 0–8%, with an easterly aspect.

Rattlesnake Fire

In June and July of 1994 a fire ignited by lightning burned approximately 11,000 ha of forested land in the upper elevations of the Chiricahua Mountains. This was the first large fire that had burned these mountains since the late 19th Century (Bahre 1991). The fire burned at varying intensities through almost all the coniferous and aspen forest of the mountain range. It was propelled by erratic winds, steep terrain, and it burned through fuel types ranging from Madrean oak woodlands to Englemann spruce. Within the perimeter of the fire almost all forested areas burned. In some watersheds all trees were killed and the soil was destroyed by heat so intense that boulders shattered and cylindrical holes more than a meter deep were left where trees were burned below the surface of the ground. In other areas, however, very light surface fires consumed only small-diameter fuels and lightly scorched the bases of trees.

Sampling Methods

In the summers of 1998 and 1999 perpendicular belt transects were established on a gently sloping plateau where most trees had been killed by the Rattlesnake Fire (figure 1). These sampling belts were 4 m wide, with lengths of 255 and 300 m, with a total sampling area of 2,204 m^2 that crossed an area of 7.75 ha. The location and height of all tree seedlings and resprouts within the belts were recorded annually between 1999 and 2003, using the center lines of the sampling belts to define orthogonal coordinates. In 1998 the composition of the pre-fire forest was measured from 21 points at 15 m intervals along a line defined by the center of the 300 m belt using the point-quarter method (Cottam and Curtis 1956). Tree densities were calculated using the mean distance from the sampling point to the tree as the mean distance between trees. At that time all snags of canopy trees present before the fire were still standing, and the species and dbh of both dead and living tree stems greater than 10 cm in diameter were sampled.

