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Fuel treatment impacts on estimated wildfire carbon loss from forests in Montana, Oregon, California, and Arizona

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Abstract. Using forests to sequester carbon in response to anthropogenically induced climate change is being considered across the globe. A recent U.S. executive order mandated that all federal agencies account for sequestration and emissions of greenhouse gases, highlighting the importance of understanding how forest carbon stocks are influenced by wildfire. This paper reports the effects of the most common forest fuel reduction treatments on carbon pools composed of live and dead biomass as well as potential wildfire emissions from six different sites in four western U.S. states. Additionally, we predict the median forest product life spans and uses of materials removed during mechanical treatments. Carbon loss from modeled wildfire-induced tree mortality was lowest in the mechanical plus prescribed fire treatments, followed by the prescribed fire-only treatments. Wildfire emissions varied from 10–80 Mg/ha and were lowest in the prescribed fire and mechanical followed by prescribed fire treatments at most sites. Mean biomass removals per site ranged from approximately 30–60 dry Mg/ha; the median lives of products in first use varied considerably (from <10 to >50 years). Our research suggests most of the benefits of increased fire resistance can be achieved with relatively small reductions in current carbon stocks. Retaining or growing larger trees also reduced the vulnerability of carbon loss from wildfire. In addition, modeled vulnerabilities to carbon losses and median forest product life spans varied considerably across our study sites, which could be used to help prioritize treatment implementation.

Key words: carbon sequestration; fire suppression; fire surrogates; fuel management; mixed conifer; *Pinus ponderosa*; wildfire.

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Introduction

The use of forests to sequester carbon (C) in response to anthropogenically induced climate change is being considered across the globe (Choi et al. 2006). Large quantities of C can be stored or released to the atmosphere by soils and plants and this varies by ecoregion, vegetation type, climate, disturbance history, and land use practices (Finkral and Evans 2008, Bowman et al. 2009, Mitchell et al. 2009). Recent decades have brought increased concern over the Earth's changing climate and shifting C balance, which is coincident with increases in both area burned by wildfire and wildfire severity (McKenzie et al. 2004, Stephens 2005, Westerling et al. 2006, Miller et al. 2009).

Various forms of vegetation management, largely consisting of prescribed burning and/or mechanical thinning, have been recommended for forests that are currently susceptible to high intensity wildfires, particularly for those forest types that historically burned frequently under low-moderate intensity fire regimes. These management activities or treatments are intended to reduce fire-caused overstory tree mortality and commonly involve reducing surface fuels and removing intermediate sized trees that represent ladder fuels (Fig. 1) (Agee and Skinner 2005, Fule et al. 2012). In the short-term, reducing surface fuels and numerous small- to intermediate-sized trees will result in both a release of accumulated C into the atmosphere and an initial reduction in C stocks (Hurteau and Brooks 2011). These outcomes may seem inconsistent with overall goals of reduced greenhouse gas emissions and increased C sequestration. However, studies have demonstrated that fuel reduction treatments can reduce C losses when treated stands are subsequently burned by wildfire (Finkral and Evans 2008, North et al. 2009, Hurteau and North 2010, Zhang et al. 2010). Furthermore, treated stands in many dry forest types in the western U.S. represent more stable structures for long-term forest C sequestration (Hurteau and Brooks 2011). Recent research has analyzed the impacts of fuel treatments on C stocks within a risk assessment framework that not only examines the differential C losses from wildfire in treated and untreated stands, but incorporates the probability of fire occurrence as well (Ager et al. 2010, Campbell et al. 2011). These studies suggest overall C losses associated with fuel treatments can outweigh the benefit of reduced wildfire-related C losses, except in landscapes that burn most frequently.

The reduction of fire hazards in forests that once burned frequently with low-moderate intensity can be complimentary to ecological restoration (Moore et al. 1999, Allen et al. 2002, Fulé 2008, Stephens et al. 2012). Fuel reduction treatments can create stand structures that maintain or restore several forest attributes and processes including, but not limited to, snag and coarse woody debris recruitment, floral and faunal species diversity, and seedling establishment. Managing to increase resilience in response to novel climate conditions has become a frequent goal in many western U.S. forests (Millar et al. 2007, Stephens et al. 2010) with carbon sequestration one of the possible ecosystem services that forests provide.

Recently President Obama issued an Executive Order (No. 13514: "Federal Leadership in Environmental, Energy, and Economic Performance" October 5, 2009) that focused on reducing greenhouse gas emissions. This executive order mandated that all U.S. federal agencies develop plans that "consider and account for sequestration and emissions of greenhouse gases resulting from federal land management practices." This highlights the importance of understanding not only C sequestration and emissions in forests, but how forest C stocks are influenced in areas of high fire potential. Several studies have investigated C dynamics related to implemented (as opposed to simulated) fire mitigation treatments from the southwestern U.S. (Finkral and Evans 2008, Hurteau et al. 2011), the southern Cascade Range (Zhang et al. 2010), and the Sierra Nevada (Hurteau and North 2009, North et al. 2009, Stephens et al. 2009*a*). However, it is unclear if the findings from these studies can be applied to other dry forests throughout the western U.S. where large-scale fire mitigation efforts are being planned. Given the scope of the 2009 Executive Order on greenhouse gas emissions and C sequestration, multi-site comparisons spanning climatic gradients and forest types are needed to better inform policy development.

This study offers an opportunity to compare the impacts of fuel treatments on C emissions and stocks among several sites across the western U.S.

