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Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions

Malcolm North,[1,4] Matthew Hurteau,[2] and James Innes[3]

[1]*USDA Forest Service, Sierra Nevada Research Center, 1731 Research Park Drive, Davis, California 95618 USA*
[2]*National Institute for Climatic Change Research, Western Region, Northern Arizona University, P.O. Box 6077, Flagstaff, Arizona 86011 USA*
[3]*Lolo National Forest, Missoula Ranger District, Fort Missoula Building 24, Missoula, Montana 59804 USA*

Abstract. Depending on management, forests can be an important sink or source of carbon that if released as CO_2 could contribute to global warming. Many forests in the western United States are being treated to reduce fuels, yet the effects of these treatments on forest carbon are not well understood. We compared the immediate effects of fuels treatments on carbon stocks and releases in replicated plots before and after treatment, and against a reconstruction of active-fire stand conditions for the same forest in 1865. Total live-tree carbon was substantially lower in modern fire-suppressed conditions (and all of the treatments) than the same forest under an active-fire regime. Although fire suppression has increased stem density, current forests have fewer very large trees, reducing total live-tree carbon stocks and shifting a higher proportion of those stocks into small-diameter, fire-sensitive trees. Prescribed burning released 14.8 Mg C/ha, with pre-burn thinning increasing the average release by 70% and contributing 21.9–37.5 Mg C/ha in milling waste. Fire suppression may have incurred a double carbon penalty by reducing stocks and contributing to emissions with fuels-treatment activities or inevitable wildfire combustion. All treatments reduced fuels and increased fire resistance, but most of the gains were achieved with understory thinning, with only modest increases in the much heavier overstory thinning. We suggest modifying current treatments to focus on reducing surface fuels, actively thinning the majority of small trees, and removing only fire-sensitive species in the merchantable, intermediate size class. These changes would retain most of the current carbon-pool levels, reduce prescribed burn and potential future wildfire emissions, and favor stand development of large, fire-resistant trees that can better stabilize carbon stocks.

Key words: biomass; carbon-emission costs of different fuels treatments; carbon sequestration; forest management; global warming; management and treatment of different forest fuels; Sierra Nevada (California, USA) mixed-conifer forest; Teakettle Experimental Forest (USA).

Introduction

With fire suppression, many western forests in the United States have high stem densities from decades of infilling with shade-tolerant, fire-sensitive regeneration. Some research suggests an unintentional benefit of this change has been an increase in forest biomass, sequestering carbon that might otherwise contribute to global warming (Houghton et al. 2000, Hurtt et al. 2002). This putative carbon increase, however, poses a problem for land managers because fuel-loaded forests are susceptible to large carbon emissions if they burn in a catastrophic wildfire. In general, mechanical thinning, prescribed fire, or both are often used to reduce fuels, producing an immediate carbon release in an effort to reduce potential future wildfire emissions. Under current, widely followed California Climate Action Registry (CCAR) guidelines, landowners are not penalized for wildfire emissions. Fuels treatments that reduce live-tree biomass are considered a reduction in forest carbon stocks, and landowners are penalized for removing or releasing carbon that could contribute to greenhouse gas emissions. In fire-prone forests, some researchers have suggested these calculations are short sighted because untreated forests may release many times as much carbon if they burn at high-intensity in a wildfire (Hurteau et al. 2008, Hurteau and North 2009). Rapid changes are occurring in political policies, and nascent carbon-trading markets are already occurring (e.g., Chicago Climate Exchange, European Energy Exchange). Yet there are many uncertainties in how management practices affect forest carbon dynamics. Two questions at the core of these uncertainties are: (1) What are the carbon emission costs of different fuel treatments? and (2) How do managers maximize carbon stocks while minimizing catastrophic loss by wildfire?

Current carbon-policy initiatives range from voluntary, multi-state agreements (e.g., Western Climate Initiative, Regional Greenhouse Gas Initiative) to

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[4] E-mail: mpnorth@ucdavis.edu

state-government mandates such as California's 2006 Global Warming Solutions Act (Assembly Bill 32, *available online*)[5] requiring 2020 statewide greenhouse gas emissions to be reduced to1990 levels. A significant concern in these policy initiatives is forest carbon stocks, spurring the CCAR to develop a protocol for forest-offset projects (CCAR 2007). Currently these protocols do not penalize for forest carbon emissions or stock reductions by wildfire. Yet rising atmospheric greenhouse gas concentrations are predicted to influence the climate, having a positive feedback on wildfire. Recent research suggests fire size and fire-season length may already be increasing as the climate warms (Westerling et al. 2006, Westerling and Bryant 2008, Miller et al. 2009). Any management efforts to reduce forest fuels and potential wildfire intensity come at a cost of reduced carbon stocks and increased emissions. While fuel-treatment effectiveness has been the subject of extensive research and controversy (Johnson 2003, Martinson et al. 2003, Agee and Skinner 2005, Odion and Hanson 2006), there has been comparatively little research on different treatment effects on carbon stocks and emissions.

The focus of our research was to compare the effects of different fuels treatments on carbon pools and emissions, forest stand structure, and fire resistance in Sierra Nevada mixed-conifer forest. We also compare the effects of fire suppression on current live-tree carbon stocks against historic conditions when the forest had an active fire regime and was likely more resistant to stand-replacing fire. Our research focused on three questions: (1) Do current fire-suppressed forests have higher live-tree carbon stocks than historic forests exposed to frequent fire? (2) What are the relative carbon emission costs of different fuels treatments and their potential effectiveness at reducing wildfire intensity? and (3) What fuel treatments favor future stand development of higher carbon stocks? We examine fuel-treatment effects on carbon dynamics at the Teakettle Experimental Forest, which, typical of many fire-suppressed western forests, had an active fire regime (12–17 year fire return interval) until the 1860s when all wildfires stopped (North et al. 2005). In an effort to include both biological and industrial components (Gower 2003), we examined changes in carbon stocks in the soil, surface fuels, and live and dead trees, and fossil-fuel use for logging, yarding (moving the logs from point of felling to a central loading zone), and truck equipment because these are emission sources common to many fuels-treatment operations. We included calculations of carbon emissions due to the prescribed burn and how much of the tree carbon ended up in milled lumber and waste for the merchantable-sized trees in each treatment. Finally we compare the stand structure and fire resilience produced by the different fuel treatments as these conditions affect future carbon emissions when the forest inevitably burns.

