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Carbon recovery rates following different wildfire risk mitigation treatments

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ABSTRACT

Sequestered forest carbon can provide a climate change mitigation benefit, but in dry temperate forests, wildfire poses a reversal risk to carbon offset projects. Reducing wildfire risk requires a reduction in and redistribution of carbon stocks, the benefit of which is only realized when wildfire occurs. To estimate the time needed to recover carbon removed and emitted during treatment, we compared the 7-year post-treatment carbon stocks for mechanical thinning and prescribed fire fuels reduction treatments in Sierra Nevada mixed-conifer forest and modeled annual carbon accumulation rates. Within our 7-year re-sample period, the burn only and understory thin treatments sequestered more carbon than had been removed or emitted during treatment. The understory thin and burn, overstory thin, and overstory thin and burn continued to have net negative carbon stocks when emissions associated with treatment were subtracted from 7-year carbon stock gains. However, the size of the carbon deficit in the understory thin and burn 7 years post-treatment and the live tree growth rates suggest that the remaining trees may sequester treatment emissions within several more years of growth. Overstory tree thinning treatments resulted in a large carbon deficit and removed many of the largest trees that accumulate the most carbon annually, thereby increasing carbon stock recovery time. Our results indicate that while there is an initial carbon stock reduction associated with fuels treatments, treated forests can quickly recover carbon stocks if treatments do not remove large, fire-resistant overstory trees.

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1. Introduction

Recent policy interest in forests' climate change mitigation potential has resulted in rapidly evolving carbon accounting protocols. Many of these protocols enable landowners to alter their management practices and financially benefit from sequestering more carbon than their business-as-usual management practices. This can include reforestation and increasing the time between harvests. Another option is altering management actions to increase carbon density (Canadell and Raupach, 2008). In Oregon and California, Hudiburg et al. (2009) estimate that baring stand-replacing disturbance, landscape carbon stocks could be increased by 46% to achieve a theoretical maximum carbon stock. In temperate wet forests, such as those found west of the Pacific Northwest's Cascade Mountains, the high productivity and long time interval between natural disturbance events increases the likelihood that a theoretical maximum carbon stock could be achieved. However, in dry temperate forests, natural disturbances, such as fire, were historically more frequent than in these temperate wet forests.

Prior to the 1880s, fire frequency ranged from 2 to 21 years in ponderosa pine and Jeffrey pine forests to 11–30 years in Sierran

mixed-conifer forests (Swetnam and Betancourt, 1990; McKelvey and Busse, 1996; Brown et al., 1999; Everett et al., 2000; Taylor and Skinner, 2003; North et al., 2005; Taylor and Beaty, 2005). Past management activities, including fire suppression, have decreased the frequency of this natural disturbance resulting in an increase in stem density. If forest conditions are otherwise unchanged (i.e., *ceteris paribus*), in-growth due to fire suppression should have increased forest carbon stocks (Hurtt et al., 2002). In California, however, a comparison between forest inventory data from the 1930s and 1990s indicates that while there has been a lack of disturbance and increased stem density, a net loss of large trees has resulted in decreased live tree carbon stocks (Fellows and Goulden, 2008). Commensurate with this increase in stem density is an increase in high-severity fire resulting from increased fuel accumulations that allow surface fire to move into the forest canopy (Stephens, 1998; Miller et al., 2009), increasing fire severity and presenting a risk to forest carbon offset projects (Galik and Jackson, 2009; Hurteau et al., 2009a).

In dry temperate forests that have heavy fuel loads, reducing the reversal risk from fire (i.e., the risk that carbon sequestered in trees reverts back to the atmosphere) requires a near-term reduction in the live tree carbon stock (Hurteau et al., 2008; Campbell et al., 2009; Hurteau and North, 2009; North et al., 2009; Stephens et al., 2009). These treatments can reduce wildfire emissions when ignition occurs making the near-term carbon stock reduction bene-

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ficial if a forest burns (Hurteau and North, 2009; North et al., 2009). However, it has been suggested that reducing high-severity fire risk runs counter to maximizing long-term carbon storage if fuels reduction treatments remove more carbon than would have been lost by wildfire burning in untreated forest (Mitchell et al., 2009). Thus, determining the time period necessary to re-sequester the carbon removed through fuels reduction treatments is important for determining the long-term carbon costs and benefits of fuels reduction treatments.

In this study we build upon our previous work that quantified the pre- and post-treatment carbon stocks for six treatments in Sierran mixed-conifer forest (North et al., 2009). Seven years after treatments, we re-measured carbon stocks in soil, fine and coarse woody debris, fine tree roots, and all trees and snags ≥ 75 cm dbh to quantify changes in these carbon stocks. Unlike our complete stem survey in 2002, for trees and snags < 75 cm dbh we re-measured a sub-sample ($n = 240$) of these smaller stems which typically make up $< 35\%$ of the live tree carbon in old-growth mixed conifer (North et al., 2009). Using this sub-sample we model changes in carbon stocks in the smaller live and dead stems and add these changes to the re-measured values in the other carbon stocks to compare carbon stocks in 2002 and 2008. We use this estimate of carbon stock change in live trees and shrubs over the post-treatment period to quantify the net gain or loss in carbon stocks between different treatments when emissions associated with each of the treatments are subtracted. We focus our analysis on the carbon pools that are most directly affected through management activities and most commonly quantified in forest carbon offset projects.