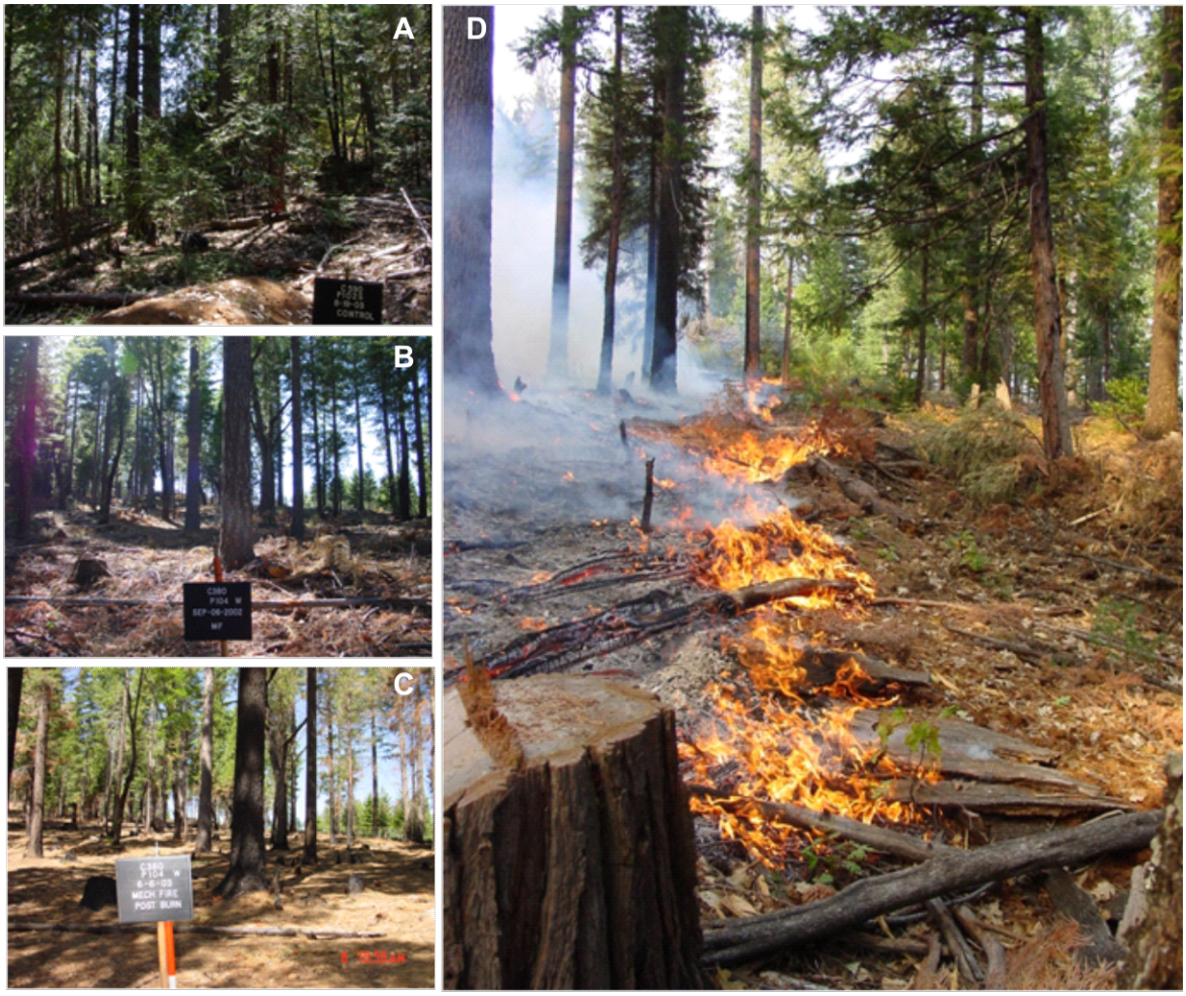

Fig. 1. Examples of fire and fire surrogate treatments applied to mixed conifer forest stands in the Central Sierra Nevada that have been repeatedly harvested and fire has been excluded for 100 years: (A) control, (B) after mechanical treatment, (C) after mechanical treatment followed by prescribed fire, and (D) backing prescribed fire in unit previously treated by mechanical methods.

This effort expands on an earlier analysis that investigated how fuel treatments affected C stocks at a single site (Stephens et al. 2009*a*). Our objective is to elucidate how some of the most common forest fuel treatments used in Montana, Oregon, California, and Arizona affect C pools in live biomass, dead biomass (surface woody debris, litter, and duff), and estimated wildfire emissions. Additionally, we analyze how these treatments influence the potential for loss of live tree C after subsequent wildfire. Median forest product life span and uses of the C removed from treatments are also analyzed. We do not, however, estimate C emissions resulting from the treatments themselves, i.e., prescribed fire emissions, harvesting equipment, and transportation but make estimates in this area from previous studies. Our null hypothesis is that there will be no significant difference among treated and untreated stands in the Mg of emissions released from wildfire and the amount of live C susceptible to high severity fire among our research sites.

Methods

Study sites

This paper uses data from the Fire and Fire Surrogate study (FFS), a national multi-disciplin-

ary project implemented from 2000–2008 at 12 sites across the U.S. (McIver and Weatherspoon 2010). Treatments and data collection methods varied somewhat among sites; however, sufficient similarity in how the experiments were conducted facilitated comparison of results across sites (Schwilk et al. 2009). This paper focuses on fuel treatment effects for a subset of six sites that are representative of the most common dry coniferous forest types in the western U.S. (Fig. 2; McIver et al. 2009).

The FFS sites were selected to represent forests characterized by historical fire regimes of primarily frequent, low-moderate intensity fires. The six sites included in this study are: (1) Southern Cascades, within the Klamath National Forest in northern California; (2) Central Sierra Nevada, at Blodgett Forest adjacent to the El Dorado National Forest in east-central California; (3) Southern Sierra Nevada, within Sequoia National Park in east-central California; (4) Blue Mountains, within the Wallowa-Whitman National Forest in northeastern Oregon; (5) Northern Rocky Mountains, at Lubrecht Forest surrounded by the Lolo and Bitterroot National Forests in western Montana; and 6) Southwestern Plateau, within the Coconino and Kaibab National Forests in northern Arizona (Fig. 2).

Study sites span a latitudinal range of more than 12 degrees and contain forests that experience both summer rain and summer drought. Historical mean fire return intervals of the six sites ranged from 5–30 years (Fig. 2) and all sites have experienced a century of near total fire exclusion (Stephens et al. 2009*b*). Sites represented a diversity of past land management practices; five had been harvested repeatedly with the sixth at Sequoia National Park being an unharvested old-growth forest. Most sites are dominated by ponderosa pine (*Pinus ponderosa* Laws.) while others were considered mixed conifer (Southern Cascades, Central and Southern Sierra Nevada) with ponderosa pine as a component.