Our analysis of carbon dynamics is limited to tree and soil-based carbon stocks and does not measure all carbon fluxes as others have done (e.g., Misson et al. 2005, Dore et al. 2008). Our paper is also focused on immediate changes in carbon stocks and emissions, although in previous work we have modeled longer trends in tree-based carbon dynamics due to stand development and future wildfire (Hurteau and North 2009).

Methods

Study area

The study took place within the Teakettle Experimental Forest (see Plate 1), a 1300-ha reserve of old growth within the Sierra National Forest, 80 km east of Fresno, California, USA (information *available online*).[6] The elevation ranges from 1900 to 2600 m, and annual precipitation of ~125 cm falls almost entirely as snow between November and April (North et al. 2002). Our experiment occurred within the mixed-conifer forest type, which characteristically contains white fir (*Abies concolor*), red fir (*A. magnifica*), California black oak (*Quercus kelloggii*), sugar pine (*Pinus lambertiana*), incense-cedar (*Calocedrus decurrens*), and Jeffrey pine (*Pinus jeffreyi*) (Rundel et al. 1988). Fuels treatments were applied to 18 permanent 4-ha plots established using variogram analysis to estimate an area sufficiently large to include the range of variable forest conditions found in mixed conifer. An analysis of the forest structure found no significant pretreatment differences among the 18 plots (North et al. 2002).

Treatments

The 18 plots were assigned to one of six treatments determined by the full- factorial experimental design crossing two levels of burning treatments (prescribed fire and no burn) and three levels of thinning treatments (none, understory, and overstory). The understory prescription followed guidelines in the California Spotted Owl (CASPO) report (Verner et al. 1992), which removes all trees between 25 and 76 cm (10 and 30 inches) diameter at breast height (dbh) while retaining at least 40% canopy cover. Although designed initially for minimizing impact to Spotted Owl habitat, the CASPO guidelines became the standard forest practice in the 1990s and are still widely used as a fuel reduction treatment (SNFPA 2004). The overstory prescription removed all trees >25 cm dbh except for 22 large-diameter trees per hectare, which were left at regular spacing (~20 m apart). The overstory thinning was widely practiced in Sierran forests before CASPO, and leaving trees widely spaced reduces canopy connectivity and bulk density. At Teakettle this marking resulted in a

[5] ⟨http://www.arb.ca.gov/cc/cc.htm⟩

[6] ⟨http://teakettle.ucdavis.edu⟩

Plate 1. Prescribed fire emissions in a burn-only plot in Teakettle Experimental Forest, California, USA. Photo credit: M. North.

prescription of cutting dominant overstory trees up to 100 cm (40 inches) dbh. The thinning treatments were applied in fall of 2000 (thin and burn plots) and early spring of 2001 (thin-only plots). Trees were limbed and topped where they fell, and merchantable logs were removed. Mechanical thinning operations were conducted for 38 days in October and November of 2000 (202 truckloads of logs) in the six plots whose treatment included both thinning and prescribed burning. In June through September of 2001 the remaining six plots (thin-only treatments) were thinned over 51 operating days, resulting in an additional 312 truckloads of logs.

The Sierra National Forest applied the prescribed fire following their standard operating procedures. Fuels from the thinning operations were left to dry for one year, and the prescribed fires were ignited in fall of 2001 a week after the first substantial (2 cm) rainfall. All plots were burned within a one-week period and the fire was extinguished by snow a week later.

Data collection

Using a surveyor's total station (Topcon 313; Topcon America Corporation, Paramus, New Jersey, USA), all trees and snags ($\geq$5 cm dbh; $N = 35\,418$ snags) in the 18, 4-ha plots were measured, identified to species, mapped, and permanently tagged during the 1998–2000 field seasons before treatments were applied. Following treatments, all plots were resampled and mapped during the 2002–2004 field seasons using the same protocols. For the three control plots, we reconstructed stand structure and composition in 1865, immediately after Teakettle's last wildfire (North et al. 2005) when the forest still had an active fire regime. The reconstruction used the current complete inventory of trees, snags and logs, and calculated approximate 1865 diameters using a series of species-specific decay and growth rates (described in detail in North et al. [2007]).

Prior to treatment, either 49 or 9 permanent sample points were established on a 25-m and 50-m grid, respectively, in each plot. The mass of the fine-wood debris (FWD) was estimated pre- and posttreatment (the controls were only sampled once) using the planar intercept method (Brown 1974) where debris is tallied by 1-, 10-, 100-, and 1000-hour fuel-moisture classes. For the 1000-hour 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. CWD, defined here as downed logs $\geq$30 cm in diameter (Maser and Trappe 1984) and $\geq$2 m in length, were mapped and inventoried from 1999 to 2004 (pre- and posttreatment). Log decay was determined using a modification of Maser et al.'s (1979) classification. The volume of each log was estimated as a frustrum paraboloid (Husch et al. 2002) and mass (in Mg/ha) was estimated using the specific gravities of Harmon et al. (1987). Since downed log species were often unidentifiable in the pretreatment survey, we averaged the specific gravities of Harmon et al. (1987) by decay class for the dominant species found at Teakettle.

In 2003 (two years after treatments), three 2-cm-diameter soil cores were taken from the nine sample points in each plot. The three samples from each gridpoint were compiled by 0–10 cm and >10–30 cm layers, kept on ice for no more than 10 hours, and air dried to constant mass (Wayman and North 2007). Soils were passed through a 2-mm sieve, and then analyzed for total carbon by the ANR analytical laboratory at the University of California, Davis.

