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Interactions of fuel treatments, wildfire severity, and carbon dynamics in dry conifer forests

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ABSTRACT

Wildfires have been increasing in size and severity over recent decades. Forest managers use fuel treatments, including tree thinning and prescribed burning, to reduce the risk of high-severity fire. The impact of fuel treatments on carbon dynamics is not fully understood; previous research indicates that because carbon is removed during fuel treatments, the net effect may not be a reduction of carbon lost in the case of wildfire. The Rodeo–Chediski Fire, which burned in Arizona in 2002, was one of the largest and most severe wildfires recorded in the southwestern United States. Our objectives were to quantify carbon in three pools (live overstory trees, standing snags, and forest floor debris) across a combination of burn severities and pre-fire treatments, 2 years and 8 years after the Rodeo–Chediski Fire. Treatments included prescribed (Rx) fire, a cut and burn treatment, and no treatment. We sampled 106 plots in 36 sites in our ponderosa pine-dominated study area. We found that treatments strongly influenced fire severity; high- and moderate-severity fire was reduced from 76% in untreated areas to 57% in Rx fire treatments and 38% in cut and burn treatments. Fire severity, year, and severity X year were significant factors affecting carbon in the three different pools across the landscape. Eight years post-fire, high-severity burned areas had only 58% of the total carbon (live + dead) that low-severity areas had, and only 3% of the live carbon. Live carbon increased over time in low-severity sites but decreased over time in high-severity sites. We conclude that fuel treatments can significantly influence fire severity, which in turn influences carbon pools. However, treatments may or may not reduce overall carbon loss from an ecosystem in the event of a wildfire given that treatments remove carbon too. Finally, long-term monitoring is important to gain a more complete understanding of post-fire carbon dynamics.

1. Introduction

Fires have been increasing in size and severity in recent decades in many forests of the western United States (Westerling et al., 2006; Miller et al., 2009; Attiwill and Binkley, 2013; Mallek et al., 2013). This has been attributed to the effects of changing climate as well as over 100 years of fire exclusion. Fire exclusion led to a buildup of fuels in ecosystem types that historically burned more frequently and less severely, such as ponderosa pine forests (Swetnam and Baisan, 2003; Westerling et al., 2006). To address altered forest structure and reduce the risk of associated large, severe wildfires, managers use thinning treatments, prescribed fires, and combinations of the two to reduce flammable fuels in southwestern forests (Fulé et al., 2012). Fuel treatments have multiple benefits for forests in addition to reduction of hazardous fuels, including higher understory biodiversity and a more heterogeneous habitat mosaic (Laughlin et al., 2008). It has been suggested that fuel reduction treatments also may help mitigate the sudden loss of carbon from a system in the event of a wildfire (Breshears and Allen, 2002; Hurteau and Brooks, 2011).

On average across the continental U.S., fires emit the equivalent of 4–6% of anthropogenic CO_2 per year (Wiedinmyer and Neff, 2007). However, at a state level, during particularly large fire years, fires can produce more CO_2 per year than fossil fuel burning (Wiedinmyer and Neff, 2007). Carbon stored in forests has become a critical focus of research that aims to improve our understanding

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of a key component of the global carbon cycle as well as how carbon pools change with forest management and natural disturbance.

Whether fuel reduction treatments reduce fire-related loss of carbon from ecosystems in the long term is uncertain. Several modeling studies have agreed that treated stands lose less carbon in subsequent wildfires compared to untreated stands (Finkral and Evans, 2008; Hurteau and North, 2009; Stephens et al., 2012). Additionally, stores of live carbon after wildfire tend to be higher in forests that were previously treated (Dore et al., 2008, 2010, 2012; Meigs et al., 2009; Sorensen et al., 2011; Stevens-Rumann et al., 2013). For example, in California, fuel treatments and wildfires both reduced carbon stored in a forest. However, in untreated portions of the forest that were burned in fire, most carbon (70%) was concentrated in decomposing wood (snags and surface fuels) compared to 19% of carbon stored in decomposing stocks in stands that had been treated before wildfire (North and Hurteau, 2011). Wildfire-related reduction in carbon stocks can last hundreds of years. It was estimated that it will take approximately 230 years to regain the carbon load in Yellowstone that was lost in the 1988 Yellowstone fires (Kashian et al., 2006).