Treatments

Site-level treatments included an unmanipulated control, prescribed fire only (in the fall, spring, or both), mechanical only (including mechanical methods such as thinning and mastication), and mechanical plus prescribed fire (in the fall or spring). Regional variations in treatment implementation were reflective of local mechanical and prescribed burning practices (Stephens et al. 2009*b*, McIver and Weatherspoon 2010). All mechanical treatments included removal of merchantable sawlogs (generally trees greater than 20 to 25 cm diameter at breast height (dbh)) and some sites also removed biomass or pulp trees (trees 5 to 25 cm dbh). At the Southern Sierra Nevada site, mechanical treatments were not used; instead fall and spring prescribed burns were implemented to compare differences in burn seasonality.

Treatments were replicated three times at each FFS site except the Blue Mountains site, which had four replicates. Experimental units were at least 10 ha with a central measurement area used for field measurements to reduce edge effects. Treatments were randomly assigned to experimental units, except at the Southwestern Plateau site, where one experimental block required specific arrangements of burn units for safety reasons (Stephens et al. 2009*b*).

Assessment of trees, woody fuels, and forest floor changes

At the Central Sierra Nevada and Blue Mountains sites, trees (dbh $>$ 10 cm) and fuels were measured on a systematic grid of 0.04 ha circular plots (20 and 25 plots, respectively, in each experimental unit) following methods described in Youngblood (2010). The other four FFS sites used 10 modified Whittaker plots (0.1 ha) randomly located in each experimental unit to sample trees and fuels (Schwilk et al. 2009). Surface and ground fuels were sampled using the planar-intercept method (Brown 1974) along two randomly chosen azimuths (36 grid points per experimental unit), with duff and litter depths (cm) measured at two to three points along each transect (Schwilk et al. 2009, Stephens et al. 2009*b*).

For all sites, forest floor fuel mass (litter and duff) was calculated using either published equations (Brown 1974, van Wagtendonk et al. 1996, 1998) or site-specific fuel depth to weight relationships developed from destructive sampling of the forest floor (Stephens et al. 2009*b*). Data analyzed in this study were collected one year post-treatment, except at the Blue Mountains site, which were collected two years post-treatment.

Fuels were converted to C biomass assuming a

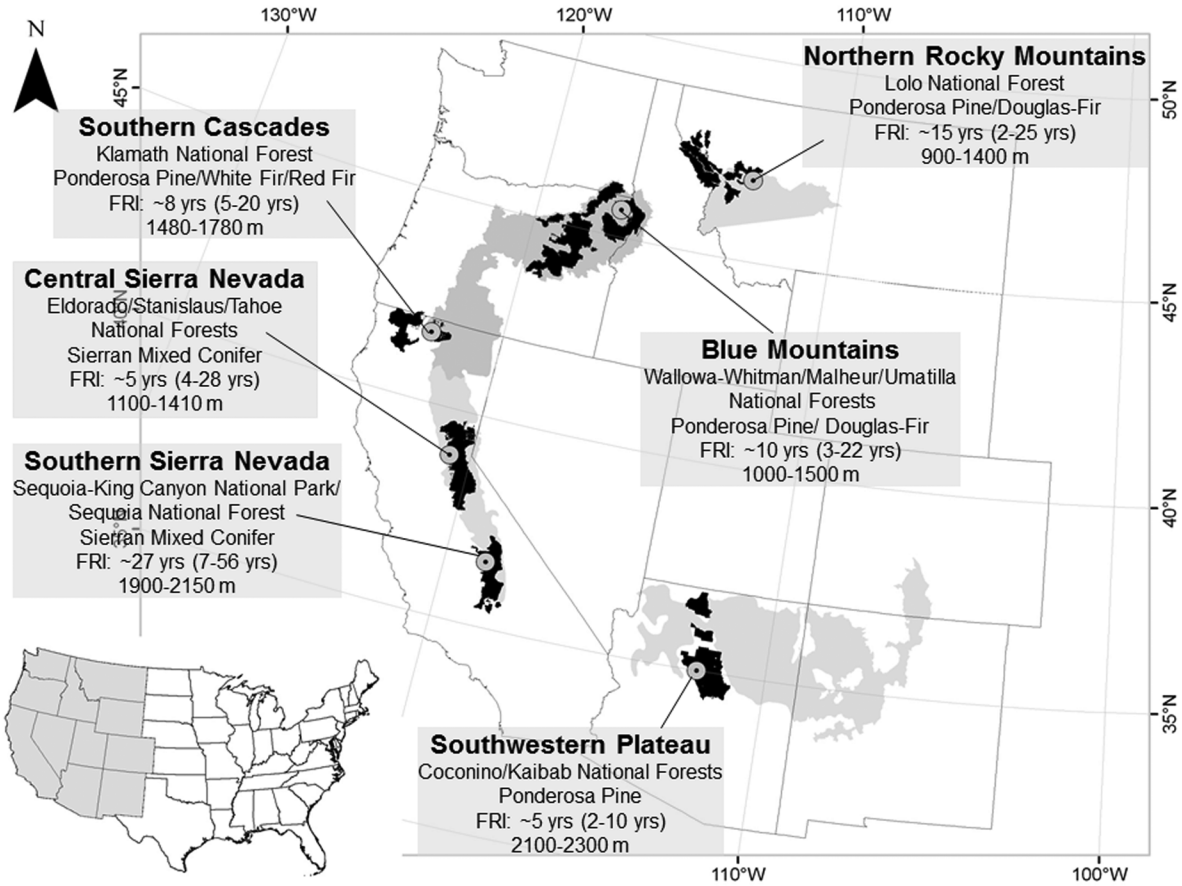

Fig. 2. Location, forest type, average fire return interval (FRI, mean with range in parentheses), and elevation of the six western U.S. Fire and Fire Surrogate study sites used in this work.