Soil roots were sampled using a 7-cm-diameter soil sampler at two depths: 0 to ~10 cm and 10 to ~20 cm. In the field each sample was stored in a cooler at <4°C and then frozen in the laboratory for longer storage. After thawing, each sample was washed using a root washer, and roots were manually separated into fine (≤2 mm in diameter) and coarse (>2 mm) roots. Samples were stored in paper bags, dried at 65°C for 48 hours and then weighed. Consistent with other research findings (Arkley 1981, Hubbert et al. 2001*a*, *b*, Witty et al. 2003), soil pits dug at our study site found that many coarse roots extend more than 2 m deep. Therefore, we did not use the coarse root material in 0–20 cm soil samples and instead used allometric equations from Jenkins et al. (2003) to estimate coarse-root biomass. For both soils and roots, we did not sample pretreatment conditions.

We examined each treatment's effect on potential fire behavior using plot data and the USDA Forest Service forest vegetation simulator (FVS) model (*available online*).[7] We calculated stand density and quadratic mean diameter from the posttreatment census of trees. We calculated 95% weather conditions for Teakettle using Fire Family Plus software (Main et al. 1990) and the two nearest Remote Access Weather Stations (RAWS), Dinkey Creek (30 km west) and Cedar Grove (30 km south). We modeled canopy bulk density, and "torching" (the 6-m wind speed at which surface fire is expected to ignite the crown layer) and "crowning" indices (the 6-m wind speed needed to support an active crown fire) using the Fire and Fuels Extension submodule of FVS.

Carbon calculations

Using the stem map data set, we applied genus-specific allometric equations using Jenkins et al. (2003) methods to calculate live-tree and snag carbon biomass before and after treatments. For the coarse and fine woody debris, we converted mass to carbon biomass assuming a carbon concentration of 50% (Penman et al. 2003). To quantify carbon in litter and duff, we used a biomass-to-carbon conversion factor of 37% (Smith and Heath 2002).

To calculate forest fuels consumed and emissions produced by the prescribed burn, we used pre- and posttreatment values for the different tree-based pools, surface fuels, and thinning byproducts. We totaled all pretreatment carbon in live trees, snags, coarse and fine woody debris, and litter, and then subtracted posttreatment values for the same fuels plus slash piles, lumber, milling waste, and stumps. Although we assume the difference between these two values was the amount of carbon released by the prescribed burn, these estimates should be viewed with caution. We did not measure soil black carbon, which has been identified as a potentially significant C pool (Deluca and Aplet 2008). Therefore our emissions estimates may be higher than actual totals. However, we believe the estimates are closer to actual emissions than using a generalized model such as CONSUME, and any calculation bias will not effect the relative differences among treatments.

From USDA Forest Service records and personal observation, we knew the number of people commuting to the site each day, the equipment being used, and the total number of workdays. We used the U.S. Environmental Protection Agency's published estimates for average pickup-truck mileage combined with hourly fuel-consumption rates for chainsaw, yarding, and loading equipment from company sources (e.g., Stihl [Virginia Beach, Virginia, USA] and Caterpillar [Peoria, Illinois, USA]). We calculated diesel use by logging trucks to haul the logs 235 km to the Sierra Forest Products lumber mill in Terra Bella, California and back using mileage estimates from Kenworth Truck Company (Kirkland, Washington, USA) for the trucks used. All of the fuel-use data was converted to megagrams (Mg) of C. We calculated the timber volume that reached the mill using the scaling records for all the truckloads. One of the mill owners (L. Ducent, *personal communication*) estimated that ~60% of each log was converted to lumber and 40% ended up as waste (wood scraps and sawdust), an estimate consistent with other published studies (Skog and Nicholson 2000). Waste was burned or sold for landscaping bark and mulch. We considered this an emission although decomposition of the landscape material may take several years. We also calculated fuel use for the Forest Service crew that administered the prescribed burn.

Analyses

All data were standardized to per hectare values. We evaluated each variable for normality with the Kolmogorov-Smirnov test and for homoscedasticity with Levene's test. We tested for significant differences in treatment mean values with ANOVA and used Tukey's hsd post hoc analysis to detect which treatments significantly differed ($P < 0.1$) from each other.

Results

Carbon pools and stocks

Posttreatment live-tree carbon stocks ranged from a low of 65.8 Mg C ha^{-1} in the most intensive fuel treatment, the overstory thin and burn, to 249.8 Mg C/ha in the control (Table 1). We estimated 1865 live-tree stocks were 345.5 Mg C/ha. Treatments significantly reduced live-tree carbon by 6.8–65% reflecting treatment intensity. The overstory thin and burn was the only treatment in which snag carbon decreased, due to a higher burn intensity that incinerated many snags. In all other treatments, snag carbon storage (25.2–34.1 Mg C/ha) increased 30–65%. The next largest carbon pool,

[7] ⟨http://www.fs.fed.us/fmsc/fvs/?⟩

TABLE 1. Carbon storage (Mg C/ha) for different pools, for each of the six fuels treatments.