On the other hand, treatments themselves remove carbon from ecosystems (Finkral and Evans, 2008; Hurteau and Brooks, 2011), and several modeling studies have shown that treated stands store less total carbon, even after wildfire, than untreated stands (Mitchell et al., 2009, Reinhardt and Holsinger, 2010; Sorensen et al., 2011). Campbell et al. (2012) conclude in a review that more carbon is typically removed during treatments than is protected from consumption in subsequent wildfires. They write that the only way that fuel treatments could positively impact carbon storage is if they prevent changes to a site's carbon storage ability (e.g., prevent soil loss to erosion after a high-severity fire, or prevent type conversion to a vegetation type that stores less carbon).

Carbon dynamics in forests subjected to fuel treatments and wildfire are not fully understood. Further information is needed to advance our ability to successfully manage forests in the face of climate change, and for a better understanding of the role of forested ecosystems in the global carbon budget (Breshears and Allen, 2002; Restaino and Peterson, 2013). Of particular interest is how carbon dynamics change over time in disturbed landscapes (Goetz et al., 2012), how that carbon is allocated between live and dead carbon pools, and how carbon pools are related to fire severity and forest management history.

Our objectives in the present study were to quantify three above-ground carbon pools (live tree carbon, dead tree carbon, and forest floor carbon) across a combination of burn severities and pre-fire treatments in ponderosa pine forests, 2 years and 8 years after the Rodeo–Chediski Fire. This 2002 wildfire was one of the largest and most severe on record in the Southwest. Our questions were:

1. At the landscape level, how did fuel treatments affect fire severity?
2. How did the three main pools of carbon change over time, in different treatments, in low- and high-severity burned areas?

2. Materials and methods

2.1. Site description

The Rodeo–Chediski Fire burned approximately 189,650 ha in 2002 in northeastern Arizona. At the time, it was the largest recorded fire in the Southwest. Approximately 111,837 ha of the fire were located on White Mountain Apache Tribal (WMAT) land (Fig. 1). The fire exhibited extreme fire behavior, with multiple plume collapses per day and flame lengths up to 60–120 m (Strom, 2005). On WMAT land, nearly 20% of the fire was classified as high-severity, nearly 30% was classified as low-severity, and just over half was classified as moderate severity using remotely sensed Differenced Normalized Burn Ratio (dNBR) maps from the National Park Service/US Geological Survey Burn Severity Mapping Project (2002; Strom, 2005).

Before the Rodeo–Chediski Fire burned in 2002, the White Mountain Apache Tribe had carried out various treatments, including uneven-aged cutting and prescribed (Rx) fire, and prescribed fire alone (description of treatments in Shive et al., 2013). In the previous 11 years, treatments had been completed on approximately 22% of land subsequently burned in the Rodeo–Chediski Fire. Treatments more than 11 years old (pre-1991) were not considered for sampling because burn severity distributions for areas treated before 1991 were not significantly different from the entire burn area (Strom, 2005).

Mid-elevation forests in the study area are dominated by ponderosa pine (*Pinus ponderosa* P. & C. Lawson) with Gambel oak (*Quercus gambelii* Nutt.), alligator juniper (*Juniperus deppeana* Steud.), New Mexico locust (*Robinia neomexicana* A. Gray), and pinemat manzanita (*Arctostaphylos pungens* Kunth). At higher elevations, species include ponderosa pine, white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) (Strom, 2005). Average yearly precipitation at the nearby Heber RAWS Station (2022 m elevation) from 2001 to 2012 was 34.7 cm. Average daily maximum temperature in July from 2001 to 2012 was 29.9 °C, and average daily minimum temperature in January was −7.5 °C.

2.2. Site selection

Two years post-fire (2004), we stratified the WMAT area of the Rodeo–Chediski Fire by pre-fire treatment (cut and burn, Rx fire and no treatment) as well as by fire severity (low and high). Fire severity classes were determined by a combination of remotely sensed dNBR maps and ground-truthing. dNBR has been shown to correspond well with burn severity in the Southwest as measured by the composite burn index (Miller and Yool, 2002; van Wagtendonk et al., 2004; Cocke et al., 2005). We restricted our sampling area to locations with elevations ranging from 2000to 2295 m and with slopes <45%. The elevation constraint ensured that the sampling area was restricted to areas dominated by ponderosa pine. The total sampling area, given elevation, slope, and fire severity constraints (no moderate-severity fire), was 18,378 ha. We did not measure sites for this study in moderate-severity fire areas due to time and funding constraints. In each of the six treatment/severity combinations, we randomly placed six 6-ha sites, for a total of 36 sites. Within each site, we established a systematic grid of 5 plots (Strom, 2005). Eight years post-fire (2010), we remeasured 3 plots in each of the original 36 sites; not all plots were remeasured due to time and funding constraints. We were able to relocate 106 out of 108 of the plots, and we used data from those 106 plots, measured 2 years and 8 years post-fire, in our analysis.