2. Materials and methods

2.1. Study area

This study was conducted at the Teakettle Experimental Forest (<http://teakettle.ucdavis.edu>) in California's southern Sierra Nevada Mountains. The site ranges in elevation from 1900 to 2600 m and has a Mediterranean climate with almost all of the 125 cm of annual precipitation falling as snow (North et al., 2002). Teakettle's mixed-conifer forest is comprised primarily of white fir (*Abies concolor*), red fir (*A. magnifica*), incense-cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), and Jeffrey pine (*P. jeffreyi*) (Rundel et al., 1988). Eighteen permanent four-hectare plots were established that represented the range of variable forest conditions at the site. A pre-treatment analysis of stand conditions indicated that there were no significant differences in forest structure between plots (North et al., 2002).

2.2. Treatments

Six treatments from a full factorial design that included three levels of mechanical thinning treatment (no thin, understory thin, overstory thin) and two levels of prescribed burning treatment (no burn, prescribed fire) were applied to the plots from 2000 to 2001. The understory thin treatment removed all trees between 25 and 75 cm diameter at breast height (dbh), following California spotted owl guidelines (Verner et al., 1992). Although initially designed to minimize impacts on spotted owl habitat, the guidelines have been widely implemented to reduce fuels in Sierran mixed-conifer forests. The overstory thin removed all trees > 25 cm dbh, except 22 large trees ha^{-1} left regularly spaced approximately 20 m apart. In general this resulted in thinning trees up to 100 cm dbh, and produced a sparse canopy with widely separated tree crowns. The thin and burn plots were mechanically treated in 2000 and burned in 2001. The thin only plots were treated in 2001.

2.3. Data collection

Pre-treatment data collection methods included mapping, using a surveyor's total station, measuring, and permanently tagging all trees and snags ≥ 5 cm dbh. Fuels, fine roots, soil carbon, and understory plant cover were measured on permanent sample points established on a grid within each plot. Understory plants (herbs and shrubs) were sampled using a 10 m² circular plot and re-surveyed each year through 2006. Mass of the fine woody debris (FWD) was estimated before and after treatment, (the controls were only sampled once), using the planar intercept method (Brown, 1974), with modifications, at nine sample points within each plot. At each sample point a random bearing was chosen and two additional bearings chosen at 120° from the first. At each bearing a 15 m line transect was established. The number of intercepts of 0.1–0.6 cm diameter pieces (1-h fuels) and 0.6–2.5 cm diameter pieces (10-h fuels) were recorded along the first 2 m of the transect and 2.5–7.6 cm diameter pieces (100-h fuels) along the first 4 m. Pieces 7.6–29 cm diameter (1000-h fuels) were recorded along the entire 15 m transect. For the 1000-h fuels a cut-off was made in the upper range of the fuel size to avoid overlapping with a complete coarse woody debris (CWD) inventory of all pieces ≥ 30 cm diameter (see below). In this CWD inventory the end-points of each qualifying log were mapped using the total station, and the diameters recorded. Log decay was determined using a modification of the five decay classes of Maser et al. (1979). We did not include decay class 5 CWD in our inventory because field technician estimates of piece sizes were inconsistent and our soil samples included representative sampling of carbon from highly decayed CWD. Using the mapped coordinates, log length was calculated. The volume of each log was estimated as a frustrum paraboloid (Husch et al., 1993). Mass (Mg ha^{-1}) was estimated using the specific gravities of Harmon et al. (1987). Since we did not record the species of the downed logs in the pre-treatment survey and species were often unidentifiable we averaged the specific gravities of Harmon et al. (1987) by decay class for the dominant species found at Teakettle: decay 1 = 0.38; 2 = 0.32; 3 = 0.27; 4 = 0.15 g cm^{-3} . A detailed reporting of the pre and post-treatment methods can be found in North et al. (2009), Wayman and North (2007), and Innes et al. (2006).

In 2008, 7 years post-treatment, we re-measured soil carbon, fine roots, and fuels at the same nine sample points in each plot (total $N = 162$) following the same protocols used in previous measurement periods. We re-measured all trees ≥ 75 cm dbh ($N = 1908$) and a sub-sample of trees < 75 cm dbh ($N = 240$). In our analysis we separated trees greater and less than 75 cm dbh because trees > 75 cm dbh are now rarely harvested on federal forest land in the Sierra Nevada (SNFPA, 2004) and in previous research we found they constitute $> 65\%$ of live tree carbon (North et al., 2009). In the 2008 sampling, a mapped inventory of coarse woody debris (CWD) was not conducted. In this sampling we included CWD pieces > 30 cm diameter in our planar intercept fuels transect (Brown, 1974). In our 2008 sample our calculations of C in the litter also included duff, as in the 2003 sampling North et al. (2009) reported C in the litter alone. Soil carbon and fine root sampling in 2008 followed the same protocol used in 2002 immediately after treatments (Wayman and North, 2007; Ryu et al., 2009). At each of 9 grid points, three 2 cm diameter soil cores were extracted and aggregated by 0–10 and > 10 –30 cm depths. Samples were kept on ice for no more than 10 h, air dried to a constant weight, and passed through a 2 mm sieve. The ANR analytical lab at the University of California, Davis, conducted total carbon analyses. Fine roots were sampled at two depths (0–10 and 10–20 cm) using a 7 cm diameter soil corer. Using a root washer, roots were separated into fine (≤ 2 mm) and coarse (> 2 mm) roots. Following drying at 65 °C for 48 h, fine and coarse roots were weighed. Previous work at Teakettle found that coarse roots often extend more than 2 m deep and as a