Pool	Fuel treatment					
	Control	Burn only	Understory thin	Understory thin and burn	Overstory thin	Overstory thin and burn
Trees						
C storage (Mg C/ha)	249.8^{a}	198.3^{b}	172.6^{c}	144.1^{d}	89.4^{e}	65.8^{f}
Δ (%)		−6.8	−28	−34	−56	−65
Total C (%)	55.9	55.8	40.1	39.2	26.0	22.7
Snags						
C storage (Mg C/ha)	20.8^{a}	25.2^{a}	34.1^{b}	26.4^{a}	25.6^{a}	16.7^{c}
Δ (%)		+49	+65	+30	+55	−16
Total C (%)	4.7	7.1	7.9	7.2	7.4	5.8
Coarse woody debris						
C storage (Mg C/ha)	27.6^{a}	9.4^{b}	20.3^{c}	9.0^{b}	17.6^{c}	8.4^{b}
Δ (%)		−61	−24	−61	−21	−55
Total C (%)	6.2	2.6	4.7	2.5	5.1	2.9
Fine wood debris						
C storage (Mg C/ha)	4.2^{a}	4.2^{a}	7.7^{b}	5.1^{a}	8.4^{b}	4.3^{a}
Δ (%)		−26	−24	−50	+24	−58
Total C (%)	1.0	1.2	1.8	1.4	2.4	1.5
Litter						
C storage (Mg C/ha)	7.9^{s}	4.7^{b}	7.4^{a}	4.4^{b}	9.6^{c}	1.6^{d}
Δ (%)		−41	+18	−45	+28	−79
Total C (%)	1.8	1.3	1.7	1.2	2.8	0.6
Piled slash						
C storage (Mg C/ha)	n.a.	n.a.	1.1^{a}	1.1^{a}	1.8^{b}	2.0^{a}
Total C (%)			0.3	0.3	0.5	0.7
Stumps and their coarse roots						
Mean C storage (Mg C/ha)	n.a.	n.a.	17.0^{a}	29.2^{b}	30.7^{b}	38.2^{c}
Total C (%)			4.0	8.0	9.0	13.2
Soil (0–30 cm)						
Mean C storage (Mg C/ha)	78.1^{a}	67.6^{a}	103.0^{b}	82.0^{a}	85.2^{a}	81.4^{a}
Total C (%)	17.5	19.0	23.9	22.3	24.3	28.0
Fine roots						
Mean C storage (Mg C/ha)	5.5^{a}	4.0^{b}	3.0^{c}	2.7^{c}	2.0^{cd}	1.7^{d}
Total C (%)	1.2	1.1	0.7	0.7	0.6	0.6
Coarse roots						
Mean C storage (Mg C/ha)	52.8^{a}	42.0^{b}	36.5^{b}	30.4^{b}	18.9^{c}	13.9^{c}
Total C (%)	11.8	11.8	8.5	8.3	5.5	4.8
Lumber						
Mean C storage (Mg C/ha)	n.a.	n.a.	27.5^{a}	32.8^{a}	54.4^{b}	56.2^{b}
Total C (%)			6.4	8.9	15.8	19.4
Total						
Mean C storage (Mg C/ha)	446.7^{a}	355.4^{b}	430.2^{a}	367.2^{b}	343.6^{b}	290.2^{c}

Notes: Key to variables: Δ%, the change from pretreatment C storage mean; total C (%), percentage of total C in each pool. Surface woody material is divided into two groups: coarse, pieces with a diameter ≥30 cm; or fine, with a diameter <30 cm. Values in a row with different lowercase superscript letters are significantly different at $P < 0.1$; n.a. indicates not applicable.

the top 30 cm of soil, did not significantly differ among five of the treatments (67.6–85.2 Mg C/ha). The understory thin treatments had a significantly higher soil carbon average of 103.0 Mg C/ha than the other treatments. Litter and fine woody debris (FWD) had similar patterns, where biomass generally increased with thinning intensity in the unburned treatments (excepting FWD in the understory thin) and decreased in the burns. Coarse woody debris significantly declined in all treatments, with higher losses in the burn treatments (–55% to –61%) than the thin-only treatments (–21% to –24%). Fine- and coarse-root biomass decreased with treatment intensity from 5.5 and 52.8 Mg C/ha, respectively, in the control, to 1.7 and 13.9 Mg C/ha, respectively, in the overstory thin and burn. Carbon totals for lumber were similar within thinning intensity, with the overstory thin averaging 183% of the understory thin (Table 1).

Allocation of carbon among the different pools varied considerably among treatments (Table 1). As treatment intensity increased, the relative proportion of carbon in live pools (trees, fine and coarse roots) decreased.

TABLE 2. Carbon releases (Mg C/ha) from five different fuel treatments in the Teakettle Experiment.

Fuel treatment	Carbon releases, by source			
	Prescribed burn	Total equipment releases	Trucking to mill	Milling waste
Burn only	14.791[a]	0.014[a]†	n.a.	n.a.
Understory thin	n.a.	0.641[b]	1.126[a]	18.323[a]
Understory thin and burn	23.397[b]	0.696[b]	1.200[a]	21.889[a]
Overstory thin	n.a.	1.086[c]	1.852[b]	38.282[b]
Overstory thin and burn	27.224[c]	1.197[c]	2.081[b]	37.472[b]

Note: Data in a column with different lowercase superscript letters are significantly different at $P < 0.1$.

† For commuting fuel used by prescribed-burn crew.

Compared to the control, treatments increased snag and decreased log carbon pools. Burn treatments reduced the relative proportion of carbon in surface fuel pools but had little effect on soil carbon pools. Understory and overstory thinning-only treatments significantly reduced live-tree carbon from pretreatment levels (67.1 and 113.8 Mg C/ha, respectively) with an average of 78% of that reduction reaching the mill (lumber and milling waste).

Milling waste was the highest source of carbon release and significantly differed (20.1 and 37.9 Mg C/ha mean) between thinning intensities (Table 2). Prescribed fire was the next highest source, with significantly greater emissions in the thin-and-burn treatments (23.4 and 27.2 Mg C/ha) than in the burn-only treatment (14.8 Mg C/ha). With only one remaining sawmill in the southern Sierra Nevada, the long haul distance to the processing mill made the carbon release from log truck diesel (1.1–2.1 Mg C/ha) much greater than on-site equipment releases. On-site releases varied by the amount of time spent in each treatment, which was proportional to the timber volume removed.

Changes in stand structure and potential fire behavior

Treatments significantly reduced stand density to a range of 94–354 stems/ha compared to the control's 469 stems/ha (Table 3). None of the treatments, however, killed enough small trees (<25 cm dbh) to significantly boost average live-tree diameter (19.6 cm in the control and 22.0–28.9 cm in the treatments). Canopy bulk density significantly decreased with treatment intensity, with the overstory-thin treatments substantially reducing tree-canopy continuity and foliage volume. Fire Family Plus software (Main et al. 1990) calculated 95th-percentile weather conditions at Teakettle Experimental Forest (Sierra National Forest, USA) as: a one-minute maximum wind speed of 27.4 km/h, dry-bulb temperature of 33.3°C, relative humidity of 11%, and fuel moistures of 2%, 3%, and 6% for 1-, 10-, and 100-hour fuels, respectively. Compared to the control, all thinning treatments significantly increased both the torching and crowning index (for definitions, see *Methods: Analyses*, above), with gains of 6.3–30.7 and 8.4–23.2 km/h, respectively.

