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The carbon balance of reducing wildfire risk and restoring process: an analysis of 10-year post-treatment carbon dynamics in a mixed-conifer forest

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Abstract Forests sequester carbon from the atmosphere, helping mitigate climate change. In fire-prone forests, burn events result in direct and indirect emissions of carbon. High fire-induced tree mortality can cause a transition from a carbon sink to source, but thinning and prescribed burning can reduce fire severity and carbon loss when wildfire occurs. However, treatment implementation requires carbon removal and emissions to reduce high-severity fire risk. The carbon removed and emitted during treatment may be resequestered by subsequent tree growth, although there is much uncertainty regarding the length of time required. To assess the long-term carbon dynamics of thinning and burning treatments, we quantified the 10-year post-treatment carbon stocks and 10-year net biome productivity (NBP) from a full-factorial experiment involving three levels of thinning and two levels of burning in a mixed-conifer forest in California's Sierra Nevada. Our results indicate that (1) the understory thin treatment, that retained large trees, quickly recovered the initial carbon emissions (NBP=31.4

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±4.2 Mg C ha^{-1}), (2) the carbon emitted from prescribed fire in the burn-only treatment was resequestered within the historical fire return interval (NBP=32.8±3.5 Mg C ha^{-1}), and (3) the most effective treatment for reducing fire risk, understory thin and burn, had negative NBP (−6.0±4.5 Mg C ha^{-1}) because of post-fire large tree mortality. Understory thinning and prescribed burning can help stabilize forest carbon and restore ecosystem resilience, but this requires additional emissions beyond only thinning or only burning. Retaining additional mid-sized trees may reduce the carbon impacts of understory thinning and burning.

1 Introduction

Forests store substantial amounts of carbon (C) globally and provide approximately 2.4 Pg C yr^{-1} of climate-regulating benefit (Canadell and Raupach 2008; Pan et al. 2011). Yet, disturbances can negatively impact this ecosystem service over large geographic areas, resulting in a transition from C sink to source for years to decades within the disturbed area (Campbell et al. 2007; Dore et al. 2008; Kurz et al. 2008). Many terrestrial systems have evolved with fire, a globally distributed disturbance, the frequency of which varies across biomes and through time (Bowman et al. 2009). Worldwide, biomass burning is a significant contributor to total C emissions and has garnered increasing attention because of its role in climate forcing (Kloster et al. 2012). Average annual global emissions from biomass burning were 2.0 Pg C year^{-1} from 1997 to 2009, with forest fires contributing approximately 15 % of total C emissions (van der Werf et al. 2010). While climate change is a global issue, forest-based C sequestration requires local ecological knowledge, including an understanding of disturbance dynamics (Hurteau et al. 2013b).

In the western United States wildfire remained in dynamic equilibrium with climate from 500 CE to the 1800s (Marlon et al. 2012). During the 1800s, biomass burning increased with European settlement, causing a peak in fire activity (Marlon et al. 2012). By the 20th century, forest recovery following logging and human factors (i.e., grazing, fire suppression) led to a decline in area burned, causing substantial ecological changes (Covington and Moore 1994; Marlon et al. 2012). Since the early 1900s, when federal fire suppression policy was enacted, stem density of shade-tolerant, fire-intolerant species has increased and surface fuels have accumulated in many dry forest types (Stephens and Ruth 2005; Scholl and Taylor 2010). This recent decrease in fire activity coupled with changing climate has altered fire type and size. In the most recent decades large wildfire frequency and wildfire severity have increased in much of the western US (Westerling et al. 2006; Miller et al. 2009). In the Sierra Nevada mountains of California, mixed-conifer forests developed with low to moderate severity fires approximately every 11–30 years (Taylor and Skinner 2003; North et al. 2005; Van de Water and Safford 2011). Fire-exclusion and associated changes in forest structure and composition have created forests that are more susceptible to stand-replacing wildfires (Miller et al. 2009) that are beyond the historical range of variability. The increase in high-severity fire events can result in mortality of much of the overstory, triggering changes in forest structure and composition and converting forests from a C sink to a C source (Stephens and Moghaddas 2005; Dore et al. 2008; Westerling and Bryant 2008; Hurteau and North 2009).

