Fuel treatment effects on stand-level carbon pools, treatment-related emissions, and fire risk in a Sierra Nevada mixed-conifer forest[1]

Scott L. Stephens, Jason J. Moghaddas, Bruce R. Hartsough, Emily E.Y. Moghaddas, and Nicholas E. Clinton

Abstract: Policies have been enacted to encourage carbon (C) sequestration through afforestation, reforestation, and other silvicultural practices; however, the effects of wildfires on forest C stocks are poorly understood. We present information from Sierran mixed-conifer forests regarding how control, mechanical, prescribed-fire, and mechanical followed by prescribed-fire treatments affected C pools. Secondly, we report CO_2 emissions from machinery and burning associated with the treatments. Lastly, the effects of treatments on the potential for C loss to wildfire are presented. The amount of aboveground C in live trees was significantly reduced in mechanical-only and mechanical plus fire treatments; C contained in dead trees was not significantly different. There was no significant difference in aboveground live and dead tree C between the fire-only and control treatments. Fire-only and mechanical plus fire treatments emitted significantly more CO_2 than the mechanical treatment and control. Modeling results for the control demonstrated 90% of the live tree C had a high (>75%) chance of being killed in a wildfire; in contrast, all three active treatments had low vulnerabilities to C loss. With wildfire severity increasing in most Sierran forests, management actions designed to increase fire resistance are justified for long-term C sequestration.

Résumé : Des politiques ont été adoptées dans le but de favoriser la séquestration du carbone (C) grâce au boisement, au reboisement et à d'autres pratiques sylvicoles. Cependant, les effets des feux de forêt sur les stocks de C dans les forêts sont mal compris. Nous présentons des informations provenant de forêts mixtes de conifères de la Sierra Nevada concernant la façon dont la mécanisation, le brûlage dirigé, la mécanisation suivie d'un brûlage dirigé ainsi qu'un traitement témoin influencent les réservoirs de C. Deuxièmement, nous rapportons les émissions de CO_2 provenant de la machinerie et du brûlage dirigé associés aux traitements. Finalement, les effets des traitements sur la perte potentielle de C due aux feux de forêt sont présentés. La quantité de C dans la partie aérienne des arbres vivants était significativement réduite dans les traitements impliquant la mécanisation seule et la mécanisation suivie du brûlage dirigé; le C contenu dans les arbres morts n'était pas significativement différent. Le contenu en C dans la partie aérienne des arbres vivants et morts n'était pas significativement différent dans le traitement impliquant le brûlage dirigé seul et le traitement témoin. Les émissions de CO_2 étaient significativement plus élevées dans le cas du brûlage dirigé seul et de la mécanisation suivie du brûlage dirigé que dans les cas de la mécanisation ou du traitement témoin. La modélisation des résultats du traitement témoin a démontré que 90 % du C dans les arbres vivants avait une forte probabilité (>75 %) d'être détruit lors d'un feu de forêt. Par contre, les trois traitements impliquant une intervention étaient peu vulnérables à la perte de C. Avec l'augmentation de la sévérité des feux de forêt, dans la plupart des forêts de la Sierra Nevada, les pratiques d'aménagement qui visent à augmenter la résistance au feu sont justifiées pour la séquestration durable du C.

[Traduit par la Rédaction]

Introduction

The use of forests to sequester carbon (C) in response to anthropogenically induced climate change is being discussed across the globe (Saikku et al. 2008). Large quantities of C can be stored or released to the atmosphere by soils and plants, and this will vary by ecoregion, vegetation type, climate, disturbance history, and land-use practices. For example, recent work has identified many old-growth forests in temperate and boreal regions as global C sinks (Luyssaert et al. 2008), but how long-term forest C stocks are influenced in regions with active fire regimes is an important question that requires further examination.