result we estimated coarse root biomass using allometric equations developed by Jenkins et al. (2004).

2.4. Carbon calculations

We used genus-specific allometric equations presented in Jenkins et al. (2004) to calculate tree and snag biomass. Coarse and fine woody debris biomass was calculated following Brown (1974) and carbon concentration was assumed to be 50% of biomass (Penman et al., 2003). Carbon in litter and duff were quantified using a carbon concentration of 37% (Smith and Heath, 2002). In a previous Teakettle study (Hurteau and North, 2008; Hurteau et al., 2009b) we determined that shrub biomass could be estimated from percent cover estimates using:

$$y = 1.372014x + 2.576618, \quad r^2 = 0.80$$

where y is equal to biomass and x is equal to percent cover. We are not aware of any studies estimating the carbon content of the California shrub species that occur at this site and therefore used an estimate from a site in northern California where carbon is equal to 49% of biomass (Campbell et al., 2009). Shrub cover was measured annually at Teakettle through 2006. We modeled gains in shrub carbon from 2006 to 2008 using the methodology described below.

2.5. Analysis

We used two different analyses of changes in carbon (C) stocks. Since we did not conduct a complete re-measure of all trees and snags in year seven, for stems < 75 cm dbh we compared the sub-sampled 2008 live tree C values with their 2002 C values and the live tree C in 2002 that transitioned to snags in 2008. We used these values to model changes in small stem carbon dynamics. Since we completely re-measured all trees and snags ≥ 75 cm dbh ($n = 1908$), we were able to conduct a between treatment comparison of the percent change (to normalize for differences in initial carbon stocks between treatments) in large stem carbon stocks 7 years post-treatment.

Field measured litter and duff, fine and coarse woody debris, fine roots, and soil carbon were standardized to per hectare values (MgC ha^{-1}). We tested differences in treatment means using ANOVA with Tukey's HSD mean comparison and considered treatments significantly different when $p < 0.05$. All variables were evaluated for normality and homogeneity of variance using Shapiro–Wilk and Levene test statistics, respectively. Mean treatment annual %C gains were also compared for the 2002–2008 period for modeled small tree and shrub values using ANOVA with Tukey's HSD mean comparison.

2.6. Modeling

To facilitate extrapolating the 2008 re-measurement values to all trees and snags < 75 cm dbh and the 2006 shrub cover to 2008 values, we employed a two-step process. The first step involved calculating the annualized rate of carbon sequestered by small trees and shrubs and the rate of carbon accumulation in the dead tree pool. We randomly selected half of the immediate and 7-year post-treatment small tree and coarse root carbon values, and immediate and 5-year post-treatment shrub carbon values (sampled in 2006) to calculate the annualized rate of carbon accumulation as:

$$\text{rMgC}_{\text{yr}} = \left(\frac{\text{MgC}_{t+n}}{\text{MgC}_t} \right)^{1/n} - 1 \quad (1)$$

where the annualized rate of carbon accumulation for an individual tree (rMgC_{yr}) is equivalent to individual tree carbon in 2008 (MgC_{t+n}) divided by individual tree carbon in 2002 (MgC_t) raised to one divided by the number of years between measurements. We

raise the dividend value by $1/n$ to obtain the geometric mean of annual carbon accumulation per tree. We then subtract one from this value to obtain the difference. We averaged individual tree values of rMgC_{yr} by treatment to obtain a mean rMgC_{yr} value that represents the annualized rate of change in MgC on a per hectare basis.

We made a similar calculation for shrub C. However, instead of calculating the annualized rate of carbon accumulation on a per shrub basis we calculated the MgC ha^{-1} in 2002 and in 2006 (in Eq. (1), MgC_t and MgC_{t+n} , respectively).

We made two calculations to obtain mortality rates for small trees only, since we had captured large tree mortality in the complete large tree re-measurement. To obtain initial post-treatment mortality rates we calculated the percentage of tree C that was in live trees in 2002 that had died by 2003 using the sub-sampled small trees. To calculate the annualized mortality rate of small trees over the 2003–2008 period, we used the same structure as Eq. (1) and changed the inputs to $\text{MgC}_{t+n} \text{ ha}^{-1}$ and $\text{MgC}_t \text{ ha}^{-1}$ of dead tree C.