2.3. Field measurements

Two and eight years post-fire, we measured dead and live trees with variable-radius plots, using a prism with a basal area factor (BAF) of $\sim$2.2 $m^2\ ha^{-1}$ per tree to select trees in the plot. The diameter of each tree that fell in the plot was measured at breast height. We measured forest floor fuels on one 15.24-meter (50-foot) planar transect at a random direction from each plot center, using Brown's (1974) method. Fine woody debris was measured in size classes (0–0.64, 0.65–2.54, 2.55–7.62 cm). Coarse woody debris (>7.62 cm diameter) was classified as sound or rotten, and diameter and length of each piece were measured. Species-specific

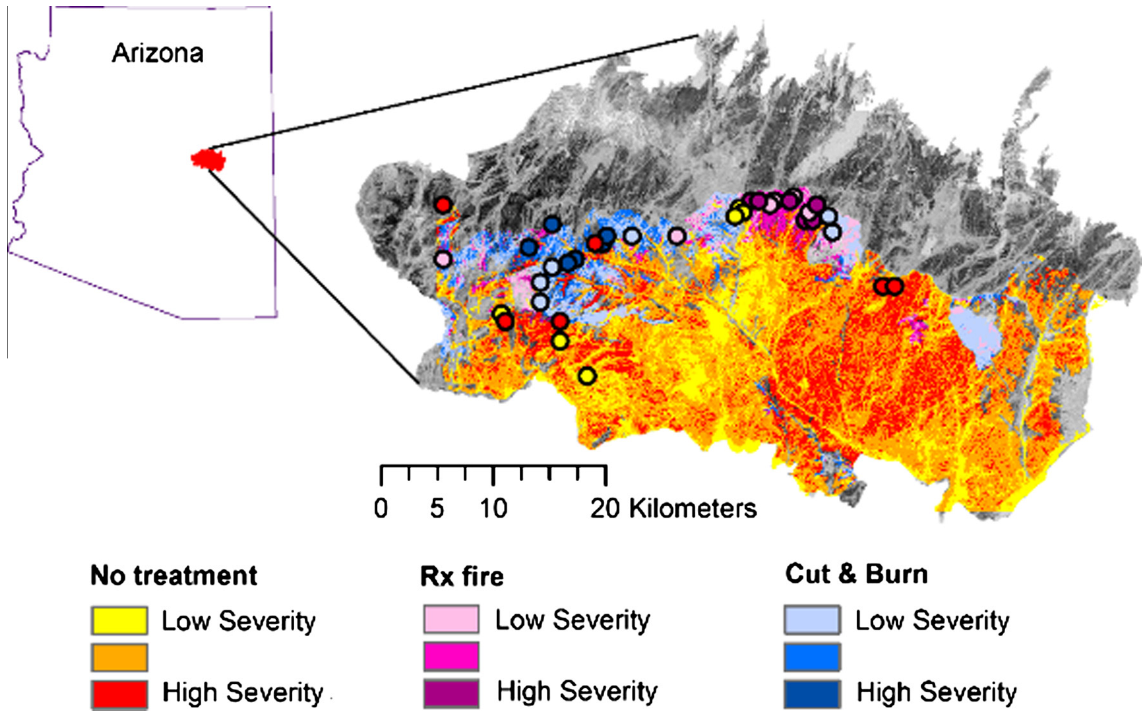

Fig. 1. Map of the 2002 Rodeo–Chediski burn area. Colored areas represent the portion of the fire that burned on White Mountain Apache Tribal lands; black and white areas represent the portion of the fire that burned on the Apache-Sitgreaves National Forest. Circles are sites measured 2 years post-fire (2004) and remeasured 8 years post-fire (2010), color coded by site type. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

coefficients for planar transect calculations are from Sackett (1980).