Treatments significantly altered stand structure changing both fuel loading and the stand's diameter distribution by species. Burn treatments substantially reduced all fuel classes, as thinning-only treatments had more modest reductions (Fig. 1). Burning treatments reduced fine (1 and 10 hour classes) and coarse (1000-hour) fuel loads while slightly increasing intermediate (100-hour) fuels in two of the three treatments (burn only and understory thin and burn). Although less than the burn treatments, thinning reduced very fine (1-hour) and coarse (1000-hour) fuels, but left fine and intermediate fuels unchanged (excepting an increase in 100-hour fuels in the overstory thin). Of all the treatments, only the overstory thin and burn (87 stems/ha) substantially reduced the density of small (5–19 cm dbh class), flammable trees. However, the historic 1865 small-tree density (15 stems/ha) was much lower (Fig. 2). All of the thinning treatments reduced the number of trees in the 20–39, 40–59, and 60–79 cm dbh class. Reductions in the 20–39 cm class can substantially reduce ladder fuels (structures that provide vertical fuel continuity between the forest floor and overstory tree crowns), but thinning of larger trees may reduce the supply of intermediate-sized trees needed to grow future large trees, particularly in the overstory thins. All posttreatment plots had many more stems in the 20–39 cm ladder-fuel class and many fewer intermediate size trees than were present in 1865. Overstory-thinning treatments, which reduced stem densities in the 80–99 cm and greater diameter classes, reduced the number of fire-resistant trees, which are also a substantial portion of the forest's carbon stock. Most of the live-tree carbon in 1865 was in the large-diameter classes (88% in ≥80 cm dbh), as all treatments including the control now have <68% in these size classes. Although the control has more trees in the >120 cm dbh class (Fig. 2), many of these are white fir 121–130 cm dbh, compared to 150–210 cm dbh trees in the 1865 reconstruction. Changes in species composition, particularly an increase in the percentage of pine, can increase fire resistance. The thinning prescriptions, however, were diameter based and applied uniformly across species. Consequently none of the treatments significantly increased pine percentages.

TABLE 3. Posttreatment stand characteristics of the six fuel treatments in the Teakettle Experiment Forest.

Treatment	Stand characteristics				
	Stand density (stems/ha)	Quadratic mean dbh (cm)	Canopy bulk density (kg/m^3)	Torching index (km/h)	Crowning index (km/h)
Control	469^{a}	19.6^{a}	0.78^{a}	9.8^{a}	43.2^{a}
Burn only	354^{b}	22.0^{a}	0.078^{a}	16.1^{b}	43.2^{a}
Understory thin	240^{c}	23.4^{a}	0.058^{b}	25.4^{c}	53.3^{b}
Understory thin and burn	143^{d}	28.9^{a}	0.061^{b}	25.7^{c}	51.6^{b}
Overstory thin	150^{d}	21.9^{a}	0.052^{c}	40.5^{d}	59.4^{bc}
Overstory thin and burn	94^{e}	24.2^{a}	0.052^{c}	37.6^{d}	66.4^{c}

Note: Values in a column with different lowercase superscript letters are significantly different ($P < 0.1$).

DISCUSSION

In our study, modern fire-suppressed forests had substantially lower live-tree carbon stocks (–28%) than historic active-fire conditions (Fig. 3) and are at risk of creating large emissions if burned by wildfire (Hurteau and North 2009). Fuels treatments did increase stand-level fire resistance but reduced stocks even further, and produced significant milling waste and/or prescribed-fire emissions (Fig. 3). Heavier overstory thinning did not significantly improve fire resistance but substantially reduced carbon stocks. All fuels treatments create carbon emissions, but emissions can be reduced and future carbon stocks increased by modifying treatments to reduce surface fuels, small trees, and intermediate-size, fire-sensitive species.

Consistent with other studies of forest carbon dynamics (Turner et al. 1995, Smithwick et al. 2002, Li et al. 2007), we found the largest carbon pool was in live trees. Our estimates of historic (345.5 Mg C/ha), fire-suppressed (249.8 Mg C/ha) and treated (65.8–198.3 Mg C/ha) forest conditions are higher than 42.5–59.6 Mg C/ha estimates in ponderosa pine forests of the Southwest (Dore et al. 2008, Finkral and Evans 2008), but generally less than the 209.7–629.7 Mg C/ha found in a comparison of several Pacific Northwest old-growth forests (Smithwick et al. 2002). Mechanical thinning reduced the live-tree carbon pool, with the most significant reduction in the overstory-thinning treatments because of the removal of large trees that contain a significant proportion of total live-tree carbon stocks (Fig. 2). Thinning also increased snag biomass due to mechanical damage from felling and skidding operations that turned some live trees into snags. Fuels treatments had little effect on soil carbon, but by reducing live-tree pools, substantially shifted the relative proportion of total forest carbon, increasing the percentage of carbon in snag, soil, and lumber pools, and decreasing percentages in live roots and surface fuels. The overstory thin and burn reduced live-tree carbon enough for soil carbon to proportionally become the stand's largest carbon pool. This shift is consistent with other studies in harvested stands (Irvine et al. 2007, Li et al. 2007), where the relative percentage of total carbon increases in soil pools in the short term, but live-tree carbon pools increase as stands develop following treatment (Gough et al. 2007).