C concentration of 50% for coarse and fine woody fuels (Penman et al. 2003) and 37% for litter and duff (Smith and Heath 2002). We did not measure soil black C, which has been identified as a potentially significant C pool in some forests (Deluca and Aplet 2008). Total aboveground live and dead tree C was calculated using allometric equations provided in Jenkins et al. (2004). These equations, developed using a national database and parsed into ten-species groups, have been used to compute aboveground tree C in other conifer forests in the western U.S. (Hicke et al. 2007, Boerner et al. 2008, Hurteau and North 2009, North et al. 2009, Stephens et al. 2009*a*, Hurteau and North 2010, Collins et al. 2011).

Wildfire emissions

Analysis of potential wildfire C emissions from the six different FFS study sites was performed for each experimental unit using the batch input processing module in FOFEM 5.9 (Reinhardt 2003). FOFEM predicts fuel consumed by wildfire and estimates smoke emissions based on user inputs for: surface, ground, and canopy fuels, fuel moistures, proportion of stand affected by crown fire, and forest and fuel type.

FOFEM reports smoke emissions for several compounds: PM10, PM 2.5, CO, CO_2, CH_4, NO_X, and SO_2; we focused on CO, CO_2, and CH_4 because of our interest in C emissions from wildfires. For all six research sites the measured values for 1, 10, 100, and 1000 hour (sound and rotten separately) fuels were input into FOFEM. Litter and duff depths at the Southern Cascade and Central Sierra Nevada sites were actual field collected values, and at the other four sites we used values associated with the forest type

Table 1. Canopy bulk density (CBD) values and their corresponding foliage and branch biomass values used in this analysis. These biomass values were used as inputs for FOFEM (see *Methods*) for emissions modeling. CBD and biomass values were obtained from Scott and Reinhardt (2005).

Forest type	CBD (kg/m^3)	Biomass (kg/m^2)	
		Foliage	Branch
Ponderosa pine	0.166	0.88	3.66
	0.147	0.76	3.34
	0.104	0.51	2.43
	0.057	0.26	1.28
Mixed conifer	0.101	1.48	3.80
	0.101	1.44	3.75
	0.081	1.10	3.07
	0.037	0.84	2.52
	0.027	0.40	1.22

classes chosen within FOFEM.

The Pacific Ponderosa Pine (SAF 245) forest type was used for Blue Mountains, Northern Rocky Mountains, and the Southwest Plateau sites, while the Sierra Nevada Mixed Conifer type (SAF 243) was used for the Southern Sierra Nevada, Central Sierra Nevada, and Southern Cascades sites. For all six sites, herb and shrub biomass was estimated using the established values within FOFEM for these two forest types. The required canopy fuel inputs for FOFEM are crown foliage and crown branch biomass. These values were estimated based on canopy bulk density (CBD), which was calculated for each site using Fuels Management Analyst Plus (FMA) (Carlton 2004). Table 1 reports the foliage and branch biomass values used based on the closest CBD value that was compiled from Scott and Reinhardt (2005).

For the wildfire simulations, fuel moisture values for all sites were assumed to be 6% for 10 hour, 12% for 1000 hour fuels, and 20% for duff to reflect high to extreme fire weather conditions (Stephens et al. 2009*b*). These fuel moistures were used for all six FFS sites to allow for comparison of C emissions across forest types and fuel conditions. Canopy fraction burned was input individually for each experimental unit based on fire behavior/effects output from FMA (see next section for more information on this model). Region (4) and cover type (8) categories are limited within FOFEM, and as such, all forests were considered Pacific West forest types, with ponderosa pine being the primary cover group. Wildfire season was set to fall for all sites except the Southwest Plateau, which was set to spring.

Modeling potential fire behavior and severity

We simulated fire behavior and estimated C release from tree mortality (Mg/ha) using FMA. FMA uses forest structure information from field measurements, fuel models, and fire weather to simulate fire behavior and effects at the stand scale. Fuel models used for estimating fire behavior for each treatment were assigned by scientists associated with each FFS site and are given in Stephens et al. (2009*b*). Simulations were performed under the upper 97.5 percentile (extreme) fire weather conditions based on local archived Remote Access Weather Station (RAWS) data, and calculated using Fire Family Plus (Main et al. 1990).

Acknowledgment is given to the fact that the fuel and fire behavior models used in this assessment are simplified representatives of real fuel conditions (Burgan and Scott 2005) and fire behavior (Pastor et al. 2003). Further, all models have not been field validated because of the difficulty of doing so (Scott and Reinhardt 2001). Crown fire behavior is notably complex and is controlled by several interacting, highly variable elements such as weather, crown characteristics, and surface fuels, which the models homogenize (Stephens et al. 2009*b*, Cruz and Alexander 2010). That said, these models still represent the best available compilation of fire behavior science, whether empirically or theoretically derived (Pastor et al. 2003). While predictions should be used with caution for estimating absolute values of model outputs (Scott 2006), they are useful for making comparisons among different types of treatments.

Materials removed

The amount of material removed from each site was recorded in the experimental units according to local custom—e.g., thousand board feet, cubic feet or green tons—by type of product leaving the harvest unit—sawlogs, pulp logs, firewood, or chips for energy. We converted these to dry Mg and divided by the harvested area at each site to estimate removals in Mg/ha.

Following the approach described in Stephens et al. (2009*a*), we estimated median lives in first

use of the products generated from the materials removed from each site. The fractions of pulp-log and sawlog mass converted to various products were estimated from published sources (Biermann 1993, Brandt et al. 2006, Milota et al. 2005, Morgan et al. 2004, Southwest Forest Products 2011, Spoelma et al. 2008). Sources for median lives of the resulting products include McKeever (2002), Skog (2008), Skog and Nicholson (1998), and Winjum et al. (1998).

Data analysis

Analysis of covariance (ANCOVA) (Zar 1999) was performed for each FFS site using the post treatment stand structure and fuels measurements as response variables, with pretreatment values used as the covariate. No pretreatment data were collected at the Southern Cascades site, therefore ANOVA was completed on the post treatment data only. Bonferroni multiple pairwise comparisons (Zar 1999) were evaluated at the mean value of the covariate to determine if significant differences ($p < 0.05$) existed between treatment types and controls for stand structure and fuels variables analyzed. Normality of treatment group means and homogeneity of variance among means were assessed using the Shapiro-Wilk test and O'Brien's test, respectively. The JMP Statistical Software package (Sall et al. 2001) was used in all analyses.

