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Prescribed fire effects on field-derived and simulated forest carbon stocks over time

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ABSTRACT

To better understand the impact of prescribed fire on carbon stocks, we quantified aboveground and belowground carbon stocks within five pools (live trees and coarse roots, dead trees and coarse roots, live understory vegetation, down woody debris, and litter and duff) and potential carbon emissions from a simulated wildfire before and up to 8 years after prescribed fire treatments. Total biomass carbon (sum of all the pools) was significantly lower 1 year post-treatment than pre-treatment and returned to 97% of pre-treatment levels by 8 year post-treatment primarily from increases in the tree carbon pool. Prescribed fire reduced predicted wildfire emissions by 45% the first year after treatment and remained reduced through 8 year post-treatment (34%). Net carbon (total biomass minus simulated wildfire emissions) resulted in a source (10.4–15.4 Mg ha^{-1}) when field-derived values were compared to simulated controls for all post-treatment time periods. However, the incidence of potential crown fire in the untreated simulations was at least double for the 2 year and 8 year post-treatment time periods than in the treated plots. We also compared field-derived estimates to simulated values using the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS). In our validation of FFE-FVS to predict carbon stocks, the model performed well for the total biomass carbon (4% difference); however, there was great variability within the individual carbon pools. Live tree carbon had the highest correlation between field-derived and simulated values, and dead tree carbon the lowest correlation and highest percent differences followed by herb and shrub carbon. The lack of trends and variability between the field-derived and simulated carbon pools other than total biomass indicate caution should be used when reporting carbon in the individual pools.

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1. Introduction

Forest ecosystems play an important role in the global carbon cycle; they are both sources and sinks of carbon (Depro et al., 2008; McKinley et al., 2011; Pan et al., 2011). Forests store about 45% of terrestrial carbon with about 60% in trees (Malmsheimer et al., 2011; Ryan et al., 2010). Within forests, the aboveground carbon pools are more dynamic than soil pools and are more affected by near-term management activities and disturbance (Hines et al., 2010). Forest management, land use change, and disturbances such as wildfire, storms, and insects all affect carbon pools. US forests are currently a carbon sink primarily because of afforestation and fire suppression since settlement (Birdsey et al., 2006; Houghton et al., 2000); however, the current sink may decline even under current suppression tactics through the 21st century as woody encroachment reaches its maximum extent and ecosystem recovery slows (Hurtt et al., 2002). Ironically, wildfire is one of the primary threats to carbon storage in dry forests of the Western US due in part to the elevated biomass or fuel levels that create the sink (Malmsheimer et al., 2011). Fire initially releases large amounts of carbon into the atmosphere as a result of the combustion of living vegetation and dead fuels. Additional carbon is released from the decomposition of fire-killed vegetation where carbon was initially stored, which is released over time as it decomposes (Harmon and Marks, 2002; Ryan et al., 2010). Typically, the impact of fire is a short-term phenomenon offset by the uptake of carbon by surviving and new vegetation following the fire (Canadell et al., 2007; Kashian et al., 2006). The recovery time is dependent on the intensity and frequency of fires, and the ability of the system to regenerate post disturbance due to factors such as site quality, soil loss, and seed source (Kashian et al., 2006). High intensity stand-replacing fire in forests adapted to low-severity fire is one of the largest risks to carbon storage because forests may not regenerate afterward resulting in a vegetation-type conversion

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(Ryan et al., 2010). A net loss will occur if the frequency of disturbance is shorter than the recovery period (Campbell et al., 2012; Kashian et al., 2006; Smithwick et al., 2002).

Fuel treatments have been shown to reduce the severity of wildfires (i.e., Lyons-Tinsley and Peterson, 2012; Moghaddas and Craggs, 2007; Pollet and Omi, 2002; Safford et al., 2012) and therefore reduce losses of carbon (North and Hurteau, 2011). However, there is a debate on the role of fuel treatments in the carbon balance of forests. One side of this debate hinges on the likelihood of a wildfire encountering a fuel treatment. Fuel treatments may be applied to areas that do not subsequently experience wildfire resulting in carbon reductions from the treatment without the carbon benefit from reduced wildfire emissions. In the western US, Rhodes and Baker (2008) found an 8% chance that fuel treatments were subsequently burned by wildfire in a 20 year period. Similarly, Campbell et al. (2012) found that ten locations must be treated in order to beneficially impact future fire in just one location. On the other hand, carbon emissions from the fuel treatment plus the reduced emissions from subsequent wildfire may be less than the greater emissions from a more intense wildfire in untreated fuels.