Current forests with high stem densities, dominated by small (<40 cm), fire-prone trees, are dramatically different from historic stand conditions, which had much lower densities composed primarily of large trees that contained almost 90% of the live-tree carbon stock (Fig. 2a). Some research has suggested that fire suppression has increased live-tree carbon stocks as a result of increased stem density and expansion by wood biomass into areas that were historically open as a result of frequent fire (Houghton et al. 2000, Hurtt et al. 2002). However, a recent study by Fellows and Goulden (2008) found lower carbon stocks in modern fire-suppressed conditions than in 1930, due to the loss of large trees. We found the difference was due to the presence of very large trees (≥150 cm dbh) in 1865 that are lacking in current fire-suppressed forest conditions. Forest infilling by shade-tolerant species in the absence of fire may have contributed to this carbon-stock loss. An earlier study at Teakettle (Smith et al. 2005) found significantly higher than expected mortality in the largest tree size class, possibly due to collateral bark beetle attacks when high densities of small-diameter stems surround large trees of the same species. We suspect access to deep water may be one reason why historic stands with a low-density of large trees could support more biomass than modern, fully stocked, fire-suppressed old growth. Consistent with studies in southern California (Arkley 1981, Hubbert et al. 2001*a*, *b*, Witty et al. 2003), an isotope study of plant water use at Teakettle (Plamboeck et al. 2008) found large trees were almost exclusively using deep (≥70 cm) soil water, while small trees and shrubs compete for shallow (<50 cm) soil water that is rapidly exhausted during the growing season. If these patterns hold true in other forests, fire suppression may have incurred a double carbon penalty by reducing stocks and contributing to potential emissions with fuels treatment activities or inevitable wildfire combustion.

Management practices in fire-prone western forests of the United States need to balance effective fuels treatment with minimizing carbon pool reduction and carbon emissions. Our study suggests there are tradeoffs between effective, immediate fuels reduction (i.e., higher carbon release) and longer-term carbon storage (i.e.,

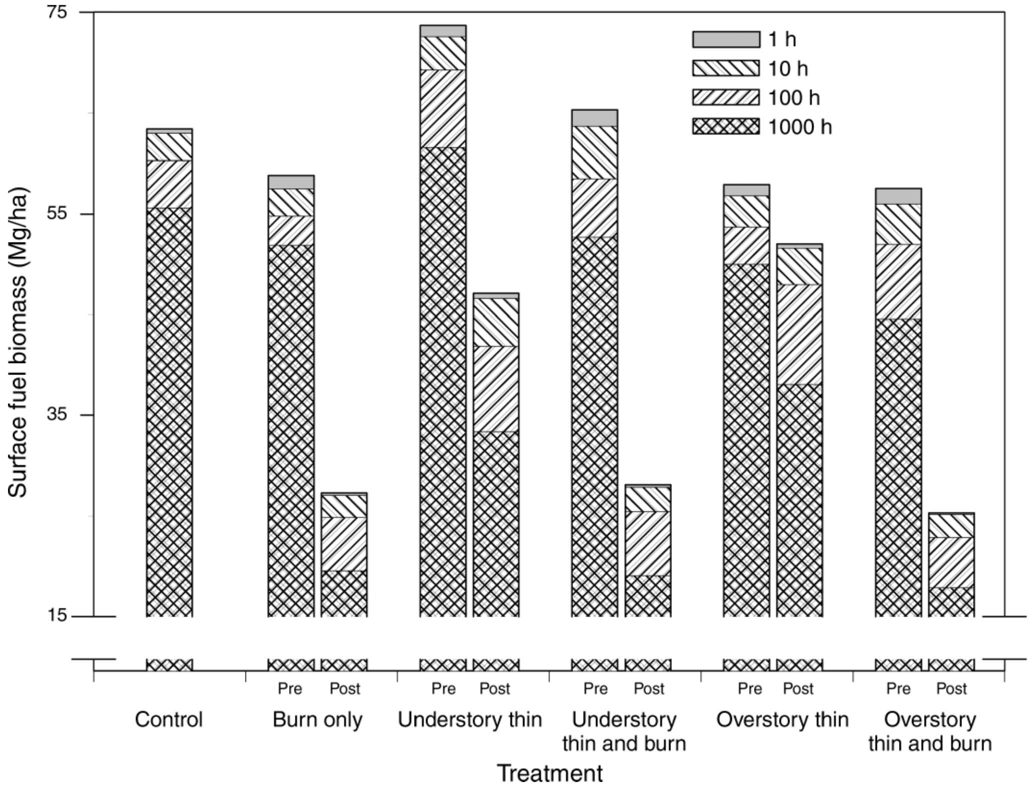

FIG. 1. Surface-fuel biomass pre- and posttreatment for six different treatments in the Teakettle Experiment, by four fuel moisture hour classes.

increasing a stand's wildfire resistance). Prescribed burning significantly reduced surface and ladder fuels, and by opening up growing space for residual trees, created stand conditions favoring faster development of more fire-resistant, large trees (Hurteau and North 2009). Prescribed burning, however, also produced an immediate carbon emission, and pre-burn thinning reduced live-tree carbon stocks by 34–65% while increasing emissions by 21.9–37.5 Mg C/ha in the form of milling waste. If milling waste is used to generate energy, however, it might be considered an offset for fossil-fuel consumption.

When evaluating carbon released by different fuels treatments, managers will need to weigh trade-offs between immediate prescribed-burn emissions, increased fuel reduction with thinning and an increase in milling waste, and potential future wildfire emissions. A thinning-only treatment avoids immediate prescribed-fire emissions, but releases 18.3–38.2 Mg C/ha in milling waste. The thin-and-burn treatments substantially reduced surface fuels compared to the thinning-only treatments, but at a higher emissions costs (23.4 and 27.2 Mg C/ha, respectively). Most of the surface fuel biomass and prescribed-fire emissions are from the 1000-hour and larger CWD (coarse woody debris) fuels, which are substantially increased as larger trees are removed (Fig. 1). The understory thin and understory thin and burn had average live-tree stock losses of 28 and 34% from pretreatment levels, respectively. Compared to the control, these reductions in the carbon stock increased the torching and crowning indices by 15.6 and 10.1 km/h for the understory thin and 15.9 and 8.4 km/h for the understory thin and burn (Table 3), respectively. Compared to the understory treatments, the overstory thin and overstory thin and burn had higher torching and crowning indices but live-tree carbon stocks were reduced by almost half. Compared to the control, most of the decrease in canopy bulk density was achieved with the understory thinning with lower additional changes in the more intensive overstory thinning.