Results

The fire only and mechanical plus fire treatments significantly reduced forest floor C at four of six FFS sites (Fig. 3A). At the Southern Sierra Nevada site, there was a significantly greater reduction in ground and downed woody fuel C with fall burning compared to spring burning (Fig. 3A, B). Forest floor C in mechanical only treatments was significantly greater than the controls at only one FFS site (Northern Rockies site, Fig. 3A). The C content in downed woody fuels was significantly reduced by burning alone in only two FFS sites (Central and Southern Sierra Nevada sites) (Fig. 3B).

At the Central and Southern Sierra Nevada, Blue Mountains, and Southern Cascades sites, untreated controls maintained significantly higher live tree C susceptible to >75% mortality under 97.5 percentile fire weather conditions compared to treated stands (Fig. 3C). At these same sites, there was no significant difference in live tree C susceptible to mortality among the mechanical only, mechanical followed by fire, and fire only (fall/spring) treatments. At the Southwest Plateau site, there was no significant difference between susceptible live tree C for any of the treatments (Fig. 3C). At the Northern Rockies site, the mechanical only treatment maintained the highest amount of live C at risk to loss by fire-induced mortality due to large amounts of activity fuels (Fig. 3A).

The untreated controls in the Southern Sierra Nevada site had the highest potential C emissions during wildfire (70 Mg/ha) followed by the Central Sierra Nevada site at 40 Mg/ha (both controls and mechanical only) (Fig. 4). The controls at the Blue Mountains, Northern Rockies, and Southern Cascades sites, mechanical only at the Northern Rockies site, and fall fire only at the Southern Sierra Nevada site all had potential emissions of approximately 30 Mg/ha. Potential wildfire emissions of 20 Mg/ha were estimated in the mechanical only treatments at the Blue Mountains, Southern Cascades, and Southwest Plateau sites, and spring fire only in the Southern Sierra Nevada site. The lowest emission estimates of approximately 10 Mg/ha occurred in the fire only and mechanical followed by fire treatments at the Central Sierra Nevada, Blue Mountains, Northern Rockies, Southern Cascades, and Southwest Plateau sites (Fig. 4). Fire only and mechanical followed by fire treatments reduced potential emissions by 75–80% in the Southern Cascades and Central Sierra Nevada sites, 60–67% in the Northern Rockies, Blue Mountains and Southern Sierra Nevada sites, and by approximately half at the Southern Plateau site when compared to controls at each location.

Mean removals per site from thinning operations ranged from approximately 30–60 dry Mg/ha, with Southern Cascades site having the highest removals (Fig. 5A). Of more interest, the median lives of products in first use varied considerably among sites (Fig. 5B). At the Southwest Plateau site, C lives were less than 10 years for all products (pallets and horticultural materials). In contrast, most material from the Central Sierra Nevada site was delivered to sawmills, so half of removals were converted to