The average annual rates of carbon accumulation (rMgC_{yr}) were then used to calculate the carbon pool size for small live trees, their coarse roots, and shrubs using a 1-year time-step as:

$$\text{MgC}_{t+1} = [\text{MgC}_t + (\text{MgC}_t \cdot \text{rMgC}_{\text{yr}})] - (\text{MgC}_t \cdot \text{rMgC}_{\text{mort}}) \quad (2)$$

where the per hectare carbon stock value for each pool in a given year (MgC_{t+1}) is equivalent to the per hectare carbon stock value in the previous year (MgC_t) plus the per hectare carbon stock value in the previous year multiplied by the average annualized rate of C accumulation from Eq. (1) (rMgC_{yr}), minus the per hectare carbon stock value in the previous year multiplied by the annualized mortality rate ($\text{rMgC}_{\text{mort}}$).

Dead tree carbon pool size is calculated as the dead tree carbon stock at time t plus the dead tree carbon subtracted from the live tree pool ($\text{MgC}_t \cdot \text{rMgC}_{\text{mort}}$). The dead tree pool increases as a function of the amount of live tree C that is lost to mortality. To estimate carbon loss from the dead pool due to decomposition, we assume a loss rate of 5% per year from the dead C pool as estimated from Harmon et al. (1987).

Emissions estimates used to quantify the carbon cost of each treatment are from North et al. (2009). Emissions varied by treatment intensity and included prescribed fire, harvesting equipment, hauling emissions, and milling waste. Total emissions ranged from 14.8 MgC ha^{-1} in the burn only treatment to 67.9 MgC ha^{-1} in the overstory thin and burn treatment (North et al., 2009).

To calculate the net change in the carbon stock for each treatment, we subtracted the treatment specific emissions from the carbon that accumulated over the 7-year period in large and small live trees and shrubs. We included these three pools in the calculation because live trees and shrubs will continue to grow and sequester carbon and only including live pools provides a conservative estimate of the payback period. This approach, accounting for carbon in live biomass, is consistent across the three primary protocols in the U.S. (Chicago Climate Exchange, Climate Action Reserve, Voluntary Carbon Standard). The protocols, however, vary in how they treat dead wood. We did not include carbon in dead trees or surface fuels because these pools are highly transient (Harmon et al., 1987) and will not contribute to long-term carbon recovery and storage.

2.7. Model validation

To validate the tree carbon accumulation model components, we used the half of the data withheld from model parameterization and calculated the MgC carbon stock for each component group of each treatment unit using Eq. (2) with a 1-year time-step for each year from 2002 through the 2008 re-measurement

period. We used the same process with the shrubs, with the exception that we modeled growth through 2006, the last year the shrubs were re-measured. We then used a paired *t*-test for a two-tailed distribution to compare modeled mean treatment unit Mg C values with measured 2008 (small tree) and 2006 (shrub) values. To estimate shrub C in 2008, we used the average annual growth rates from the 2002–2006 period for each treatment and applied them to the 2006 field measured shrub cover values using Eq.(2).

3. Results

3.1. Field measured carbon stocks

Based on our complete sample of large trees, the understory thin treatment had the largest percent C gain (9.41%) and also had the largest live tree C stock in 2008 (201.7 Mg C ha⁻¹) for large trees (Table 1). The overstory thin and burn also had a higher percent C gain (8.77%), though not significantly different from the control (7.94%) and overstory thin (7.13%) for large trees. Both the burn only and understory thin/burn had substantially lower percent carbon gains (5.08% and 4.82%, respectively) in large trees. Large tree carbon stock levels reflected the difference in treatment intensity. The burn only and both understory thins did not remove any trees > 75 cm dbh, while the overstory thin left only 22 large trees ha⁻¹. The percent mortality of large trees immediately after treatment was lower (0.53%) for the thin only treatments compared to treatments that included burning (average of 2.81%) (Table 1). This pattern was mirrored in the percent C gained in snags from 2002 to 2008, where unburned treatments averaged a 17.7% increase compared to a 50.7% increase in burned treatments (Table 1). These rates are cumulative percentage gains over the 7-year period because in most cases we do not know in which year a tree died. Most of the large tree mortality, however, occurred shortly after treatment because large tree mortality over the 2003–2008 period was too low to quantify.

Litter and duff carbon was greatest in the treatments that did not include prescribed burning and at 26.6 Mg C ha⁻¹, litter and duff carbon in the control was significantly greater than the burn only (14.1 Mg C ha⁻¹) and the overstory thin/burn (6.9 Mg C ha⁻¹) (Table 2). Fine woody debris carbon was significantly greater in the overstory thin and understory thin/burn treatments (Table 2). Coarse woody debris carbon did not significantly differ, but tended to be highest in the treatments that did not include burning (Table 2). Fine root carbon was greatest in the control and decreased with increasing treatment intensity (Table 2). While not significantly different, soil C was greatest in the understory thin (56.3 Mg C ha⁻¹) and lowest in the control (45.5 Mg C ha⁻¹) (Table 2).