In these fire-adapted ecosystems, treatments that reduce tree density and surface fuels are a common management practice used to lower the risk of stand-replacing fire. Thinning and prescribed burning reduces fuel quantity and fuel continuity (Agee and Skinner 2005) and can also meet additional management objectives, including restoration of native species (Wayman and North 2007; Laughlin and Fule 2008), protection from insect and pathogen outbreaks (Campbell et al. 2012), and increased tree regeneration (Zald et al. 2008). However, implementing these treatments may also have undesired ecological consequences, such as creating favorable conditions for invasive plant establishment (Keeley 2006; Collins et al. 2007). Because treatment implementation causes C removal (thinning) and C emissions (prescribed fire), the dynamics of the tradeoff between these C losses during treatment and C protection from reduced wildfire emissions has been the focus of recent debate. Some studies have found that treatments that reduce wildfire severity yield a net C benefit when wildfire occurs (Hurteau et al. 2008; Hurteau and North 2009; Stephens et al. 2009; North and Hurteau 2011), while others suggest that, over an entire disturbance cycle, long-term C storage of a treated forest can be similar or considerably lower than a forest that is untreated and subject to a stand-replacing fire, depending on forest type (Mitchell et al. 2009; Campbell et al. 2012). Largely absent from this discussion have been the effects of thinning and burning on long-term post-treatment carbon sequestration from continued tree growth and the species-specific effects of prescribed burning as a function of fire sensitivity. Understanding local ecological conditions following fuel reduction treatments may provide insight into the capacity of post-treatment forest growth to recover the C debt incurred during treatment.

The objective of our research was to quantify forest C dynamics in a Sierra Nevada mixed-conifer forest over the 10-years following fuel reduction treatments. We hypothesized that a range of C balance outcomes would exist 10-years following treatments. We predicted 1) post-treatment tree growth would sequester more C than was emitted from prescribed burning treatments; 2) understory thinning treatments with and without prescribed burning would recover the C removed and emitted during treatment within the 10 year period, and overstory thinning treatments would continue to have a C debt; and 3) fire-tolerant species (e.g., *Pinus spp.*) would have increased C gain relative to fire-intolerant species (e.g., *Abies spp.*) in treatments that included burning.

2 Methods

2.1 Study site

The Teakettle Experimental Forest is a 1300 ha reserve of old-growth, mixed-conifer forest located approximately 80 km east of Fresno, CA in the Sierra Nevada. The elevation ranges from 1900 to 2600 m, with an average annual precipitation of 125 cm, falling almost exclusively as snow in this Mediterranean climate (North et al. 2002). Dominant species in the mixed-conifer forest include *Abies concolor, A. magnifica, Calocedrus decurrens, Pinus jeffreyi,* and *P. lambertiana.* The Teakettle experiment, located in the Experimental Forest, was established to examine the ecological effects of a range of structural manipulations and prescribed burning. The experiment utilized a full factorial design, crossing three levels of thinning (no thin, understory-thin, overstory-thin) with two levels of burning (no burn, prescribed fire). Three replicates of each treatment were established using four-hectare treatment units. Within each treatment unit, 9–49 monumented grid points were established for sampling understory vegetation and surface fuels. The understory-thin treatment removed all trees 25–75 cm diameter at breast height (DBH). The understory-thin was initially designed to reduce impacts on California spotted owl (*Strix occidentalis occidentalis*) habitat, although the guidelines now have been primarily used to reduce stand-replacing fire risk in Sierra mixed-conifer forests. The overstory treatment removed all trees greater than 25 cm DBH, with the exception of 22 large diameter trees ha^{-1}. The thin and burn plots were mechanically treated in 2000 and burned in 2001. The thin-only plots were treated in 2001. Prescribed fires were implemented during fall 2001 (North et al. 2002).

2.2 Data collection

Prior to treatment, all trees and standing snags ≥5 cm DBH were measured, mapped (using a surveyor's total station) and permanently tagged. Measurements were consistent across measurement periods (pre-, immediate post-, and 10-years post-treatment) with one exception; during the 2011 re-measurement, logs ≥30 cm were not remapped and measured. Instead, all coarse woody debris (CWD) was measured using the planar-intercept method (Brown 1974). Three 15 m fuels transects were measured at each grid point within each treatment unit. Understory vegetation, soil C and fine roots were also measured at the grid points. A detailed reporting of the pre- and post-treatment methods can be found in the supplemental material and in North et al. (2009a), Wayman and North (2007), and Innes et al. (2006).