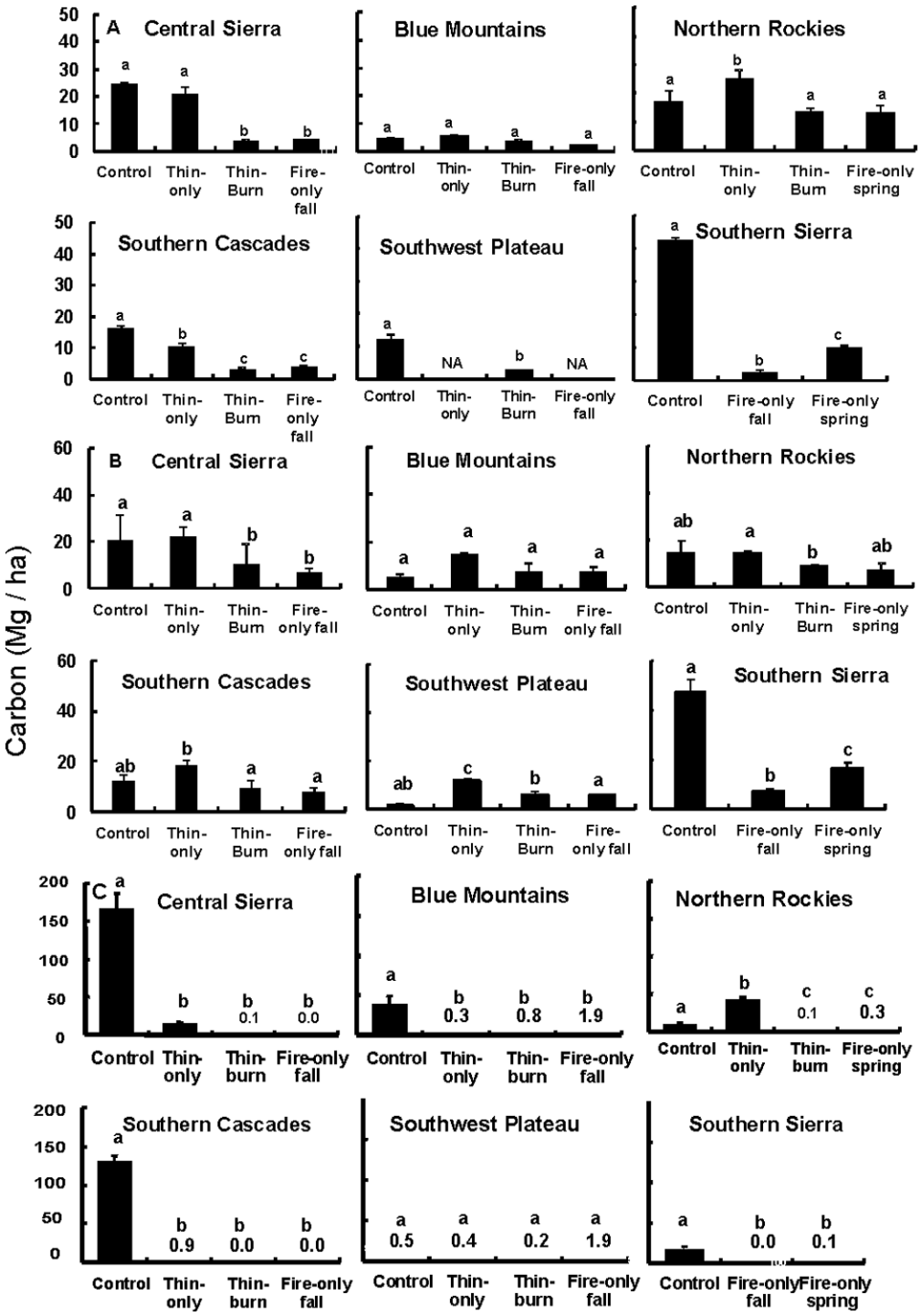

Fig. 3. Mean post-treatment carbon in (A) forest duff and litter, (B) downed woody fuels (1–1000 hour), and (C) total live tree carbon susceptible to >75% wildfire-induced mortality occurring under 97.5 percentile weather condition by treatment type in the six western U.S. Fire and Fire Surrogate study sites in Montana, Oregon, California, and Arizona. Mean values with the same letter are not significantly different ($p > 0.05$). Numerical values above very small bars are actual C values. NA, data not available.

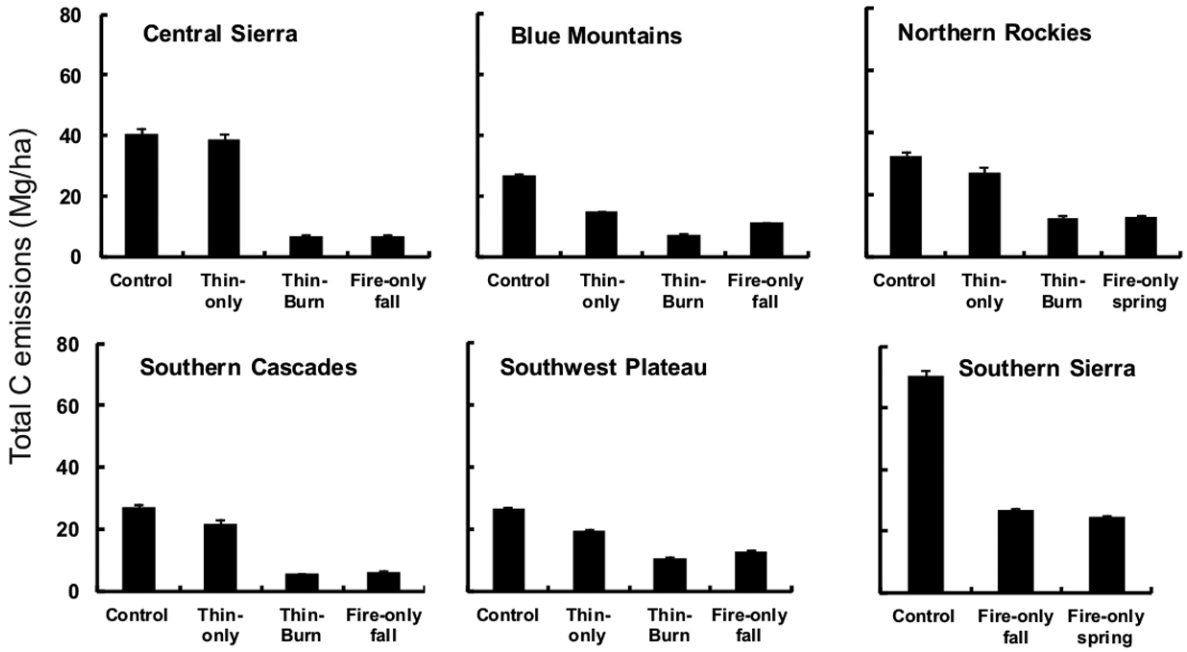

Fig. 4. Mean post-treatment C wildfire emissions estimated from FOFEM at 97.5 percentile fire weather conditions by treatment type for the six western U.S. Fire and Fire Surrogate study sites in Montana, Oregon, California, and Arizona.

construction lumber (median life = 50 years) and a quarter to energy. The other sites, where some material went to pulp mills or an energy facility, had results between those at Southwest Plateau and Central Sierra Nevada sites.

Discussion

The amount of live tree C vulnerable to wildfire-induced mortality varied from 0 to 170 Mg/ha depending on site location and treatment type, with the lowest vulnerabilities in the mechanical plus fire treatment, followed by the fire-only treatment (Fig. 3C). The mechanical-only treatment resulted in an effective reduction of potential tree mortality from wildfire compared to controls, with the exceptions of the Northern Rockies site, where potential loss increased because the harvest system used left activity fuels in the forest, and the Southwest Plateau site, where it was largely unchanged (Figs. 3 and 4). That the untreated stands at the Southern Sierra Nevada site showed a relatively low potential loss from wildfire despite having the highest live tree C stocks in the entire FFS network may seem unexpected. However, this can be explained by more of the carbon at this site being in very large (>1 m dbh), old-growth trees that are very resistant to wildfire-induced mortality even under extreme weather conditions. Other research has determined that stands with large, tall trees and few surface and/or ladder fuels are resistant to fire (Stephens and Moghaddas 2005) and store large amounts of C (Choi et al. 2006, Hurteau and North 2009, 2010, North et al. 2009).