3.2. Modeled small tree and shrub carbon stocks

Model validation results indicated that the model outputs were not significantly different ($p > 0.1$) from measured values. Shrub carbon accumulation rates, calculated using Eq. (1), ranged from 0.91% yr⁻¹ in the control to 35.9% yr⁻¹ in the overstory thin/burn treatment (Table 3). Small tree carbon accumulation rates ranged from 2.3% yr⁻¹ for the burn only treatment to 6.4% yr⁻¹ for the understory thin/burn treatment (Table 3). Small tree initial mortality rates varied from 1.85% in the overstory thin to 40.66% in the overstory thin/burn in the first post-treatment year. Following the immediate (2002) post-treatment mortality, small tree mortality rates decreased considerably, ranging from 0.38% yr⁻¹ in the understory thin to 3.99% yr⁻¹ in the understory thin and burn over the 2003–2008 period.

Seven years post-treatment, the control continued to have the largest carbon stock (357.4 Mg C ha⁻¹). Carbon accumulation in the burn only and understory thin treatments was large enough for these treatments to sequester more carbon than was emitted or removed from treatment implementation (53.6 and 13.9 Mg C ha⁻¹, respectively). The understory thin/burn, overstory thin, and overstory thin/burn treatments continued to have net negative C stocks, -12.8, -23.5, and -39.2 Mg C ha⁻¹, respectively, when treatment removals and emissions were subtracted from the 7-year carbon stock gains. Treatment mean C stocks tended to group by treatment intensity, where the control and burn only were not significantly different, the two understory thin treatments were not significantly different, and the two overstory thin treatments were not significantly different.

4. Discussion

Wildfire emissions in the US are substantial and wildfire poses a reversal risk to carbon sequestration in dry temperate forests (Wiedinmyer and Neff, 2007; Galik and Jackson, 2009; Wiedinmyer and Hurteau, 2010). This risk can be dealt with economically by devaluing carbon offsets based on their risk of loss due to wildfire or by allowing the marketplace to determine value as a function of offset effects on compliance costs (Hurteau et al., 2009a; Mignone et al., 2009). For carbon registries, however, reversal risks endanger system integrity. If an offset is sold in the marketplace and then reversed due to disturbance, that climate change mitigation benefit must be replaced. Efforts to mitigate this risk have resulted in the development of "buffer pools", which are an insurance scheme that project developers must contribute to as a function of their project specific reversal risk (Voluntary Carbon Standard, 2007; Climate Action Reserve, 2009). This equates to lost carbon offset revenue for the project owner. Under the Climate Action Reserve's Forest Project Protocol Version 3.1 (2009), the project owner can mitigate reversal risk from wildfire by implementing fuels reduction treatments, thus reducing the size of their contribution to the buffer pool. However, fuels reduction treatments result in initial carbon stock reductions and produce emissions from mechanical thinning and prescribed burning (Finkral and Evans, 2008; Hurteau et al., 2008; Campbell et al., 2009; North et al., 2009).

4.1. Fuels treatment carbon costs

In our previous research we found that emissions associated with fuels reduction treatments increased with increasing treatment intensity. While prescribed fire is beneficial for managing surface fuels, additional high-severity fire risk mitigation is obtained by increasing crowning and torching indices and reducing crown bulk density (Stephens and Moghaddas, 2005), which requires a live tree carbon stock reduction. The initial carbon stock reduction makes the forest stand more resistant to high-severity wildfire, resulting in lower emissions and reduced tree mortality when wildfire does occur (Hurteau and North, 2009; North et al., 2009). In the absence of wildfire or if wildfire emissions are lower than the carbon stock reduction necessary to mitigate high-severity fire risk, fuels treatments could have a net negative impact on carbon stocks and thus reduce the forest's potential to mitigate climate change (Mitchell et al., 2009).

The carbon costs of fuels reduction treatments depend in part on the fate of the carbon removed from the forest (Finkral and Evans, 2008). In our initial quantification of carbon implications from fuels treatments at this site we found that 6.4–19.4% of the carbon on-site ended up in lumber. However, the emissions associated with mechanical thinning, hauling the merchantable wood to the mill, and milling waste ranged from approximately 20.1–41.2 Mg C ha⁻¹.

Table 1
Carbon (Mg ha⁻¹) in 2002 (immediately post-treatment) and 2008 for large trees and snags. Immediate mortality is the percentage of large trees that died within 2 years of the treatments. The mean percent carbon gain for trees and snags is the percent change in the carbon stock over the 7-year period. Values in the same column with different letters are significantly different ($p \leq 0.05$).

Treatment	Large (dbh > 75 cm) trees				Large snags		
	Immediate post-treatment mortality (%)	2002 Mg C ha ⁻¹	2008 Mg C ha ⁻¹	%C gain	2002 Mg C ha ⁻¹	2008 Mg C ha ⁻¹	%C gain
Control	na	155.5	167.7	7.94ab	47.2	55.4	19.08b
Understory thin	0.53	184.5	201.7	9.41a	45.4	55.1	21.50b
Overstory thin	0.53	83.4	89.2	7.13ab	31.7	35.3	12.73b
Burn only	4.04	135.7	142.4	5.08b	27.7	41.8	51.64a
Understory thin/burn	2.01	136.7	143.3	4.82b	34.1	52.4	53.62a
Overstory thin/burn	2.38	47.2	51.1	8.77a	20.3	28.9	47.10a

Table 2
Mean 2008 carbon stock (Mg C ha⁻¹) with standard errors in parentheses by treatment for different pools. Values in the same column with different letters are significantly different ($p \leq 0.05$).