2.3 C calculations

We quantified C from the 10-year post treatment re-measurement (2011 data collection). We used genus-specific allometric equations from Jenkins et al. (2003) to calculate tree and snag biomass. Coarse and fine woody debris biomass was calculated following Brown (1974), and C concentration for CWD varied by decay class following Harmon et al. (1987). The C in litter and duff was quantified assuming a C concentration of 37 % (Smith and Heath 2002). We quantified C in shrubs using a site-specific relationship between percent cover and biomass (Hurteau and North 2008) and assumed a shrub C concentration of 49 % following Campbell et al. (2009). Post-treatment net biome productivity (NBP) was calculated by subtracting treatment related carbon emissions from the change in C stocks over the post-treatment period. Emissions from equipment and log hauling were estimated using trip distance and equipment time on plot. Prescribed fire emissions were calculated using pre- and immediate post-treatment fuel C stock values and a mass-balance approach. Emissions from wood processing were assumed to be 40 % of the tree C delivered to the mill (see Supplemental Material and North et al. (2009a)). We counted the C converted to wood products as sequestered from the atmosphere.

2.4 Analyses

Carbon values were scaled to a per hectare basis for treatment comparison. Because our hypotheses were based on post-treatment C dynamics, we used 2002 (immediately post-treatment) and 2011 (10-years post-treatment) C values for analysis. We used ANCOVA and Tukey's HSD post hoc analysis to determine if there were significant ($p<0.05$) differences between treatments. Each variable was evaluated for normality and equal variance; all variables met the assumptions of ANCOVA.

3 Results & discussion

3.1 Treatment-level distribution of C

Treating forests to reduce the risk of high-severity wildfire not only requires an immediate reduction in forest C stocks, but also produces emissions from thinning and prescribed burning (Finkral and Evans 2008; Hurteau et al. 2008; Campbell et al. 2009; North et al. 2009a). By employing simulation methods, Campbell et al. (2012) suggested that treatment application is burdened with a large C debt. Conversely, we found that certain low-intensity treatments allow the amount of C emitted and removed during treatments to be re-sequestered within 10-years (Fig. 1).

Total C increased over the 10-year period in all treatments except the control, with the two largest 10-year C increases occurring in the understory-thin (51.5 Mg C ha^{-1}) and burn-only (47.6 Mg C ha^{-1}) treatments (Fig. 1, Table 1). In the understory-thin treatment, the increase in live tree C (47.3 Mg C ha^{-1}) accounted for most of the total stand C increase (Fig. 1, Table 1), while in the burn-only treatment, the increase in live tree C (19 Mg C ha^{-1}) and snag C (20 Mg C ha^{-1}) accounted for the majority of the total stand C increase (Fig. 1, Table 1). Over the post-treatment decade total C in the control remained nearly constant, with decreases in live tree C compensated by increases in snag C (Fig. 1, Table 1). This may be a result of increasing background mortality, which has been documented throughout the western United States in recent decades, with warming and drought stress being the most likely drivers (van Mantgem et al. 2009; van Mantgem et al. 2013).

Of the C removed by thinning, approximately 40 % was emitted as milling waste and 60 % was sequestered as dimensional lumber. Milling waste emissions ranged from 18.3 Mg C ha^{-1} in the understory thin to 38.2 Mg C ha^{-1} in the overstory thin. Prescribed fire emissions ranged from 14.7 Mg C ha^{-1} in the burn-only to 27.2 Mg C ha^{-1} in the overstory-thin and burn (North et al. 2009a, b, Supplementary Table S1). The burn and understory-thin sequestered more C

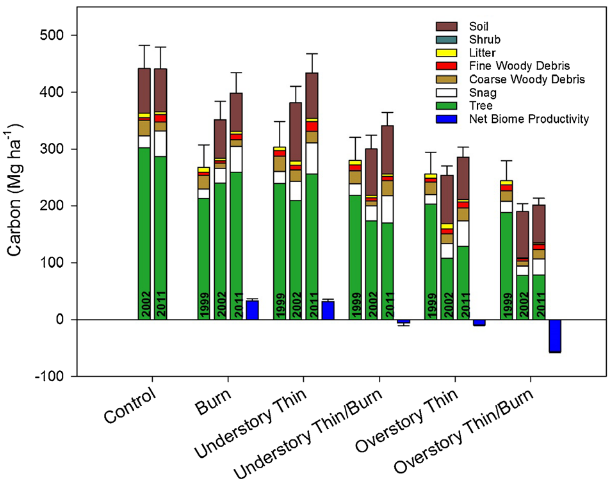

Fig. 1 Mean and standard error of C pools pre-treatment (1999), immediately post-treatment (2002) and 10-years post-treatment (2011) in Mg C ha^{-1}. Ten-year net biome productivity (solid blue bar) is the 10-year net ecosystem productivity minus C removed and emitted during treatment implementation in Mg C ha^{-1}. Soil and shrub C values are not included in the pre-treatment (1999) C stocks