It is important to note that our comparative analysis of C vulnerability is predicated on the occurrence of wildfire under fairly severe fire weather conditions. The fact that we do not explicitly incorporate the probability such an event occurring emphasizes that our analysis is not an actual risk assessment, i.e., expected loss multiplied by the probability of occurrence (Finney 2005). Rather, our assessment is more similar to a hazard analysis. While recent studies have included probability estimates of wildfire occurrence, and in particular occurrence of more severe fire in their assessments of fuel treatment impacts on C stocks (Ager et al. 2010, Campbell et

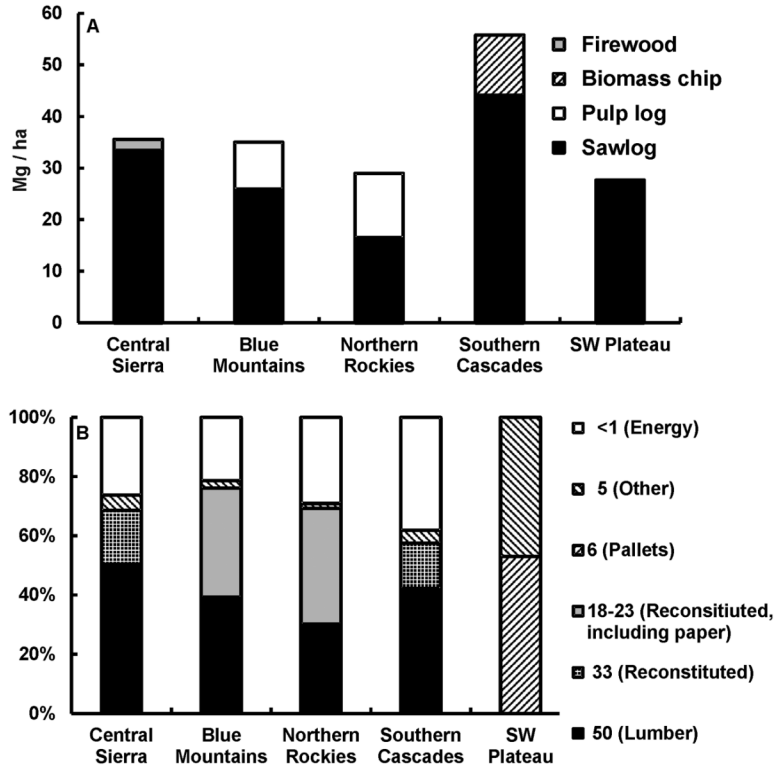

Fig. 5. Wood products removed (A) and their median life spans (years) (B) from Fire and Fire Surrogate study sites in Montana, Oregon, California, and Arizona. The Southern Sierra site is not shown because it did not include mechanical treatments. At the Southwest Plateau site all lumber produced was pallet stock and all mill residues were converted to horticultural products such as soil additives, which resulted in lower C life spans. The Northern Rockies and Blue Mountains sites have shorter median lives for reconstituted products because much of the raw material is converted to paper.

al. 2011), we submit there are several factors associated with the estimation of wildfire occurrence that lead to considerable uncertainty in calculated results. First, the use of actual wildfire occurrence over that last two to three decades to derive annual burn probabilities reflects neither the historical (pre-Euro-American) occurrence of fire in frequently burned forest types (Stephens et al. 2007, Van de Water and Safford 2011), nor the projected changes in future fire occurrence (McKenzie et al. 2004, Westerling et al. 2011). Second, changes in fire sizes resulting from fuel treatment implementation, particularly when considering the increased fire suppression efficacy associated with some fuel treatments (e.g., Moghaddas and Craggs 2007), are not evaluated. This could lead to considerable over-estimation of fire occurrence in "treated" landscapes relative to "untreated" landscapes. A third source of uncertainty in the calculation of burn probabilities is related to the potential feedbacks associated with severe fires. In many dry forest types in the western U.S. shrubs dominate for several decades following high-severity fire (McGinnis et al. 2010), and for that period of shrub-dominance the likelihood of reburning at high severity relative to intact forests is increased (Thompson and Spies 2010, van Wagtendonk et al. 2012). This potential feedback of high severity effects in initial fires leading to high severity when the same areas are reburned is not accounted for and would affect estimations of C loss and re-growth over time, particularly for "untreated" landscapes where more high severity fire would be expected.

In our research the most vulnerable sites to C losses from live tree mortality after wildfire were both in California: the Central Sierra Nevada and Southern Cascades sites (130–170 Mg/ha of

potential losses in controls). Moderately vulnerable areas (20–50 Mg/ha of potential losses) include the Blue Mountains (controls), Northern Rockies (mechanical only), Southern Sierra Nevada (controls), and Central Sierra Nevada sites (mechanical only). All other site/treatment combinations had relatively low possibilities for C loss from live tree mortality after wildfire. The site with the lowest possibilities for C loss also had the lowest total C stocks, the Southwest Plateau site (Figs. 3 and 4). The Northern Rockies site also had relatively low amounts of live tree C that were susceptible to extreme wildfire. This emphasizes a fairly intuitive but important point; different forest stands have highly variable vulnerabilities to C losses from wildfire. This could help provide information to prioritize what areas of the western U.S. would benefit most from treatments designed to reduce C losses from wildfire; all sites are not equal in terms of potential C emissions from wildfire.

Carbon emissions from wildfire had a similar pattern to that reported above regarding the potential losses of live tree C with a few exceptions. Similar to the results of the potential losses of live tree C, the fire-only and mechanical followed by fire treatments resulted in low wildfire emissions. However, the Southern Sierra Nevada site had one of the lowest potential losses of live tree C after wildfire (Fig. 3C), whereas wildfire emissions from this site were the highest estimated (Fig. 4). Even though the presence of many large, old trees made the stands more resistant to wildfire-induced mortality, these large trees also drop copious quantities of litter and wood. Indeed, the Southern Sierra site had the highest ground and surface fuel loads in the study (Fig. 3A, B), which resulted in a large proportion of the estimated emissions (Fig. 4). The mechanical only treatment in the Central Sierra Nevada site resulted in relatively low vulnerabilities of live tree C loss after wildfire but produced relatively high wildfire emissions because of high levels of activity fuels after treatment.

In addition to creating a more fire resistant forest, low intensity prescribed fires can allow surviving trees to grow at a more rapid rate, thereby increasing C sequestration over time (Hurteau and North 2010). Low intensity fires also have the advantage of reintroducing the most critical ecological processes back into these forests that have shaped them for millennia (Arno 1980, Heyerdahl et al. 2002, Grissino-Mayer and Swetnam 2000, North et al. 2005, Stephens et al. 2010).