The most productive forests in California are located in the northwestern region of the state and in the Sierra Nevada. Sierra Nevada mixed-conifer forests occupy the mid-elevations and are the primary habitat for more vertebrate species than any other Sierra Nevada forest type (North et al. 2002). Over the past 150 years, forest structure in these areas has been dramatically altered by intensive logging in

Received 1 December 2008. Accepted 29 May 2009. Published on the NRC Research Press Web site at cjfr.nrc.ca on 15 August 2009.

S.L. Stephens,[2] J.J. Moghaddas, E.E.Y. Moghaddas, and N.E. Clinton. Division of Ecosystem Science, Department of Environmental Science, Policy, and Management, 137 Mulford Hall, University of California, Berkeley, CA 94720-3114, USA.
B.R. Hartsough. Department of Biological and Agricultural Engineering, University of California, Davis, CA 95616, USA.

[1]Publication No. 143 of the National Fire and Fire Surrogate Project.

[2]Corresponding author (e-mail: stephens@nature.berkeley.edu).

the early 20th century (Beesley 1996; Stephens 2000), changing climates (Millar et al. 2007), intensive forest management through the 20th century (Beesley 1996), and fire exclusion (Ritchie et al. 2007). More recently, wildfire area and severity have increased in most Sierra Nevada forests from 1984 to 2006 (Miller et al. 2008); this trend in fire severity can have a profound effect on forest C dynamics.

Fire reduces the forest C pool by consuming forest floor materials and live and dead vegetation, and releasing C to the atmosphere as carbon dioxide (CO_2), carbon monoxide, methane, other gases, and particulate matter (Stephens et al. 2007). It is critical to understand how fire will influence long-term forest C stocks in regions with active fire regimes (Hurteau and North 2009). There currently is a paradox concerning C sequestration in some California forests. Policies have been enacted to encourage C sequestration through afforestation, reforestation, and other silvicultural practices (CCAR 2007); however, the effects of future wildfires on forest C stocks, as well as management actions that can be taken to reduce C loss to wildfire, are poorly addressed by current policy. This paper provides quantitative information on C stocks in a managed forest including the effects of fuel treatments designed to reduce wildfire severity, CO_2 emissions generated by the treatments are also summarized.

Fuel treatments that incorporate thinning from below, whole-tree removal of merchantable and small-diameter material, and the use of prescribed burning to remove surface and ladder fuels have been shown to reduce potential fire severity (Fulé et al. 2001; Peterson et al. 2005; Stephens et al. 2009) and can increase fire-suppression efficiency in mixed-conifer forests of the Sierra Nevada (Moghaddas and Craggs 2007). Given the current scale of fuel-treatment implementation in the Sierra Nevada, there are few quantitative studies that evaluate the effects of these treatments on live and dead C pools in coniferous forests (Hurteau and North 2009). Implementing fuel treatments results in C emissions from the exhaust from vehicles and machinery used to remove and process noncommercial materials, as well as smoke from prescribed fires. In addition, C can be removed from treated areas in the form of commercial wood products. Gaining an understanding of how fuel treatments affect forest C pools, CO_2 emissions, and the potential for decreasing the vulnerability of live C pools due to high-severity fire is essential.

The focus of this paper is to study the effects of the most common fuel treatments in the Sierra Nevada on C pools in live biomass (trees), dead biomass (standing dead trees, surface woody debris, litter, and duff), and surface soil. Secondly, this study reports CO_2 emissions from machinery and prescribed burning associated with implementing fuel treatments. Finally, we report the effects of these treatments on the potential for loss of live C to high-severity wildfire. The hypotheses tested for this study are as follows: (*i*) there will be no significant differences in live, dead, soil, or combined C stocks between control, mechanical, mechanical plus fire, and fire-only treatments; (*ii*) there will be no significant difference in CO_2 emissions between treatments; and (*iii*) there will be no significant difference in the tonnes of live C susceptible to high severity fire between treatments.