Table 1 The mean carbon stock size (Mg C ha^{-1}) of different pools immediately post-treatment (2002) and 10-years post-treatment (2011) for each of six treatments and the net ecosystem productivity (NEP) over the 10-year post-treatment period. Net biome productivity (NBP) over the 10-year post-treatment period was calculated as NEP minus all treatment related emissions. Standard errors are shown following the plus-minus sign. Values in the same row with different letters indicate a significant difference in carbon pools 10 years post-treatment ($p \leq 0.05$)

	Control		Burn only		Understory-thin	
	2002	2011	2002	2011	2002	2011
Live tree	302.4±40.7	286.5±34.4^{ab}	240.1±10.2	259.1±14.5^{ab}	209.0±26.8	256.3±13.0^{b}
Snag	20.8±3.6	45.6±10.1^{a}	25.2±5.6	45.2±1.2^{a}	34.1±7.3	54.6±11.9^{a}
CWD	27.6±4.8	15.4±1.9^{a}	9.4±0.8	12.1±4.4^{a}	20.3±2.9^{a}	20.3±4.4^{a}
FWD	4.17±0.4	12.7±1.4^{ab}	4.16±0.5	10.2±0.8^{ab}	7.73±1.0	16.6±0.6^{b}
Litter and duff	7.90±0.2	5.19±0.8^{a}	4.70±0.1	4.89±0.9^{a}	7.40±0.2	5.99±0.7^{a}
Soil (30 cm)	78.1±2.9	75.2±10.3^{a}	67.6±3.9	67.3±4.7^{a}	103.0±13.6	79.3±1.3^{a}
Shrub	0.03±0.01	0.02±0.00^{ab}	0.03±0.01	0.02±0.01^{a}	0.01±0.00	0.02±0.01^{ab}
Total	440.9±46.1	440.7±50.1^{a}	351.3±18.9	398.9±15.6^{a}	381.6±50.2	433.1±23.3^{a}
NEP (10-year)		−0.2±20.2		47.6±9.8		51.5±27.4
NBP (10-year)		−0.2±20.2		32.8±3.5		31.4±4.2
Live tree	173.3±9.3	169.5±3.8^{a}	107.7±9.7	128.8±7.5^{ab}	77.6±11.5	78.4±19.1^{a}
Snag	26.4±1.1	48.1±8.4^{a}	25.6±4.8	44.6±17.8^{a}	16.7±0.7	27.9±2.5^{a}
CWD	8.9±1.8	26.6±6.4^{a}	17.6±1.9	22.5±4.9^{a}	8.4±3.3	17.1±3.6^{a}
FWD	5.09±1.2	7.03±2.8^{a}	8.36±0.3	10.9±1.7^{ab}	4.32±0.4	8.19±0.9^{ab}
Litter and duff	4.40±0.2	4.83±1.5^{a}	9.60±0.3	4.08±0.5^{a}	1.60±0.11	2.91±0.5^{a}
Soil (30 cm)	82.0±5.3	84.8±5.8^{a}	85.2±5.7	74.5±4.7^{a}	81.4±3.7	66.7±1.4^{a}
Shrub	0.02±0.01	0.04±0.00^{ab}	0.01±0.00	0.03±0.00^{a}	0.01±0.00	0.06±0.01^{b}
Total	300.2±14.0	340.9±6.6^{a}	254.1±21.4	285.5±32.8^{a}	190.0±18.3	201.3±18.7^{a}
NEP (10-year)		40.7±20.2		31.4±12.1		11.2±0.9
NBP (10-year)		−6.0±4.5		−9.7±0.7		−56.6±2.2