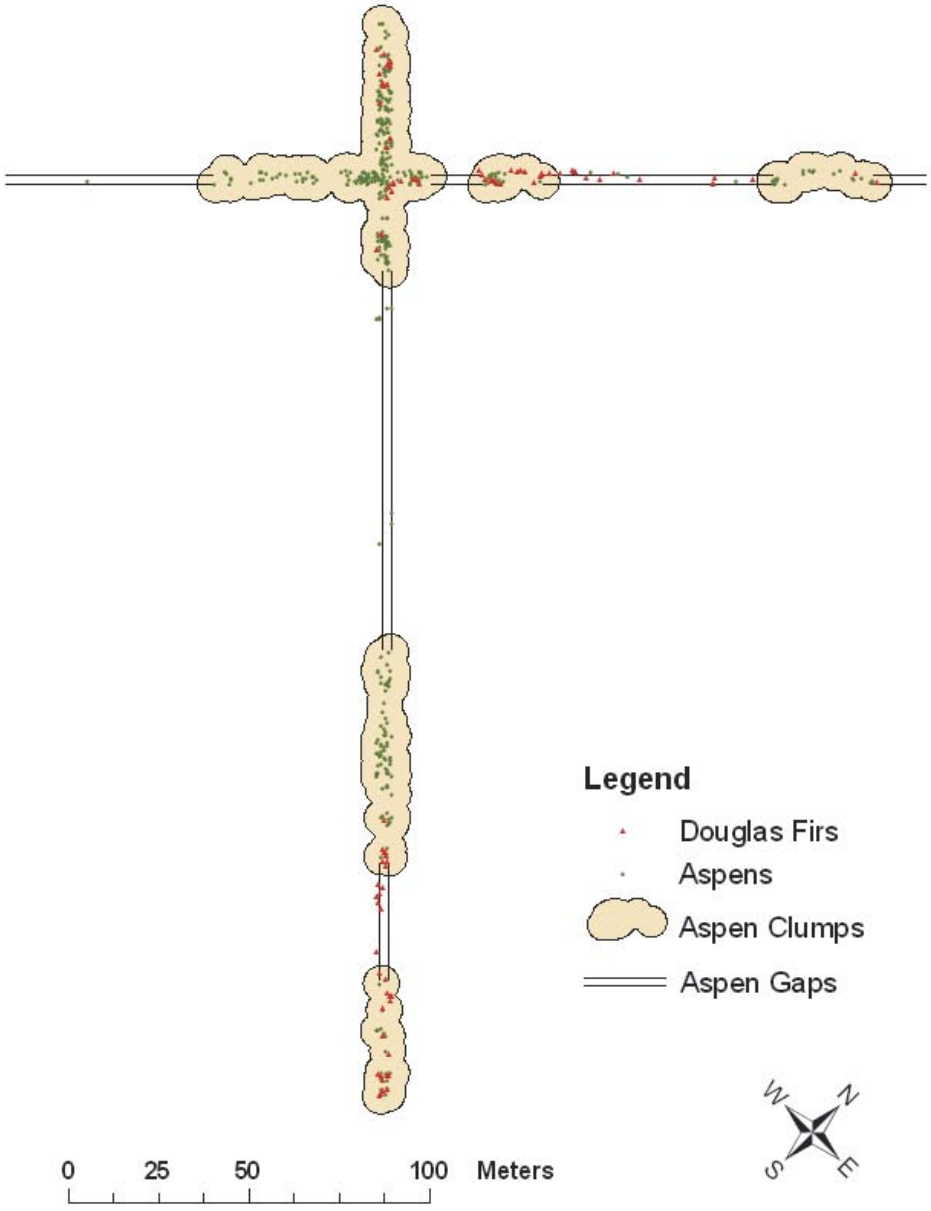

Figure 1—Aspen clumps and gaps in 2003, and distribution of Rocky Mountain Douglas-fir.

Results and Discussion

Pre-fire Forest Composition

Prior to the fire the portion of the study area along the 300 m transect contained nearly equal numbers of similarly sized Rocky Mountain Douglas-fir (DF) and Arizona pine (AP) in a combined density of 701 trees per ha, and combined basal area of 57.25 m^2 per ha. No other species of trees were found. DF had a mean dbh of 31 cm (SD = 15.5) and AP had a mean dbh of 27 cm (SD = 12.3). The two species of conifers were evenly mixed, with relative frequencies of 49% (DF) and 51% (AP). The estimated height of pre-fire canopy trees was 16–18 m, based on measurements in 1998 of trees in the study area that survived the fire. Prior to the fire the study area had a tree density typical of Southwestern forests that have been free of fire for most of the 20th century (Dahms and Geils 1997; Friederici 2003). Twenty-six percent of the AP sampled survived the fire, all in an area where the fire did burn the tree crowns. The fire killed all sampled DF, although there were some that survived nearby. There was no evidence of quaking aspen canopy trees found anywhere on or near the sampling belts or within the surrounding area of 7.75 ha, although snags or living aspens existed within 100 m of the ends of the belts. Canopy cover in places with living AP was 71%.

The Regenerating Forest

Aspens numerically dominated the post-fire forest community, accounting for four-fifths of all tree seedlings. In 1999 an analysis of the root systems of 27 randomly selected aspens excavated 1–10 m from the edges of the belt transects revealed that all these plants were seedlings, not suckers arising from established aspen roots (Quinn and Wu 2001). Aspen snags were present near the study area, but none were close enough to the area sampled to support aspen root systems within the study area.