2.4. Biomass and carbon calculations

To calculate aboveground biomass in live trees and snags, we used allometric equations incorporating relationships between tree diameter and biomass developed for ponderosa pine (Kaye et al., 2005), Gambel oak (Clary and Teidemann, 1986), Douglas-fir (Gower et al., 1992), white fir (Westman, 1987), Chihuahuan pine (*Pinus leiophylla*; Návar, 2009) and pinyon pine (*P. edulis*) and one-seed juniper (*Juniperus monosperma*) (Grier et al., 1992). We used the one-seed juniper equations developed by Grier et al. (1992) for the alligator juniper and Utah juniper (*J. osteosperma*) in our plots. We used the ponderosa pine equations for the 2 southwestern white pine trees encountered in the plots, and the Gambel oak equations for the other oak species, the 1 bigtooth maple (*Acer grandidentatum*) snag, and the 2 New Mexico locusts found in the plots. No manzanita was found in our plots. We included all tree components for live trees, but progressively eliminated foliage, branches, and bark for increasing decay classes of snags (North and Hurteau, 2011). We did not calculate reduced biomass in snags over time. Irvine et al. (2007) assumed a snag decomposition rate of 0.3% per year for ponderosa pine but we decided not to calculate reduced biomass in snags for several reasons: the 0.3% value is an estimate, we did not know death dates for snags, and this rate is small enough that it would not make a substantial difference in standing biomass over the 6 years between our measurements. We standardized all data to per-hectare estimates. To convert litter and duff depth to forest floor fuel loadings in megagrams per hectare (Mg ha^{-1}), we used coefficients from Ffolliott et al.'s (1968) models predicting forest floor weight from depth in northern Arizona ponderosa pine stands.

To calculate carbon loads, we used carbon concentrations measured in ponderosa pine-dominated forests in northern Arizona (Jain et al., 2010). We averaged the carbon concentration values for the two ponderosa pine forests measured in Jain et al. (2010); the values for the two forests were similar. Carbon concentrations we used were: overstory crown 49.9%; bole 49.3%; sticks < 7.5 cm diam 49.1%; solid logs > 7.5 cm diam 50.0%; rotten logs > 7.5 cm diam 48.6%; litter 37.0%; humus 32.7%.

2.5. Analysis

We evaluated our first question, about how fuel treatments affect fire severity, in the portion of the fire on WMAT lands where elevation ranged from 2000 to 2295 m and slopes were <45%. We included moderate-severity burned areas in this analysis. This resulted in a study area of 34,020 ha. We tested for differences in fire severity distributions (high-, moderate-, and low-severity) in different treatment types: no treatment, Rx fire, and cut and burn. We used a Kolmogorov–Smirnov test of the raw dNBR distributions for the treatment types.

We evaluated our second question, about the effects of time since fire, fuel treatments and fire severity on carbon stored in live trees, dead trees and forest floor material using repeated-measures multivariate analysis of variance (MANOVA). Plot data were averaged within each site prior to the analysis. The MANOVA was used because we had three response variables that were not independent. We considered test statistics significant when p-values <0.05. We checked for compliance with normality and homogeneity of variance assumptions for the three response variables using a Shapiro–Wilk test of normality on MANOVA residuals and a Levene's test of homogeneity of variance. All three variables were log +1 transformed, but even after transformation not all variables met all assumptions. MANOVAs are robust to the violation of homogeneity of variance assumption if sample sizes are equal (as in this case) and robust to the violation of the assumption of normality if non-normality is caused by skew and not outliers (as in this case) (Bray and Maxwell, 1985); nevertheless, *p*-values close to $\alpha = 0.05$ should be interpreted with caution. All analyses were done in SPSS version 21 (SPSS Inc., Chicago, U.S.A.).

3. Results

3.1. At the landscape level, how do fuel treatments affect fire severity?

Fuel treatments had a large effect on fire severity at the landscape level. All pairwise comparisons of the severity distributions for each treatment type were significantly different. The combination of high-severity and moderate-severity fire was reduced from 76% of untreated areas to 57% of the area in Rx fire treatments and 38% of cut and burn treatments (Fig. 2).