In practice, evaluating these trade-offs will also hinge on other factors such as the availability of treatment funds, air-quality restrictions on burning, and how much risk the managers are willing to accrue. In weighing these options, we suggest uncertain future wildfire emissions not be heavily discounted. Modeled future climate conditions in California suggest increases in temperature and growing-season length are likely to occur (Field et al. 1999, Hayhoe et al. 2004, Cayan et al. 2008) which may already be increasing fire-season length in the western United States (Westerling et al. 2006). Recent research also suggests current estimates of wildfire carbon emissions may only be a portion of actual carbon losses if the fire is high intensity leaving

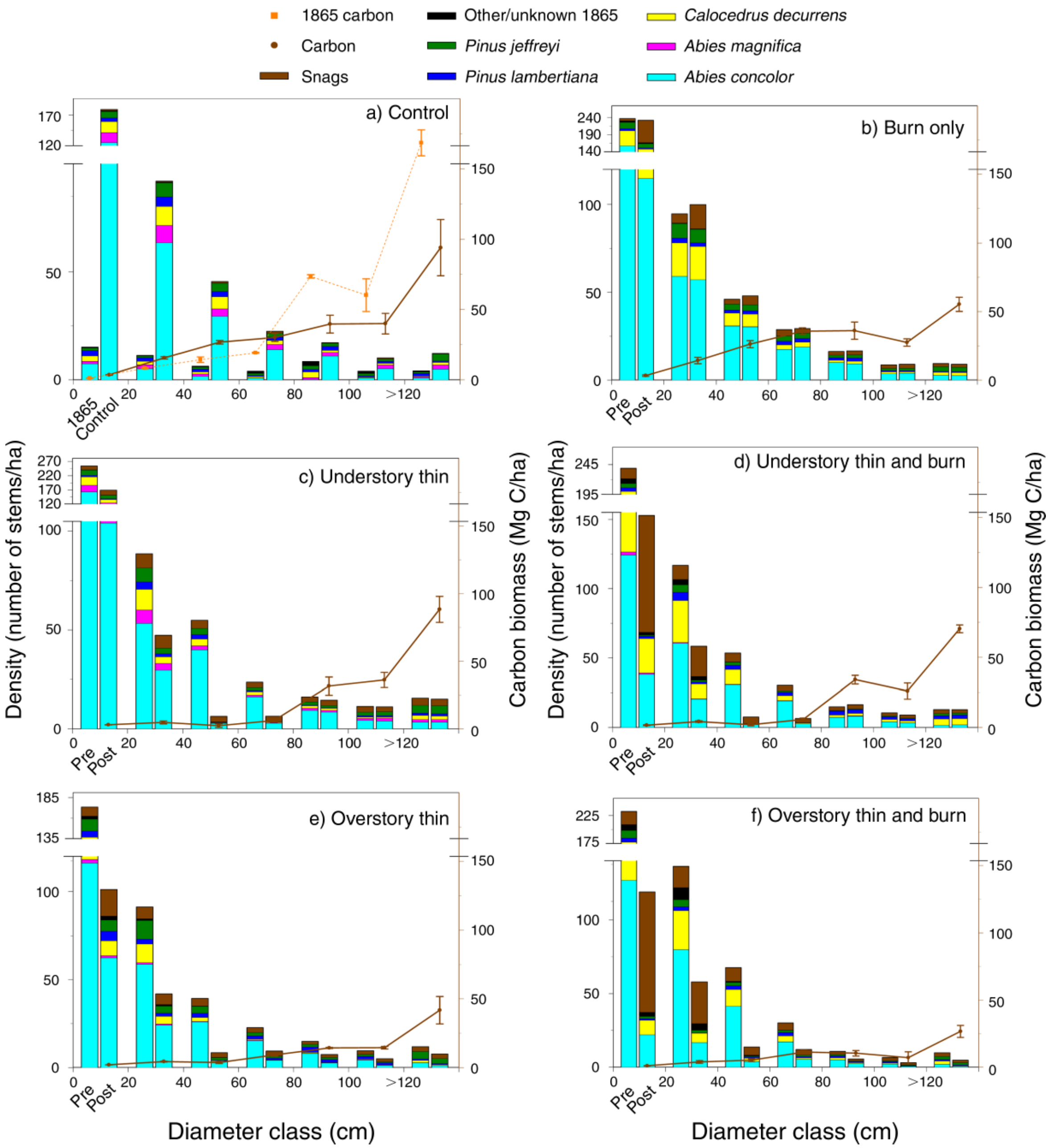

FIG. 2. Density of tree stems by species (histogram bars) and carbon biomass (data points connected by lines; right-hand axis) in 20-cm dbh classes for (a) the 1865 reconstruction and the control and (b–f) for five fuel treatments. Note that the last dbh class is for all stems >120 cm. The pairs of bars in panel (a) represent the 1865 reconstruction and the control; in all other panels, pairs of bars present data pre- and posttreatment. The black segment in each bar represents unknown species in the 1865 reconstruction; in all other treatments it indicates "other" species (primarily hardwoods). Snag density and size could not be calculated for the 1865 reconstruction. The data for carbon biomass are means ± SE; the dotted and solid lines simply connect the data points to help one visualize trends. In panel (a) the dotted line connects the 1865 carbon stock data for reconstruction of stand condition and diameter distribution.

few surviving trees (Kashian et al. 2006, Bormann et al. 2008, Dore et al. 2008).