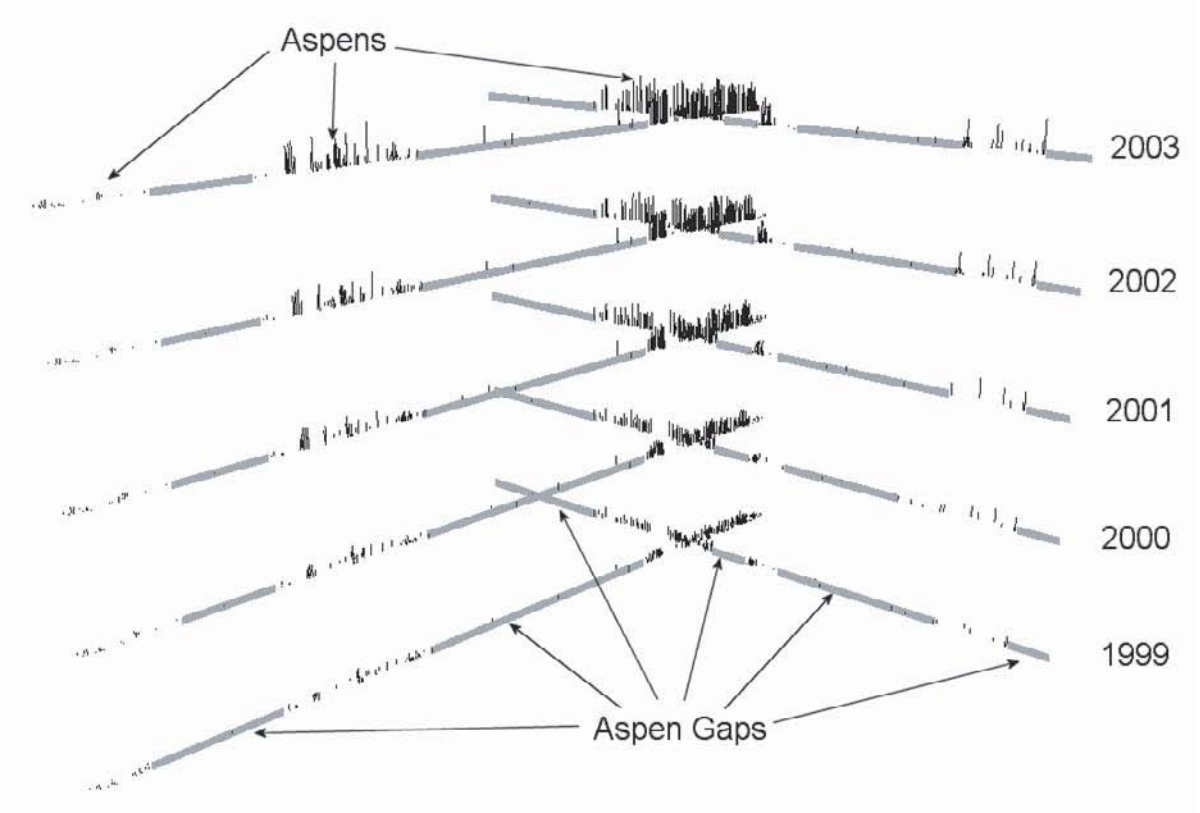

Figure 2—Distribution of aspen gaps from 1999 to 2003. Lengths of vertical lines are proportional to heights of aspen trees.

GIS mapping of tree seedlings revealed that aspens occurred in clumps separated by pronounced gaps, and this pattern of gaps persisted through the period of the study (figures 1, 2). The gaps and clumps were defined by using the buffer function of GIS. A buffer with a radius of 5 meters was generated around each of the live aspen trees. If the buffers overlapped, the boundaries between them were eliminated to create a single polygon. These polygons were defined as clumps if they contained 5 or more aspens. The area between two clumps along the belt transects was defined as a gap. The gap length was measured from the last tree in one clump to the first tree in the next clump or to the end of the sample belt. Based on this method, all gaps were at least 10 meters long and contained no clusters of 5 or more trees. Forty-nine to 55% of the sampling belts were gaps that persisted for the duration of the study (figure 2).

Rocky Mountain Douglas-fir and Arizona pines, the two tree species of conifers present before the fire, returned to the study area as seedlings. Both species grew at much lower densities than aspens, and their numbers within aspen clumps and in the overall study area increased slowly and unevenly with time (table 1). Unlike the aspens, which have light seeds that can be dispersed over long distances by the wind, the relatively heavy seeds of DF and AP can only be passively dispersed over distances of approximately 40 m (AP) to a few hundred meters (DF) as they fall from seed trees (Howard 2003; Steinberg 2002). Consequently, the seedlings of both species tended to be situated near areas where canopy seed trees survived the fire.

The DF seedlings are concentrated in and around the aspen clumps (figure 1). Following the method of McAuliffe (1988) the association between DF and aspens was tested by generating a number of randomly placed points within the belts equal to the number of DF seedlings, measuring the distance from each point to the nearest aspen, and then comparing those values to the actual distances between all DF seedlings present in 2003 and the nearest aspen. The mean distance between DF and nearest aspen was 2.14 m (SD = 2.87, $n = 89$), while the comparable value for the random array was 3.92 m (SD = 4.61), which is significantly greater ($t = 3.09$, $P = 0.002$). The association between DF and aspens may be explained by the presence of shade provided by the rapidly growing QA, over half of which by 2003 were more than a meter tall (figure 2). Those DF that grew outside of the aspen clumps were located beneath or near canopy trees that survived the fire. Since DF germinates and grows best in shade (Steinberg 2002), the aspens and surviving canopy trees provided a framework of shade for the establishment of DF seedlings.

The number of AP was relatively small in the areas sampled (table 1), and unlike DF they were not concentrated in or around the aspen clumps. Applying the same test of association between AP and aspen as compared with AP and random points showed a lack of a significant difference between the means at the 5% level of significance ($t = 1.54$, $P = 0.129$, $n = 34$).

If the easily dispersed seeds of aspen trees had been randomly scattered across the study area since the fire it remains to be explained why seedlings were not more evenly distributed. Aspen tree gaps along the belt transects constituted 52% of

Table 1—Tree density by species 1999 - 2003 (trees per ha).