Materials and methods

Study site

This study was conducted in the mixed-conifer zone of the north-central Sierra Nevada at the University of California Blodgett Forest Research Station (Blodgett Forest), approximately 20 km east of Georgetown, California. Blodgett Forest (38°54′45″N, 120°39′27″W) encompasses an area of 1780 ha, with elevations between 1100 and 1410 m above sea level. Tree species at Blodgett forest include sugar pine (*Pinus lambertiana* Dougl), ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.), white fir (*Abies concolor* Gord. & Glend) Lindl., incense-cedar (*Calocedrus decurrens* (Torr.) Floren.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), and California black oak (*Quercus kelloggii* Newb.). Species present in minor abundance include tanoak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehder), bush chinkapin (*Chrysolepis sempervirens* (Kell.) Hjelmg.), and Pacific madrone (*Arbutus menziesii* Pursh). Experimental units used in this research were similar in terms of stand structure and species composition. Prior to treatment, there were no significant differences in species composition, quadratic mean diameter at breast height (DBH), tree density, surface fuel loads, and canopy cover among treatment types (Stephens and Moghaddas 2005*a*).

Soils in the study area are well-developed, well-drained Haploxeralfs derived from Mesozoic granitic material. Soils are predominantly classified as the Holland and Musick series (fine-loamy, mixed, semiactive, Mesic Ultic Haploxeralfs) (Moghaddas and Stephens 2007). Soils are deep, weathered, sandy loams overlain by an organic forest floor horizon. Common soil depths range from 140 to 175 cm. Mean slopes across Blodgett Forest are <30%. Climate is Mediterranean with a summer drought period that extends into the fall. Winter and spring receive the majority of precipitation, which averages 160 cm annually (Stephens and Collins 2004). Mean temperatures in January range between 0 and 8 °C. Summer months are mild with mean August temperatures between 10 and 29 °C, with infrequent summer precipitation from thunderstorms (mean 4 cm over the summer months from 1960 to 2000) (Stephens and Collins 2004).

Fire was a common ecosystem process in the mixed-conifer forests of Blodgett Forest before the policy of fire suppression began early in the 20th century. Between 1750 and 1900, median composite fire intervals at the 3–5 ha spatial scale were 6–14 years with a fire-interval range of 2–29 years (Stephens and Collins 2004). Forested areas at Blodgett Forest have been repeatedly harvested and subjected to fire suppression for the last 100 years, reflecting a management history common to many forests in California (Laudenslayer and Darr 1990; Stephens 2000) and elsewhere in the western United States (Graham et al. 2004).

Fuel treatments

The primary objective of the treatments was to modify stand structure such that ≥80% of the dominant and codominant trees in the posttreatment stand would survive a wildfire modeled under 80th percentile weather conditions (McIver et al. 2009; Schwilk et al. 2009). The secondary objective was to create a stand structure that maintained or restored several forest attributes and processes including, but not limited to, snag and coarse woody debris recruitment (Stephens and Moghaddas 2005*b*), floral and faunal species diversity (Apigian et al. 2006; Amacher et al. 2008), and seedling establishment (Moghaddas et al. 2008). To meet these objectives, three different treatments (mechanical only, mechanical plus fire, and prescribed fire only) as well as an untreated control were each randomly applied (complete randomized design) to 12 experimental units that varied in size from 14 to 29 ha. Total area for the 12 experimental units was 225 ha. To reduce edge effects from adjoining areas, data collection was restricted to a 10 ha core area in the center of each experimental unit.

Control units received no treatment during the study period (2000–2005). Mechanical-only treatment units had a two-stage prescription. In 2001, stands were moderately to heavily thinned from below (Graham et al. 2004) to maximize crown spacing while retaining 28–34 $m^2 \cdot ha^{-1}$ of basal area (Stephens and Moghaddas 2005*a*). This mechanical treatment was implemented across all tree diameters >25 cm DBH in the mechanical-only and mechanical plus fire treatments, resulting in a significant decrease in the number of trees per hectare in the 25–51 cm and 51–76 cm DBH classes (Stephens et al. 2009). The relatively larger dominant and codominant trees were given overall preference for retention. This harvest treatment favored the removal of trees showing disease or physical damage first, followed by harvest of relatively smaller trees, then larger diameter trees, to achieve vertical and horizontal crown separation. Prescriptions favored retention of conifer species such that a relatively even proportion of each species would be represented on the study site after treatment (Stephens and Moghaddas 2005*a*). Individual trees were felled, limbed, and cut into specified log lengths using a chainsaw, and removed with either a rubber-tired or track-laying skidder for eventual processing at a sawmill. All limbwood and tops were left in the experimental unit after removal from harvested trees. During harvest, some hardwoods, primarily California black oak, were coppiced to facilitate their regeneration. In the posttreatment stand, trees were spaced such that there was little horizontal and vertical overlap of live crowns between residual intermediate, codominant, and dominant trees.