Treatment	Litter/duff	FWD	CWD	Fine roots	Soil (0–30 cm)
Control	26.65 (20.1)a	5.55 (0.9)ab	17.01 (34.9)a	1.17 (0.03)a	45.53 (52.1)a
Understory thin	19.97 (7.7)ab	4.76 (0.6)ab	10.30 (5.3)a	1.09 (0.01)ab	51.69 (29.5)a
Overstory thin	24.33 (5.3)ab	7.34 (0.01)a	12.01 (55.1)a	0.88 (0.13)abc	51.65 (18.3)a
Burn only	14.10 (15.4)bc	5.65 (1.0)ab	7.93 (15.6)a	0.50 (0.0005)bc	47.51 (13.0)a
Understory thin/burn	18.87 (11.7)ab	7.72 (2.2)a	16.80 (6.9)a	0.70 (0.01)abc	56.33 (2.6)a
Overstory thin/burn	6.96 (0.7)c	3.64 (0.08)b	8.42 (3.8)a	0.24 (0.0005)c	48.48 (0.6)a

Table 3
Carbon (Mg ha⁻¹) in 2002 (immediately post-treatment), 2008, and the annual percent change in carbon for shrubs and small (dbh < 75 cm) trees. Immediate mortality is the percentage of small trees that died within 2 years of the treatments. Mortality 2003–2008 is the annual percent mortality rate over the 6-year period. Small tree and shrub values in 2002 were measured. All 2008 values were modeled using Eq. (2). Values in the same column with different letters are significantly different ($p \leq 0.05$).

Treatment	Shrubs			Small (<75 cm dbh) trees			Small (<75 cm dbh) snags		
	2002 Mg C ha ⁻¹	2008 Mg C ha ⁻¹	%C gain	2002 Mg C ha ⁻¹	2008 Mg C ha ⁻¹	%C gain	Immediate mortality (%)	Mortality 2003–2008 (%)	2008 Mg C ha ⁻¹
Control	0.013	0.014	0.91c	81.69	98.39	3.18b		0.72	6.25
Understory Thin	0.005	0.008	7.84c	13.47	20.96	5.60a	11.11	0.38	1.93
Overstory thin	0.002	0.008	24.23ab	18.74	25.42	5.00a	1.85	0.47	0.78
Burn only	0.011	0.012	1.05c	69.58	95.22	2.30b	17.29	0.96	18.29
Understory thin/burn	0.009	0.018	10.65bc	7.63	16.78	6.40a	36.64	3.99	6.45
Overstory thin/burn	0.002	0.017	35.93a	12.19	23.29	2.40b	40.66	0.53	8.67

Prescribed fire resulted in another 14.7–27.2 Mg C ha⁻¹ of emissions (North et al., 2009). While the emissions associated with milling waste (18.3–38.2 Mg C ha⁻¹) could be used to offset fossil-fuel derived energy (Richter et al., 2009), the other emissions are essentially a debit against the climate change mitigation benefit of a forest stand treated for wildfire risk mitigation.

4.2. Post-treatment carbon recovery

Determining the time necessary to recapture the carbon emitted and removed from treatment can help identify the trade-offs between immediate carbon stock reductions and longer-term wildfire risk mitigation. Previous research has shown that reducing stem density results in increased growth in the leave trees (Latham and Tappeiner, 2002; Sala et al., 2005; Skov et al., 2005). While direct measurement comparisons of total live tree C were not possible because we only sub-sampled small trees (<75 cm dbh), we were able to make direct treatment comparisons of large tree (≥ 75 cm dbh) C. We found that the understory thin had the largest percent change in live tree C stock, closely followed by the overstory thin and burn. Both of these treatments exceeded the percent change in live tree C in the control's large trees. Both overstory thin treatments had lower large tree C stocks than the other treatments. These findings are not unexpected given that no large trees were removed in the control, burn only, and both understory thin treatments and large trees make up a substantial proportion of the

aboveground carbon. Total live and dead tree and shrub carbon stocks were substantially lower in the two overstory treatments.

Immediate post-treatment mortality rates were highest in the treatments that included prescribed burning, suggesting that in an effort to minimize live tree C loss it may be beneficial to factor fire-related mortality into marking guidelines. Both small and large tree mortality dropped considerably after the initial post-treatment period, suggesting that the initial effects of the burn treatment are short lived. This decrease in mortality also allowed live tree C stocks to begin recovering. Combining the field measurements and model results, we found the control continued to have the largest total tree-based C stock, which was not significantly different from the burn only total tree-based C stock (Fig. 1). However, 7 years after treatment, total tree-based C stocks in the overstory thinning treatments, even with high annual large tree carbon gains (Table 1), were still significantly lower than treatments that involved no thinning or understory thinning.