than was removed and emitted during treatment implementation, resulting in a positive NBP of 32.8 and 31.4 Mg C ha^{-1}, respectively (Fig. 1). While prescribed fire does emit C to the atmosphere, the C recovery period for the burn-only treatment is well within the site-specific historical mean fire return interval (17.3 years, (North et al. 2005)). However, in the understory thin and burn, mortality resulted in a large increase in dead tree C and a small decrease in live tree C over the 10-year period, causing a NBP of −6.0 Mg C ha^{-1} (Fig. 1, Table 1). In the understory-thin and burn, the increase in snag C is largely due to the higher mortality in trees >130 cm diameter (Fig. S1d, Fig. 1). This finding is especially relevant today, as thinning from below followed by prescribed burning has become a common management practice for its effectiveness in treating both ladder and surface fuels (Raymond and Peterson 2005; Stephens and Moghaddas 2005). The high mortality of large trees following treatment may result from long-term litter build-up at the base of the tree, increasing the risk of cambial and root injury from smoldering combustion (Swezy and Agee 1991; Fule et al. 2002; Stephens and Finney 2002). The overstory-thin, and overstory-thin and burn continued to have negative NBP of −9.7 and −56.6 Mg C ha^{-1}, respectively (Fig. 1, Table 1).

It is important to note that we evaluated post-treatment C recovery in terms of direct treatment emissions (prescribed fire, equipment, and hauling emissions) and indirect (milling waste) losses. We counted the C stored in long-lived wood products, which ranged from 27.5 to 56.2 Mg C ha^{-1} (North et al. 2009a, supplementary Table S1), as sequestered because of the life span of dimensional lumber (Skog and Nicholson 1998). For the forest to resequester the C stored in wood products and return to pre-treatment C stock levels while maintaining a structure that is resistant to high-severity wildfire will likely take a considerably longer period of time as it requires additional growth in the retained trees, rather than in-growth from regeneration.

Treatments that included burning experienced a greater C increase in CWD than treatments that did not, likely due to CWD addition by fire-induced tree mortality. Treatments that combined thinning and burning had much larger increases in CWD than burn-only and thin-only treatments (Table 1). We did not detect differences in litter and duff C because of within treatment variability (Table 1). However, litter and duff C increased in all burn treatments, and decreased in all treatments that were not burned (Table 1). The litter layer increase in burn treatments is likely due to combined inputs from scorched and fire-killed trees.

Shrub C increased most in treatments that included overstory-thinning or understory-thinning and burning, likely due to increased light availability (Table 1). Shrub C decreased in the control and burn-only treatments, possibly because of continued tree in-growth several years following treatment (Table 1). Soil C did not significantly vary between 2002 and 2011 (Tables 1).

Some of the variability in C pool size between post-treatment measurement periods was unexpected. Our results show a reduction in coarse woody debris (CWD) in the control over the 10-year post-treatment measurement period. This reduction may be due to decomposition of CWD or may be an artifact of the change in sampling methodology from 2002 to the 2011 measurement period. In this latter sampling period we used fuel transects to quantify CWD, whereas North et al. (2009a) used a full CWD inventory. Shrub C had a 2–7 fold increase in all treatments except the control and burn-only. However, given the small contribution to total ecosystem C, the changes in shrub cover likely have much larger ecological effects than effects on C dynamics (Hurteau and North 2008; Hurteau and North 2010). Differences in the soil C stock between sampling periods are likely explained by the variability that is inherent in soils, coupled with the sampling intensity employed in both measurement periods. More intensive sampling for soil C would likely provide a more accurate representation of treatment effects.

3.2 Live tree distribution of C

Prior to treatment, aboveground live tree C in the six treatments ranged from 188.0 to 249.8 (se±3.82) Mg C ha^{-1} (North et al. 2009a). The overall mean of this pre-treatment C pool was similar to that of the 2011 aboveground portion of the live tree C in the control (236.5 Mg C ha^{-1}). In 2011, live trees >130 cm DBH continued to account for the greatest amount of live tree C across all treatments (Fig. S1). Live tree C in the largest diameter class increased over the 10-year post-treatment period in the burn-only and overstory-thin, and had a significant increase in the understory-thin ($p<0.05$, Student's *t*-test, Fig. S1). Although not significantly different, the control, understory-thin and burn, and overstory-thin and burn all had decreases in total live tree C in trees greater than 130 cm DBH, leading to a decrease in live tree C for two of the treatments (Fig. 1, S1). The live tree C decline in the control highlights the difficulty in quantifying absolute change in C stocks against a pre-treatment baseline, because the control results suggest that even in the absence of treatment live tree C stocks would have been variable over the post-treatment decade.