There are three approaches available to explore the impacts of fuel treatments on carbon stocks if a wildfire occurs. The first simulates stand data, treatments, and effects (i.e., Harmon and Marks, 2002; Mitchell et al., 2009). The second uses empirical stand data coupled with simulated fuel treatments and effects (i.e., Hurteau and North, 2009; Reinhardt and Holsinger, 2010). The third uses purely empirical data collected before and after fuel treatments were conducted. To date, the majority of publications that quantify fuel treatment effects on forest carbon stocks use empirical data with pre-treatment and immediate or near immediate post-treatment data (i.e., Finkral and Evans, 2008; North et al., 2009; Sorensen et al., 2011; Stephens et al., 2009, 2012). Currently only three studies go beyond the scope of immediate effects of fuel treatments on carbon stocks with empirical data (Boerner et al., 2008; Hurteau and North, 2010; Hurteau et al., 2011). Simulation modeling permits assessment of the long-term impacts (>20 years) of treatments on carbon stocks. However, more empirically based research is needed to understand the effects of fuel treatments on carbon pools, and to assess the accuracy of simulated outputs over the same time span.

In this study we calculated carbon stocks in various aboveground and belowground pools based on field data before and up to 8 years after treatment by prescribed fire in central and northern California. The goals of this study were to better understand how prescribed fire treatments affect forest carbon stocks over time and to assess the accuracy of modeling carbon stocks into the future using the Fire and Fuels Extension (FFE-FVS, Rebain, 2010; Reinhardt and Crookston, 2003) to the Forest Vegetation Simulator (FVS, Crookston and Dixon, 2005). The specific questions addressed are: (1) How do forest carbon stocks change over time? (2) How do potential carbon emissions vary from simulated wildfire over time? and (3) Do forest carbon stocks differ between field-derived and simulated values? This study is unique from existing research (Boerner et al., 2008; Hurteau and North, 2010; Hurteau et al., 2011) because of the regional scope, and it will be a first to assess the accuracy of simulated versus field-derived forest carbon stocks between various carbon pools.

2. Methods and materials

2.1. Study area

California is divided into three broad eco-region divisions based on precipitation amount and patterns as well as temperature (Bailey et al., 1994; Bailey, 1996). All of our plots fall within the Mediterranean division, which is characterized by temperate rainy winters and hot dry summers. Further classification into eco-region domains, provinces, and sections are based on vegetation, natural land covers, and terrain features (Bailey, 1996; Bailey et al., 1994; Miles and Goudey, 1997). Sugihara and Barbour (2006) created nine bio-regions in California by combining the 19 eco-region sections within California (Miles and Goudey, 1997) based on vegetation and fire regime. Our plots were within five of the nine bio-regions (Fig. 1): Sierra Nevada ($n = 9$), North Coast ($n = 4$), Southern Cascade ($n = 7$), Klamath Mountains ($n = 2$), and Northeastern Plateau ($n = 3$). Conifer species present in the plots included: white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), incense cedar (*Calocedrus decurrens* (Torr.) Florin), western juniper (*Juniperus occidentalis* Hook.), Jeffrey pine (*Pinus jeffreyi* Balf.), sugar pine (*Pinus lambertiana* Douglas), ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Hardwood species present in the overstory included: big leaf maple (*Acer macrophyllum* Pursh), tanoak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehder), canyon live oak (*Quercus chrysolepis* Liebm.), and California black oak (*Quercus kelloggii* Newberry). The elevation of the plots ranged from about 700–1650 m on all aspects. Slopes ranged from level ground to 48%.

2.2. Field sampling

The data used in this study were from a larger regional monitoring program to characterize pre- and post-treatment fuels and vegetation as a result of fuel treatments on national forests in California (Vaillant et al., 2009a; Vaillant et al. 2009b). Personnel on each national forest were contacted and asked to provide candidate fuel treatment projects that they expected to treat in the near future. This study includes only prescribed fire treatments that

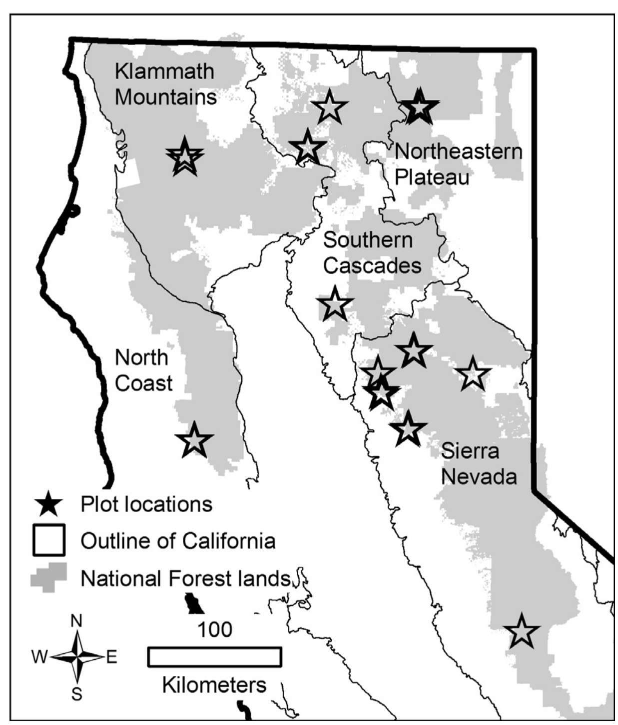

Fig. 1. Study plot locations, national forests, and ecoregions (Sugihara and Barbour, 2006) within California. All plots were established prior to treatment, then revisited 1 year, 2 year, and 8 year after the prescribed fire.