4.3. Model uncertainty and error

Our modeled Mg C ha⁻¹ values should be viewed in the context of the assumptions made in the model. Since we only had C stock values at two points in time for each carbon pool, we had to assume that the change in C stocks occurred consistently from year-to-year, as was calculated using Eq. (1). While it is unlikely that growth was consistent from year-to-year, for the 240 trees in

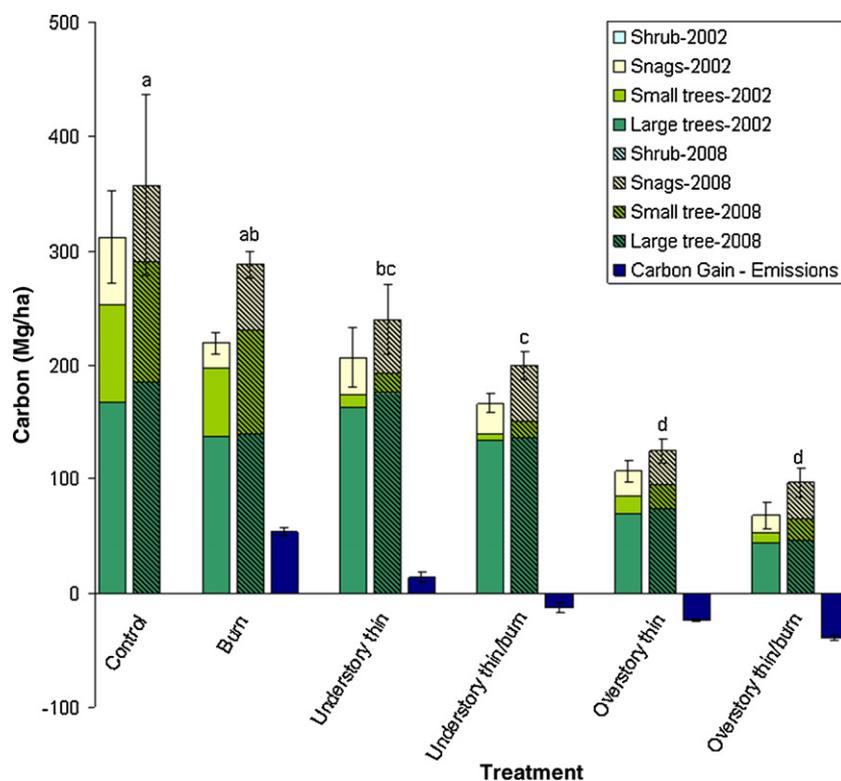


Fig. 1. Mean and standard error for the carbon stock including large trees, small trees, snags, and shrubs immediately (2002, colored bars) and 7 years (2008, colored and hashed bars) after treatments, and 7-year carbon stock gains minus treatment related removals and emissions (blue) in Mg C ha^{-1} . Using an ANOVA with Tukey's HSD mean separation, 2008 treatment means with different letters are significantly different ($p < 0.05$). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

our sub-sample, *t*-test comparisons of modeled 2008 C stock values did not significantly differ from field measured 2008 C stock values. The biggest uncertainty in annual growth rate likely lies in the shrub C pool. However, model outputs for 2006 (the last shrub measurement period) were not significantly different from measured values. Our 2008 model outputs for the overstory thin/burn treatment had the lowest total tree-based carbon stock ($96.8 \text{ Mg C ha}^{-1}$) and the highest shrub growth rate (35.93%), resulting in a shrub C stock of $0.017 \text{ Mg C ha}^{-1}$. As a result, even if shrub growth decreased by a substantial fraction for the 2 years following 2006, the change would not substantially affect total tree and shrub carbon stocks because the shrub C pool only accounts for 0.018% of the C pool.

The results of the surface fuel, soil, and fine root measurements conducted during this re-measurement differ from measurements reported in North et al. (2009). In this study, litter and duff were aggregated, which resulted in higher carbon values than those reported for litter alone in North et al. (2009). Estimates of coarse woody debris (CWD) were lower in some treatments because in this study we used fuels transects to quantify CWD, whereas North et al. (2009) used a full CWD inventory. Discrepancies in the soil carbon stock are likely explained by the variability that is inherent in soils, coupled with the sampling technique employed in both measurement periods. It is important to note that while there is some disparity between estimates of surface fuel carbon between this study and earlier estimates made by North et al. (2009), this carbon pool is relatively transient and is not considered in our estimates of the treatment payback period. Additionally, we did not inventory black carbon in this study. Black carbon forms from the incomplete combustion of biomass and is a relatively stable form of carbon. DeLuca and Aplet (2008) estimate that in thin and burn treatments this could result in the formation of $0.17\text{--}1.7 \text{ Mg C ha}^{-1}$ of black carbon.