Following the harvest, approximately 90% of understory conifers and hardwoods between 2 and 25 cm DBH were masticated in place using an excavator mounted with a rotary masticator. Mastication has become a common fuel treatment in plantations and some mixed-forest stands in California because it shreds and chips standing small diameter materials, both live and dead, in place. Masticated material was not removed from the experimental units. The remaining unmasticated understory trees were left in scattered clumps of 0.04–0.20 ha in size.

Mechanical plus fire experimental units underwent the same treatment as mechanical-only units; however, in addition, they were prescribed burned using a backing fire. Fire-only units were burned with no pretreatment of fuels using strip head fires. All prescribed burning was conducted during a short period (23 October to 6 November 2002) with the majority of burning occurring at night because relative humidity, temperature, wind speed, and fuel moistures were within predetermined levels to produce the desired fire effects. Prescribed fire prescription parameters for temperature were 0–8 °C, relative humidity >35%, and wind speeds between 0.0 and 5.0 $km \cdot h^{-1}$. Desired 10 h fuel stick moisture content was 7%–10%. All prescription parameters were met with the exception of fuel moisture, which was slightly drier at 5%–9% (Kobziar et al. 2006).

Field measurements

Live and dead vegetation was measured using twenty-five 0.04 ha circular plots installed in each experimental unit (300 total plots). Individual plots were placed on a systematic 60 m grid with a random starting point. Plot centers were permanently marked with a pipe, and three witness trees were tagged to facilitate plot relocation after treatments. Tree species, DBH, total height, height to live crown base, and crown position (dominant, codominant, intermediate, and suppressed) were recorded for all trees >10 cm DBH. The same information (except crown position) was recorded for all trees >1.37 m tall on a 0.004 ha nested subplot in each of the 25 plots (Stephens and Moghaddas 2005*a*).

Surface and ground fuels were sampled with two random azimuth transects at each of the 300 plots using the line-intercept method (Brown 1974). A total of 600 fuel transects were installed. The 1 h (0–0.64 cm) and 10 h (0.64–2.54 cm) fuels were sampled from 0 to 2 m; 100 h (2.54–7.62 cm) fuels, from 0 to 3 m; and 1000 h (>7.62 cm) and larger fuels, from 0 to 11.3 m on each transect. Duff and litter depth in centimeters were measured at 0.3 and 0.9 m on each transect. Fuel depth was measured at three points along each transect. Fuel transects were sampled prior to treatment (2001), after the commercial harvest (for mechanical-only and mechanical plus fire units in 2001), after mastication (for mechanical-only and mechanical plus fire units in 2002), and 8 months after burning was completed (2003, all treatment units).

Pretreatment sampling of forest floor and mineral soil materials occurred from late May to August 2001; posttreatment sampling occurred from June to August 2003. During each sampling period, mineral soil was collected from 20 of the 0.04 ha plots within each of the 12 treatment units. At each plot, samples were pooled from six subplots for a total of 1440 subplots across the 12 treatment units. A slide hammer core sampler (262 cm^3 core volume, 15 cm core length) was used to collect soil at each subplot. Treatment effects were expected to be limited to the surface mineral soil, so samples were collected from the 0–15 cm layer.