For future stand development, a significant shortcoming in the treatments was leaving all trees <25 cm believing logging operations and the prescribed burn would kill most of them (Mark Smith [Sierra National Forest silviculturist], *personal communication*). In many fuels treatments, this size class is either left on site or removed using expensive service contracts (i.e., from \$1200 to \$3500 per hectare) if funds are available. Many of these trees survived treatment, reducing fuel-reduction effectiveness, and leaving stand densities much

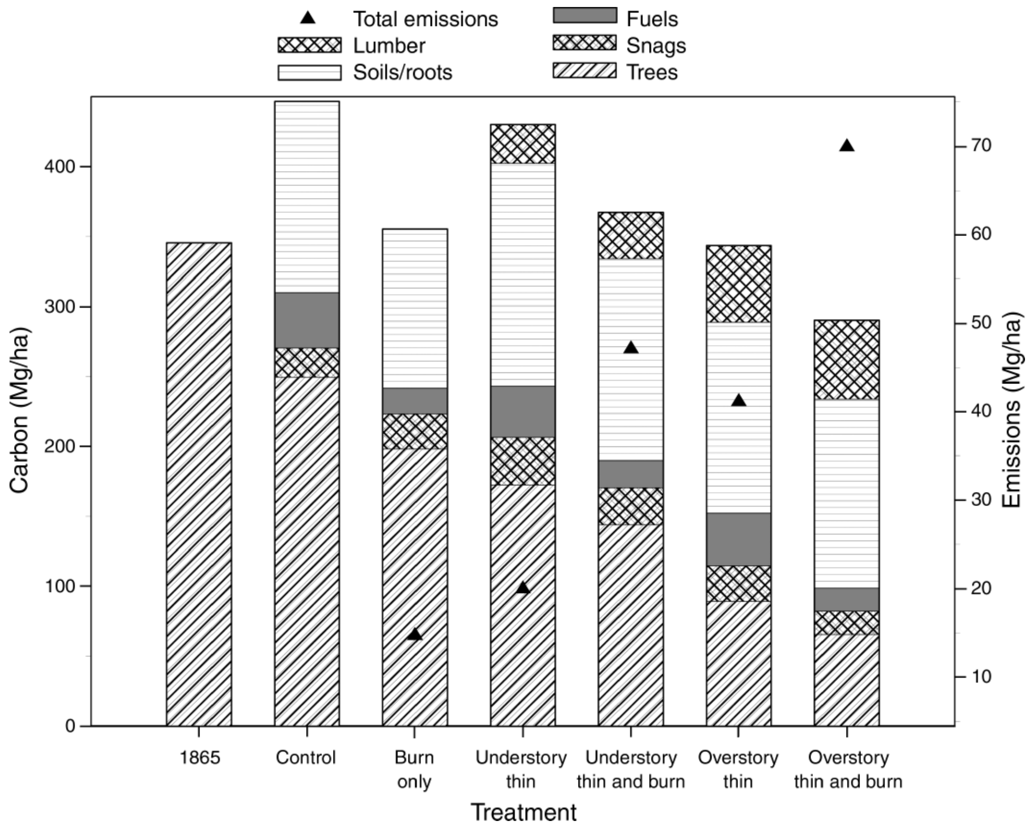

FIG. 3. Total carbon pools (histogram bars) and emissions (triangles; right-hand axis), by treatment. The soils/roots pool includes soil carbon in the top 30 cm, fine and coarse roots, and coarse roots for stumps in the thinning treatments. Snag, fuels, and soils/roots pools could not be calculated for the 1865 reconstruction. Total emissions are the sum of prescribed fire, fossil fuels burned, and milling waste.

higher (94–354 stems/ha) and quadratic mean diameter much lower (19.6–28.9 cm) than the 1865 active-fire conditions (67 stems/ha and 49.7 cm) (North et al. 2007). Removing all trees <25 cm in diameter would only slightly decrease live-tree carbon stocks (e.g., a 5.3% reduction in the control) while substantially raising height to the base of the live crown, a key influence on wildfire behavior (Reinhardt and Crookston 2003). Stand composition also shifts to a higher percentage (by stem frequency) in fire-resistant pines, with increased growing space for potentially more rapid development of large trees.

Previous Teakettle studies (Innes et al. 2006, North et al. 2007, Hurteau and North 2009) coupled with our present research suggest treatments could be modified to more effectively minimize carbon stock reductions while still significantly reducing fuels and promoting large-tree development. Significant increases in wildfire resistance can be achieved by thinning only smaller ladder fuels and fire-sensitive intermediate trees without reducing the majority of the live-tree carbon pool in intermediate pines and large trees of all species.

The trajectory of future carbon stocks in fire-prone forests hinges on management actions that influence both stand dynamics and when wildfire occurs. At Teakettle we have found rapid growth of large trees after past fire events that presumably reduced stand density (North et al. 2005, Hurteau et al. 2007). Thinning and prescribed-fire treatments that reduce small-tree densities may influence stand development by redirecting growth resources and carbon storage into more stable stocks such as large, long-lived fire-resistant pines (Hurteau and North 2009). Thin-only treatments, however, add surface fuels, incurring a risk of higher burn intensity and larger carbon release if wildfire occurs shortly after treatment. While incurring an immediate carbon "penalty," prescribed burning has lower emissions, benefits many ecosystems processes (North 2006), favors more fire-resistant pine regeneration (Zald et al. 2008) and allows managers to better control fire intensity, carbon release, and smoke drift. The results of this present research coupled with other studies indicates that over time the carbon stock will recover to its pre-fuels reduction state and likely be more resistant to high-severity fire. The growth release from thinning may expedite carbon-stock recovery, however; further research is needed to evaluate how many years of posttreatment forest growth is needed to offset immediate carbon releases from different thinning and prescribed-fire treatments.

Forests in the United States sequester ~10% of annual anthropogenic CO_2 emissions (Woodbury et al. 2007). Wildfires are increasing in size and severity (Westerling et al. 2006) and produce large direct CO_2 emissions on the order of 4–6% of annual U.S. anthropogenic emissions (Wiedinmyer and Neff 2007). Treating fire-suppressed forests to reduce potential wildfire emissions creates short-term carbon emissions. However with proper fuels treatment creating favorable stand conditions for increasing large-tree growth, forests could be a substantial future sink, sequestering carbon in relatively stable, long-lived structures. Our research suggests most of the benefits of increased stand-level fire resistance can be achieved with small reductions in carbon pools. Prescribed-fire and milling-waste emissions could be substantially reduced by changing treatments to vary thinning prescriptions by species and focusing more on reducing surface and small-diameter fuels that most affect fire severity. Forest carbon-stock stability can be improved by incorporating our understanding of stand and fire dynamics into current carbon-accounting policy.

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