4.4. Weighing carbon costs and benefits

The results of this study suggest that when treatment intensity is low, such as in the burn only or understory thin, continued tree growth can re-sequester the carbon removed and emitted during treatment in the relatively short period of 7 years (Fig. 1). Large tree carbon accumulation alone in the understory thin treatment sequestered approximately 85% of the C removed from thinning over the 7-year period. When burning is combined with understory thinning, the payback period for the treatment is extended. In 2008 the understory thin/burn treatment still had a mean carbon deficit of $12.8 \text{ Mg C ha}^{-1}$. However, over the 7-year period tree C increased by an average of $10.8 \text{ Mg C ha}^{-1}$, suggesting that tree growth alone could re-sequester the remaining carbon deficit in as few as nine more years. This timeline most likely represents an upper bound of the length of time necessary to sequester the remaining C deficit from treatment since large tree carbon accumulation rates were lower over this 7-year period in the understory thin/burn treatment than they were in the understory thin and control. The higher intensity overstory thinning treatments will require longer periods of time before they sequester the carbon removed and emitted during treatment. By removing large overstory trees that often contain >65% of the aboveground carbon, these treatments incur a substantial immediate carbon stock reduction and require a much longer recovery period. While large tree percent changes in C were high in these treatments (Table 1), there are simply fewer large trees ha^{-1} to store C. In earlier research we also found overstory thinning did not substantially decrease the risk of high-severity fire compared to understory thinning treatments (North et al., 2009). The understory thinning treatment at Teakettle removed trees $\leq 75 \text{ cm dbh}$, approximating principles employed in restoration treatments (Fiedler and Keegan, 2003). While thinning treatments likely result

in a permanent reduction in the live tree carbon stock, as compared to the control, moderate thinning is effective at altering fire severity and thus the potential loss of carbon from wildfire (Hurteau and North, 2009; North et al., 2009).

The time period required to re-sequester the carbon removed during treatment in the remaining trees will largely depend on longer-term growth rates, which are in part a function of residual density. Several of these modeled C values (burn only, understory thin and burn, overstory thin) likely represent conservative estimates because the percent change in large tree carbon was lower than the control. Previous research on tree removal suggests that growth release in large leaf trees may be delayed anywhere between 5 and 20 years after thinning (North et al., 1996; Latham and Tappeiner, 2002; McDowell et al., 2006; Fajardo et al., 2007).

Does the effective longevity of fuels reduction treatments outlast the carbon recovery period? One approach modeled by Mitchell et al. (2009) was to employ repeated mechanical entries to remove regeneration and maintain a stand structure that is resistant to high-severity wildfire. Because we only sub-sampled small trees during the 7-year post-treatment measurement period, we were unable to quantify the torching and crowning indices. However, immediately following treatment at this site, the torching index was increased by an average of 15.7 and 29.25 km h⁻¹ compared with the control in the understory and overstory thinning treatments, respectively. The crowning index was increased by 9.25 and 19.7 km h⁻¹ compared with the control in the understory and overstory thinning treatments, respectively (North et al., 2009). These immediate reductions in crowning and torching indices are consistent with other studies (Stephens, 1998; Fulé et al., 2001; Fiedler and Keegan, 2003). After an initial mechanical thinning, our work suggests that the open, fire-resistant and resilient structure of these forests can be maintained by repeated prescribed fire application at a frequency that is within the range of the historic mean fire return interval (Hurteau and North, 2009). The prescribed fire frequency that is necessary to manage surface fuels will be due in large part to fuel deposition rates, which are a function of stem diameter (van Wagtenonk and Moore, 2010). Regardless of the method employed, repeated thinning or prescribed burning, some form of continued management will be necessary to maintain a high-severity fire-resistant structure and these treatments will continue to result in a carbon penalty. Prescribed burning does emit carbon and other trace gases, some of which are greenhouse gases. We did not quantify all trace gas emissions, however carbon-based species comprise approximately 97% of biomass burning emissions by mass in conifer forest (Wiedinmyer et al., 2006). While there is a carbon cost to prescribed burning, our results suggest that total C emissions from prescribed fire can be sequestered by tree and shrub growth within a period of time that is shorter than the historic mean fire return interval. As such the wildfire risk reduction and ecological benefits of prescribed fire can offset the climate change mitigation debit. The carbon costs of these treatments should be considered in the context of the larger carbon stock that results from taking no management action to reduce fire risk. Further research is needed to examine the carbon costs and benefits of deploying fuels reduction treatments across the landscape and the relative carbon trade-offs of different treatment placement strategies such as strategically placed area treatments (SPLATs) and defensible fuels profile zones (DFPZs).

In these dry, fire-prone forests, managing for the theoretical maximum carbon stock could increase the reversal risk, especially given the projected increase in large fires under changing climatic conditions (Westerling and Bryant, 2008). Keith et al. (2009) suggest that the carbon carrying capacity of a forest, the maximum amount of carbon that can be stored with a natural disturbance regime, can be used as a baseline for comparing current carbon stocks. In dry temperate forests that have high stand-replacing

wildfire potential, reducing stem density and aggregating carbon in larger, fire-resistant trees can allow for the restoration of fire as a disturbance process that maintains carbon stocks at levels within the carbon carrying capacity of the forest.

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