3.3 Species-specific C response

Abies concolor accounted for approximately 67 % of the basal area prior to treatment (Meyer et al. 2007) and the largest fraction of live tree C following treatment (Fig. S1). *A. concolor* C increased most in the understory-thin and overstory-thin treatments (Fig. S1). We hypothesized that growth of fire-tolerant species (e.g., *Pinus spp.*) would increase more than fire-intolerant species (e.g., *Abies spp. and C. decurrens*) in treatments that included burning, yet we found no statistically significant differences in C gain between these groups (Fig. S1). In the understory-thin and burn, large (>50 cm DBH) both declined, a result that was largely driven by C reduction in the largest diameter class (>130 cm DBH) (Fig. S1). van Mantgem et al. (2013) found that mortality rates were similar for large (>50 cm DBH) *Pinus* and *Abies* tree species 5-years following prescribed fires.

3.4 Minimizing C losses while restoring process and reducing wildfire risk

Our results demonstrate that post-treatment NBP declines with increasing treatment intensity, with the burn-only and understory-thin being the only treatments with positive 10-year NBP. However, the C outcome needs to be considered in the context of treatment effects on high-severity wildfire risk; thinning without burning fails to reduce surface fuel accumulation and has the potential to produce relatively high wildfire emissions (Stephens et al. 2012). Furthermore, neglecting to restore fire does not promote the ecological benefits associated with low-severity fire in this system (North et al. 2009b; Stephens et al. 2013). While forests do sequester and store C, stabilizing C in frequent-fire forests requires periodic C emissions from a restored fire regime to reduce high-severity wildfire risk (Hurteau and Brooks 2011; Hurteau et al. 2013b). Low-intensity fires have the advantage of reintroducing critical ecological processes (Hurteau et al. 2013a). For instance, prescribed fire that emulates historic conditions prepares seedbeds for germination of conifer species, recycles nutrients, and increases soil water availability, all factors that potentially increase C fixation and creates a mosaic of open and closed habitat for threatened owl species (Kilgore 1973; Sala et al. 2005; Roberts et al. 2011).

To buffer against mortality associated with combined thinning and burning treatments, we recommend retaining trees that can serve as replacements for large individuals lost to treatment-induced mortality. As an example, the understory-thin and burn treatment removed all trees 50–75 cm DBH. Retaining several of these larger individuals per hectare would help ensure that any large tree mortality is compensated more quickly than would occur by relying on a 25 cm DBH individual to double in diameter. Additionally, post-treatment competitive release may accelerate growth of retained midsized individuals into larger diameter classes.

Retaining large, fire resistant trees in a more open structure, coupled with the heterogeneity created by prescribed fire (North et al. 2009b), will promote ecosystem characteristics that have been identified as important for building system-level resilience for both future wildfire and changing climate (Westerling et al. 2006; Millar et al. 2007; Moritz et al. 2013; Stephens et al. 2013). Shade-tolerant, fire-intolerant species (*A.concolor*, *A. magnifica*, *C. deccurrens*) not only create dense patches that have high fire risk, but are also more sensitive to precipitation fluctuations (Hurteau et al. 2007; North et al. 2009b; Collins and Stephens 2010; Earles et al. 2014), suggesting that, as climate becomes more variable, these fire-intolerant species may reduce ecosystem resilience. However, shade-tolerant, fire-intolerant species account for a large fraction of live tree C, suggesting that retention of large (≥90 cm DBH) individuals of these species can continue to contribute to total C stock. Large fire-tolerant pine species also contribute to post-treatment C increases. While the percent C increase of large pines (≥90 cm DBH) was greatest in the understory-thin for *P. jeffreyi* (Fig. S1c) and burn-only treatment for *P. lambertiana* (Fig. S1b), Zald et al. (2008) found that regeneration of these pine species only occurred in the most intensive treatments. From a climate change resilience perspective, large tree retention in a more open structure may serve to facilitate reduced water stress during periods of drought. Recent research found that large trees in thinning and burning treatments in a southwestern ponderosa pine forest were less affected by subsequent drought than were smaller individuals of the same species (Kerhoulas et al. 2013).

Post-treatment C dynamics should be considered in the context of treatment effectiveness and longevity in maintaining reduced high-severity wildfire risk, and the projected increase in large wildfire frequency with changing climate. Treatment efficacy varies as a function treatment intensity, forest type, and site productivity (Stephens et al. 2012). In the Sierra Nevada, treatment longevity ranges from 5 to 20 years (Stephens et al. 2009; Chiono et al. 2012), but repeated burning should help maintain the forest structure and fuels distribution resulting from treatment. Given the increasing frequency of high and extreme fire weather occurrence and the projected increase in large wildfire frequency for the Sierra Nevada (Westerling et al. 2011; Collins 2014), more intensive treatments with higher C costs may be required to maintain treatment efficacy.

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