	Aspen		Arizona pine		Douglas-fir	
	In clumps	Study area	In clumps	Study area	In clumps	Study area
1999	3,516	1,620	61	45	447	277
2000	3,071	1,624	170	118	518	331
2001	3,349	1,633	261	177	714	445
2002	3,745	1,856	171	118	599	377
2003	4,078	2,010	209	154	675	408

the area sampled in 2003 (figures 1, 2). A likely explanation, which may apply to the distribution of DF as well, is the uneven amount of available soil moisture during the summer growing season. The monsoon rains of July and August are irregular in onset and quantity. The first half of the growing season, May and June, has relatively little to no rainfall (table 2). The soil is quite thin over the study area, as indicated by exposed rhyolite bedrock in many places. Limited summer precipitation and thin soil could easily create an irregular pattern of relatively xeric and mesic microsites, and aspens and DF are intolerant of dry sites (Anderson 2001; Steinberg 2002).

One vegetation pattern suggests that the distribution of aspen-DF tree clumps may be governed by sufficient soil moisture. Scouler willow (*Salix scouleriana*) is a large shrub that grows well after stand replacement fires. This species requires moderately moist sites as compared to typical upland forest sites of the Southwest (Anderson 2001). The willows were abundant in patches across the sample area, and these willow patches fell almost entirely within the aspen/DF clumps. The test of association comparing distances between willows and aspens, and between willows and random points showed that willows are strongly associated with aspens ($t = 1.99$, $P < 0.001$, $n = 73$). Its presence within these clumps suggests that these are locations where moderate soil moisture has been reliably available, satisfying the requirements of both aspens and willows.

The forest that was regenerating in the study area was markedly different than that which grew there prior to the fire. It was dominated numerically by aspens (table1), which colonized the area as seedlings, while DF and AP were less numerous. The aspens occurred in clumps separated by gaps containing relatively few tree seedlings of any species (figures 1, 2). DF became established within and around most of these clumps, while AP lacked association with the clumps. In 2003 AP was too uncommon to break up the gaps by colonization, probably because of the inability of its seeds to disperse over long distances from scattered seed trees. Over the study period the distribution pattern of all 3 tree species remained stable while tree densities increased slowly (table 1). The stand replacement fire created the beginnings of a forest that was more open, diverse, and patchy than the one it replaced. The Southwestern ponderosa pine forests of the 19th century were also broken by frequent openings (Mast 2003), so in this respect the new forest may become structurally similar to forests of the past.

Over a time span of a century or more the relative abundance and spatial arrangement of the 3 tree species present will undoubtedly change as aspens thin and die in undisturbed places and are replaced by DF. AP may eventually close existing forest gaps as seeds incrementally reach suitable sites, especially during years with abundant seed crops. At the same time scale future fires of lower intensity and lesser extent, in this region of frequent summer lightning, may create new gaps and opportunities for regeneration. These events could maintain the forest as a heterogeneous mixture of vegetation types and structures in response to a more complex, fine-grained disturbance regime.

Table 2—Mean monthly precipitation at a weather station 4 km from study site (N = 39).

Month	May	June	July	August
Mean	12.7	25.4	111.2	94.7
S.D.	23.4	33.5	45.7	43.4

At the study site dry pine species of lower elevations mingle with firs and aspens of higher elevations, so that individual fire responses of three species of trees can create a shifting mosaic of forest types that depends on variations in the disturbance regime and microsite characteristics from point to point. Mixtures of tree species such as these with different responses to fire and microclimate are resilient to localized disruptions because collectively they possess several options of physiology and life history that are potentially available for regeneration. A particular species may temporarily become more or less abundant with each revolution of the fire cycle, but it is unlikely that any species will be eliminated by fire alone across its natural range on an entire landscape.

The Rattlesnake Fire, which burned most of the coniferous forest over an entire mountain range, may have set in motion events that will establish a forest that exhibits desirable properties not present before the fire, such as better overall wildlife habitat, greater species richness of plants, and a lessened propensity for extensive and uncontrollable wildfires. Under these conditions, where fuel accumulations are maintained at low to moderate levels by frequent ground fires, it may be safe and desirable to allow some fires of moderate intensity and size to burn without human intervention. Admittedly this is a more plausible scenario for a wilderness area on public land, with no structures or permanent residents, than it would be in populated areas with multiple land uses and goals. Nonetheless there are extensive parts of the Mountain West with remote forests that may fit this model once they have experienced large stand replacing fires. The extensive forest fires that have become so common in the Mountain West may not always mark an unfortunate sea change in ecological processes and management goals, producing nothing but long term resource losses.

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