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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.

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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

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