Carbon from litter, duff, surface wood, trees, and soil

Surface and ground fuel loads were calculated using appropriate equations developed for California forests (van Wagtendonk et al. 1996, 1998). Coefficients required to calculate all surface and ground fuel loads were arithmetically weighted by the basal area fraction of each tree species to produce accurate and precise estimates of ground and surface fuel loads (Stephens 2001). Calculated surface and ground fuel loads were converted to tonnes of C per hectare.

To determine soil C content, air-dry soils were sieved to <2 mm, and a subsample was dried to constant mass at 105 °C to correct for moisture. A subsample from each soil was ground in a ball mill to pass a 60-mesh screen for determination of total C by combustion (Moghaddas and Stephens 2007). The bulk density of each sample was used to determine the C pool in the surface soil on a per-hectare basis.

Total aboveground live and dead tree biomass was calculated using equations provided in Jenkins et al. (2004). These equation sets have been used to compute aboveground tree C in other published C studies in Sierran mixed-conifer forests (Boerner et al. 2008; Hurteau and North 2009). However, it would have been preferable to use regional equations to estimate total C forest stocks because they should be more accurate, but they were not available for all species studied.

Equipment and milling emissions

Chainsaws were used to fell, limb, and buck the trees within the stands and, to a limited extent, on each landing. The number of scheduled hours for each saw operator on each unit was manually recorded. Gasoline consumption was estimated by assuming one tank of fuel was burned during each scheduled hour.

Electronic activity recorders were installed on all pieces of harvesting equipment — for skidding and loading of logs, construction and maintenance of roads and trails and for mastication of small trees and residues (Hartsough et al. 2008). The recorders sensed vibration while the equipment was productively operating and recorded the number of operating minutes within each hour, as well as the date and time. The date and time each machine began and ended operations in each unit was manually recorded, and this information was used to allocate the productive machine hours to each unit. Diesel fuel consumption per productive hour was estimated with coefficients reported by Brinker et al. (2002) for various categories of equipment and the manufacturers' power ratings for the machines used on the experimental units.

We recorded the sawmill destination (three mills received material from the study units) for each load of logs from each unit. To estimate the amount of fuel used during transport, the total round-trip distance to deliver all the material from each unit was multiplied by the mean diesel fuel consumption for heavy trucks as reported by the US Department of Commerce 2002 vehicle inventory and use survey (Davis et al. 2008).

Fuel consumption rates were converted to CO_2 emissions by using the mean C contents of gasoline and diesel and the oxidation factor reported by the US Environmental Protection Agency (EPA 2005) as well as the molecular mass ratio of CO_2 to C. Emissions for all harvesting operations from the stump to the truck were summed for each experimental unit; those for mastication and transport were kept separate to show the relative contributions. Total amount of C used was divided by the treated area of the unit to obtain tonnes of CO_2 emitted per hectare treated.

Milling emissions were estimated using a coefficient derived from actual mill operations in the Pacific Northwest. Mill emissions averaged 0.13 t of CO_2 per 1000 board feet (T. Collins, personal communication, 2008). This coefficient was applied to actual harvest volumes for the mechanical and mechanical plus fire treatments.

Fate of carbon in harvested logs

Although harvesting removes C from the stand, some fraction of that C may remain sequestered for a substantial amount of time, depending on the types of products generated (Finkral and Evans 2008). As with other aspects of forestry, accounting for various products and C residence times is not a precise science, and various methods have been proposed (Lim et al. 1999). For this study, the fractions of log mass converted to various products were estimated from published studies of sawmill surveys in California and the Pacific Northwest (Morgan et al. 2004; Milota et al. 2005). The lives of these products were estimated from historical use in the United States of products for various uses, such as single-family homes (Winjum et al. 1998; McKeever 2002; Winistorfer et al. 2005; Skog 2008). These were used to estimate the fraction of C that would be emitted within 1 year of harvest and a range of sequestration periods for the remainder.