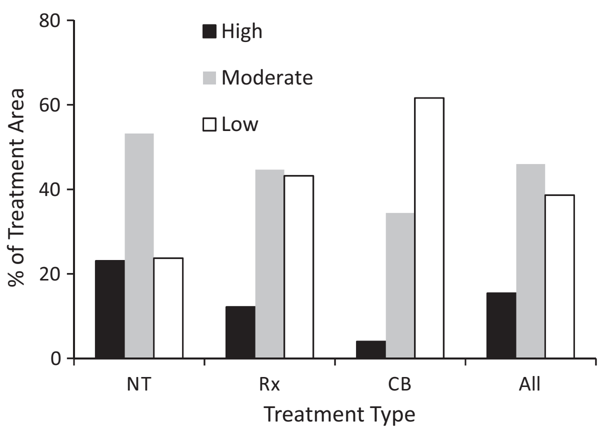

Fig. 2. Burn severity distribution after the Rodeo–Chediski Fire for untreated areas (NT), areas that underwent fuel reduction treatments between 1991 and 2001 (Rx: prescribed fire, CB: cut and burn), and all areas (All).

Table 1
Results of repeated-measures MANOVA. The dependent variables were carbon values in live trees, snags, and forest floor material. Significant effects ($p < 0.05$) are shown in bold.

Source	Wilks' Lambda	df	*F*	*P*
Between subjects				
Severity	**.080**	**3, 27**	**369.006**	**<.0001**
Treatment	.758	6, 54	1.336	.257
Severity x treatment	.888	6, 54	.551	.767
Within subjects				
Year	**.195**	**3, 27**	**37.042**	**<.0001**
Year × severity	**.373**	**3, 27**	**15.108**	**<.0001**
Year × Treatment	.781	6, 54	1.184	.329
Year × Severity × Treatment	.700	6, 54	1.757	.126

3.2. How do the three main pools of carbon (live tree carbon, dead tree carbon, and forest floor carbon) change over time, in different treatments, in low- and high-severity burned areas?

MANOVA results indicate that year, severity, and year × severity are significant factors associated with pools of carbon, whereas treatment effects were not detected (Table 1). The three main carbon pools (live trees, dead trees, and forest floor carbon) did not change uniformly over time (Table 2, Fig. 3). Two years post-fire, live tree carbon was approximately an order of magnitude higher in low-severity than in high-severity sites. This difference increased to approximately two orders of magnitude by 8 years post-fire. The quantity of live tree carbon increased between 2 and 8 years post-fire in low-severity areas, but decreased in high-severity burned areas. Carbon stored in snags in both low-severity and high-severity burn areas decreased over time from 2 to 8 years post-fire, but the patterns converged in this case: snag C in high-severity sites averaged 22.6 Mg ha^{-1} after 2 years, about 3 times greater than low-severity sites (average 6.5 Mg ha^{-1}). By 8 years post-fire, the average values were 5.4 and 1.9 Mg ha^{-1}, respectively. Forest floor carbon increased in both low- and high-severity burn areas between 2 and 8 years post-fire (Fig. 3), with the biggest shift appearing in the high-severity sites, representing a transfer from the snag C pool to the down C pool. In sum, 2 years after the burn, the low- and high-severity sites differed in the live:dead proportion of C, but total C was relatively similar, 37.2 and 32.7 Mg ha^{-1}, respectively. By 8 years post-fire, however, tree growth in the low-severity sites increased average total aboveground C by 10.7% to 41.2 Mg ha^{-1}, while decomposition of dead trees in the high-severity sites reduced average total aboveground C by 27.5% to 23.7 Mg ha^{-1}.

Fig. 3. Carbon (Mg ha^{-1}; ±1 SE) measured in three pools 2 years and 8 years post-fire, combined across treatments in high-severity and low-severity burn areas.

Table 2
Average carbon mass (Mg ha^{-1}; ±1 SE) among years, fire severity classes, and pre-fire fuel treatments after the Rodeo–Chediski Fire in Arizona. Severity: H = high severity, L = low severity. Treatment: NT = no treatment, Rx = prescribed fire, CB = cut and burn.