This work did not quantify the emissions associated with implementation of fuel reduction treatments. However, the Central Sierra Nevada site used one of the most C intensive mechanical treatments in the FFS network (moderate to heavy tree thinning using chainsaws, log transport to the landing by rubber-tired or track-laying tractor, 90% of understory trees shredded in place over the whole experimental unit using a rotary masticator mounted on an excavator, log transport to mill by truck, milling logs into lumber) but this only resulted in emissions of <5 Mg of CO_2 ha^{-1} (Stephens et al. 2009*b*), which is equivalent to <2 Mg of C ha^{-1}. As the mechanical treatments used similar types of equipment across all sites, we believe that all other FFS mechanical treatments would have similar or lower mechanical emissions, which are small compared to the other C pools and emissions on these sites (Fig. 3). Other research has also determined that implementing mechanical fuel reduction treatments released relatively low amounts of C when compared to the other C pools in southwestern ponderosa pine forests (Finkral and Evans 2008).

The cost of treatments can be offset by revenues from harvesting, however this did not occur in all of our research sites (only the Central Sierra Nevada and Southern Cascades sites produced positive revenues, the Blue Mountains and Northern Rockies sites were revenue neutral) (Hartsough et al. 2008). Transaction costs to inventory all C stocks in forests can be high relative to the limited revenue provided by forest C sequestration, which can complicate operational use of C markets (Fahey et al. 2010). In practice, evaluating trade-offs between treatments will hinge on factors such as availability of treatment funds, timber and biomass markets, air quality constraints, expected treatment longevity, other management objectives such as wildlife habitat, and how much risk to residual carbon stocks landowners are willing to accept.

Fuel reduction treatments that remove materials from the site may sequester substantial amounts of C, although the residence times in first use can

vary substantially (Fig. 5B). With increased recycling, some of the C in the lumber and reconstituted products will remain in use beyond the first-use lives. In addition, as more biomass is diverted from landfills for energy, the ultimate "fate" of a higher percentage of the materials removed from the forest may be as fuel used elsewhere, thereby displacing fossil fuels to some extent.

The C in live trees killed by high severity wildfire is not emitted immediately. However, research on the decomposition of dead wood is limited in forests that once burned frequently with low-moderate intensity fire regimes. One study in southern Sierra Nevada found that the half-life of white fir boles was 14 years (Harmon et al. 1987), which is longer than the half-life of some forest products produced in this study (Fig. 5). The half-life of carbon stocks from restoration and fuels treatments will vary based on site characteristics, tree densities, machinery used, wood utilization rates, the fate of wood products, and the reduction in wildfire threat. However, there is potential for restoration and fuels treatments to play a beneficial role in reducing greenhouse gases when they reduce the threat of wildfire released carbon to the atmosphere and when carbon can be stored in wood products or be used to offset fossil fuel use (Finkral and Evans 2008).

The research sites included in this project are all in or adjacent to U.S. federal lands, particularly lands managed by the U.S. Forest Service and U.S. National Park Service. Post wildfire treatments (i.e., salvage logging, McIver and Starr (2001)) in areas that experienced large, high severity wildfire typically are completed on only a fraction of the area burned at high severity, if at all, regarding federal lands in the western US. In contrast, private timber-lands that are burned by these fires are commonly salvage logged to produce lumber, which could increase the life span of some of the sequestered carbon. This paper is most applicable to forest lands that do not receive salvage treatments; future work could quantify the effects of salvage operations on carbon sequestration. The choice to salvage log an area is complex and usually not made based on the possible impacts to carbon sequestration alone; other values (wildlife habitat) and risks (soil erosion) are frequently critical (McIver and Starr 2001).

Conclusion

Our research suggests most of the benefits of increased stand-level fire resistance can be achieved with relatively small reductions in forest C stocks. These results are applicable to dry, coniferous forests that once burned frequently with low-moderate intensity fire regimes; other forests with mixed or high severity fire regimes would have different vulnerabilities to C loss (Campbell et al. 2011). To facilitate the detection of meaningful C storage in forests, it is important to measure both changes in C stocks over time, as well as total C stocks at a point in time (Negra et al. 2008). This work focused on the latter question. Future work could evaluate how many years of post-treatment forest growth is needed to offset immediate C releases from different mechanical and prescribed fire treatments (e.g., Hurteau and North 2010) as well as the longevity of the different treatments in reducing the risk of live tree mortality from wildfire.

There is increased recognition that most low-moderate intensity fire regimes in western U.S. forests historically included some patchy high severity fire (Arno et al. 2000, Fulé et al. 2003, Baker et al. 2007, Hessburg et al. 2007, Beaty and Taylor 2008, Perry et al. 2011). As such, it may not be advantageous to reduce forest fuels to the point that all high severity fire is eliminated but most high mortality patches should be relatively small as is the case in upper mixed conifer forests in the Sierra Nevada where median high severity patch size was small (Collins and Stephens 2010). Current wildfire high severity patch sizes and areas in some forests that once burned frequently with low-moderate intensity fire regimes are well outside historical conditions (Miller et al. 2009).

Under the Climate Action Reserve's Forest Project Protocol Version 3.1 (2009) a project owner can mitigate potential C losses from wildfire by implementing fuel reduction treatments (Hurteau and North 2010). This is an improvement from previous protocols where fuel reduction treatments were classified as an immediate emission (Hurteau and North 2009) without consideration of the vulnerability of C loss from wildfire. In many forested ecosystems, frequent low intensity fires tended to maintain heterogeneous structures composed of larger

trees with low or discontinuous surface and ladder fuel loads (Weaver 1943, Keeley and Stephenson 2000, Youngblood et al. 2004, Stephens et al. 2008, North et al. 2009). In some forest ecosystems, more rather than less low-moderate intensity fire may be the key to reducing long-term C emissions and increasing C storage (Weidinmyer and Hurteau 2010).

Forest managers face an important decision: maximize C stored on site to ensure greatest short-term benefit of C sequestration and potential C related revenue, or initially remove some of that C using active treatments, including prescribed fire and mechanical methods, thereby reducing total stored C in the short term but increasing fire resistance in the long term. Our results indicate that in many dry coniferous forests of the western U.S. that once burned frequently, the latter may be a better approach for sequestering C over the long term. However, there are different vulnerabilities to C losses (Campbell et al. 2011) and differences in the median C lives of products generated from materials removed across our study sites; these could be used to help prioritize treatment implementation. Retaining or growing larger trees also reduced the vulnerability of C loss from wildfire.

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