Year	Severity	Treatment	Live C	Snag C	Down C	Total aboveground C
2004	H	NT	2.8 (±1.8)	29.7 (±9.2)	3.6 (±0.9)	36.1 (±9.5)
		Rx	5.6 (±1.7)	19.0 (±1.5)	5.6 (±1.3)	30.3 (±2.8)
		CB	3.3 (±1.1)	19.0 (±2.8)	9.4 (±4.3)	31.7 (±5.6)
	L	NT	31.9 (±3.0)	8.8 (±4.7)	4.9 (±1.2)	45.6 (±5.1)
		Rx	16.5 (±0.9)	7.4 (±1.7)	4.9 (±1.6)	28.8 (±1.2)
		CB	29.3 (±2.7)	3.2 (±1.3)	4.8 (±0.9)	37.2 (±3.6)
2010	H	NT	0.8 (±0.6)	12.7 (±7.5)	18.8 (±3.2)	32.3 (±7.2)
		Rx	0.3 (±0.3)	1.7 (±0.3)	19.5 (±5.1)	21.5 (±5.0)
		CB	1.5 (±0.7)	1.7 (±0.7)	14.1 (±3.0)	17.3 (±3.3)
	L	NT	37.0 (±3.2)	1.9 (±0.5)	9.7 (±4.4)	48.6 (±4.0)
		Rx	21.1 (±3.0)	1.5 (±0.7)	11.9 (±4.8)	34.5 (±6.6)
		CB	29.5 (±2.2)	2.2 (±1.0)	8.6 (±1.7)	40.4 (±4.3)

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Although MANOVA results indicate that treatment was not a significant factor affecting carbon in plots that were stratified by severity and treatment, some trends are evident. Untreated sites that burned in low-severity fire had more live carbon, both 2 and 8 years post-fire, than sites that had been treated with Rx fire or a cut and burn treatment. Areas that had been cut and burned had the second-highest amounts of live carbon after low-severity fire, and Rx fire treatments had the lowest live carbon after low-severity fire. After high-severity fire, areas treated with Rx fire had the highest carbon values after 2 years but after 8 years areas that had been cut and burned had the highest carbon values. Both 2 and 8 years post-fire, untreated sites had more total carbon (live and dead) than sites in either treatment category.

4. Discussion

Fuel treatments were associated with significantly reduced wildfire severity in our study area. Untreated areas had the highest proportion of high-severity fire and moderate-severity fire, and cut and burn treatments were associated with the least amount of high- and moderate-severity fire. In turn, fire severity had a significant impact on carbon pools. We found a large difference in carbon values over time between areas that burned with high severity and those that did not. Eight years post-fire, average live tree biomass in high-severity burned areas was only 3% of live tree biomass in low-severity burned areas. Our results were similar to findings by Dore et al. (2008), who found that ten years after a severe wildfire in northern Arizona, live aboveground biomass in the burned area was only 2% of live aboveground biomass in a nearby unburned site, but woody debris in the burned area was about 5 times higher. In the Dore et al. (2008) study, the burned site was a carbon source to the atmosphere all year long, and the authors concluded that severe fire had a strong and persistent effect on the carbon balance in their study area. In southern Oregon, after the Biscuit Fire, Campbell et al. (2007) found that low-severity patches of fire released 70% as much carbon as high-severity patches of fire. Although we do not have estimates of carbon released, we can compare total post-fire carbon values in low-severity and high-severity burned areas. Two years post-fire, on average across treatments, high-severity patches contained 88% of the carbon that low-severity burned areas contained. By 8 years post-fire, high-severity burned areas had only 58% of the carbon that low-severity areas did.

We chose to lump all treatments that had taken place within 11 years pre-fire (1991–2001) because burn severity distributions in older treatments were no different than the burn severity distribution of the entire study area (Strom, 2005). This suggests that 11 years was approximately the length of time fuel treatments were effective in reducing fire severity during the Rodeo–Chediski Fire. Treatments affected fire severity within the treatment boundaries themselves and also may help protect areas on the lee-sides of treatments during severe wildfire (Finney et al., 2005).

Tree regeneration and other plant growth were not included in this study, but will affect carbon dynamics. Shive et al. (2013) found that ponderosa pine regeneration (41–137 cm in height) on the same sites described in this paper was higher in high-severity than low-severity burned areas, averaging 889 and 69 seedlings ha^{-1} respectively. Although pine seedlings contain relatively low stores of carbon compared to large trees, they will store more and more carbon as the trees grow. In places where seedlings are coming in, we would expect live carbon storage in low-severity and high-severity burned areas to equalize over the long term. However, seedling density was found to be highly variable in the Rodeo–Chediski burned area (Shive et al., 2013), and large differences in seedling density can result in highly heterogeneous mature stand densities for up to 200 years (Kashian et al., 2005). Tree establishment may take decades, centuries, or not happen at all in high-severity fire areas (Dore et al., 2008). Savage and Mast (2005) showed persistent type conversion is possible after severe fire in southwestern ponderosa pine forests, from forest to shrublands or grasslands. Long-term monitoring of the uncharacteristically large high-severity patches of fire in the Rodeo–Chediski Fire is warranted.

Our results showed that time since fire was an important factor influencing carbon measurements after the Rodeo–Chediski Fire. Between 2 and 8 years post-fire, live tree carbon decreased in high-severity areas, presumably due to delayed post-fire tree mortality (Fulé and Laughlin, 2007). Over time, snag carbon decreased and forest floor carbon increased in both low- and high-severity burn areas, as standing dead trees began to fall. Our results strengthen arguments for long-term monitoring of plots post-fire, because initial estimates may underestimate post-fire overstory mortality. Additionally, snag fall rates may have an impact on decay rates. It has been shown in other parts of the world (although not to our knowledge in the dry Southwest) that logs decay faster than snags due to their contact with the ground and higher moisture content (Mattson et al., 1987; Boulanger and Sirois, 2006).

There are several limitations to this study which we acknowledge. First, we do not have carbon estimates for all carbon pools in the ecosystem, including shrubs and small trees (trees < 1.3 m tall) and belowground carbon. Second, we do not have estimates of net primary productivity (NPP); non-tree productivity can compensate and reduce the gap between NPP in low- and high-severity burned areas (Campbell et al., 2009; Meigs et al., 2009). Third, there are carbon costs to fuel treatments as well, which we are unable to calculate with the available data. For example, Finkral and Evans (2008) estimated that carbon costs for a thinning treatment, including driving to the site, operating the logging equipment, burning slash, processing and delivering firewood, and burning firewood, were approximately 12.5 Mg ha^{-1}. Finally, we were able to estimate carbon 2 and 8 years post-fire and it would be valuable to assess carbon dynamics over an even longer time period. However, this study is valuable because we are able to report change through time in carbon in live trees, dead trees, and surface fuels (including down trees) after a wildfire in stands that were subjected to different treatments before the fire. This empirical information helps contribute to our understanding of carbon dynamics after wildfire in the Southwest.

Hurteau and Brooks (2011) describe two main approaches for carbon management in dry, fire-prone forests of the southwestern United States: carbon stabilization and carbon maximization. Carbon stabilization would prioritize reducing the risk of carbon loss in a stand-replacing wildfire by altering stand structure. Carbon maximization would prioritize carbon storage on a landscape, but this approach comes with a high risk of high-severity, stand-replacing wildfire (Hurteau et al., 2011) or other density-dependent disturbance. Campbell et al. (2012) write that over multiple disturbance cycles, over hundreds of years, the long-term average of carbon stored in forested landscapes with infrequent, high-severity fire may be higher than in forested landscapes with frequent, low-severity fire. Carbon maximization may be achieved in this scenario until a high-severity fire occurs. However, ponderosa pine forests in the Southwest historically did not experience high-intensity fire at the scale seen in the Rodeo–Chediski Fire (Swetnam and Baisan, 2003). Recovery in this southwestern ponderosa pine forest may not be equivalent to places where large stand-replacing fires were historically more common and trees have adaptive traits such as serotinous cones that help them regenerate after high-intensity fires (Turner et al., 2003). In dry forests of the Southwest, carbon maximization may be achievable only until the first high-severity disturbance occurs, if the ecosystem is not able to recover to pre-fire carbon levels due to soil loss or lack of tree regeneration.

The Rodeo–Chediski Fire significantly affected carbon storage at a landscape scale, and much of this ~190,000 ha fire will release carbon for decades or centuries. The Rodeo–Chediski Fire was surpassed in size in 2010 by the Wallow Fire (~218,000 ha), also in eastern Arizona. After over a century of considerable ecological change, these large fires are burning with substantial amounts of high severity, resulting in significant carbon losses over large scales and the conversion of some of the forests from carbon sinks to carbon sources. With wildfires in the Southwest increasingly large and fuel treatments more common, it is becoming more likely that wildfires will burn into previous treatments and that those treatments will have an impact on fire severity, subsequent forest structure, and forest carbon storage (Waltz et al., 2014). Where managing forests for carbon storage is a priority, fuels treatments that can reduce long-term carbon loss will need to play a role in forest management.

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