Prescribed fire emissions

Emissions were computed directly from consumed fuel loads as determined from the difference between pre- and post-burn loads measured in the field; the methods used are described by Clinton et al. (2006). Ratios of flaming to smoldering combustion phase were assumed to correspond to "wet" fuel conditions for the fuels ≥7.5 cm and "dry" for all other fuels. The combustion efficiency in each phase, as well as the precise ratio of phases is given in Clinton et al. (2006). Combustion efficiency was related to an emission factor in terms of grams of CO_2 per kilogram of fuel consumed using Ward and Hardy (1991, eq. 5).

Prediction of potential fire-related mortality

Fuels Management Analyst Plus (FMA) was used to estimate potential tree mortality from wildfire (Carlton 2004). FMA uses information from field measurements (tree species, DBH, tree crown ratio, tree crown position, percentage canopy cover, surface fuel loads, and slope) and fire weather (97.5th percentile: extreme conditions) to simulate fire effects at the stand scale. FMA incorporates published methodologies for computing crown bulk density, fire behavior, and predicted mortality by species. See Stephens and Moghaddas (2005*a*, 2005*c*) for summaries of the methodologies used for these computations.

Data analysis

Treatment effects on C pools were evaluated using analysis of covariance (ANCOVA). To remove the influence of pretreatment differences among treatment groups, the pretreatment data was modeled as a covariable. Interaction effects were tested by adding a treatment × pretreatment term. Differences were considered to be significant at the $p < 0.05$ level. If differences among treatments were significant, the Tukey–Kramer honest significant difference (HSD) test was used to make multiple comparisons among treatment groups (Sall et al. 2001). Normality of treatment group means and homogeneity of variance among means were assessed using the Shapiro–Wilk test and O'Brien's test, respectively. Emissions from treatments were analyzed using an analysis of variance (ANOVA). All analyses were conducted using JMPIN statistical software (Sall et al. 2001).

Results

Live, dead, and soil carbon pools and treatment emissions

The amount of aboveground C contained in live trees and live and dead trees combined was significantly reduced in mechanical-only and mechanical plus fire treatments (Fig. 1). Carbon contained in dead trees was not significantly different between treatments. There was no significant difference in aboveground live and dead tree C between the fire-only and control treatments.

Litter, duff, and all surface deadwood were significantly reduced in the mechanical plus fire and fire-only treatments (Fig. 2) but were not significantly different between the control and mechanical-only treatment. There was no significant difference in surface soil C among treatments (Fig. 3A). Total C, which was assessed as the combined sum of the aboveground live and dead trees, surface soil, and surface wood, was significantly lower in the mechanical plus fire treatment when compared with all other treatments (Fig. 3B). Total C was not significantly different between the mechanical-only and fire-only treatments, but both of these were significantly lower than the control. The fire-only and mechanical plus fire treatments emitted significantly more CO_2 than the mechanical-only treatment and control (Fig. 4). In the mechanical-only and mechanical plus fire treatments, emissions from milling accounted for 1.2 and 1.5 $t{\cdot}ha^{-1}$ of CO_2, receptively. All three active treatments had significantly less live C susceptible to mortality under 97.5th percentile weather conditions compared with the untreated control (Fig. 5).

Discussion

There is scientific consensus that climate change will result in periods of prolonged drought, which will result in potential increases in tree mortality (Das et al. 2007; van Mantgem et al. 2009), annual number of wildfires (Fried et al. 2008), length of fire season (Westerling et al. 2006), and fire severity (Miller et al. 2008). At the same time, there are intensive policy efforts underway in California (e.g., Assembly Bill 32) to assess and use these same coniferous forests for C storage in an effort to mitigate anthropogenic CO_2 emissions and reduce atmospheric C concentrations (California State Assembly 2006).

Carbon pools in live and dead trees

Overall, the mechanical-only and mechanical plus fire treatments removed the greatest amount of live tree C in the form of tree boles harvested or masticated as part of the treatment. Within these treatments, a mean of 31.7 $t{\cdot}ha^{-1}$ of live C was removed from mechanically treated units and transported to a mill for processing into wood products (more information below on products). An additional 8.8 $t{\cdot}ha^{-1}$ was converted from live tree C to slash and chips and left on site by the mastication treatment. In the fire-only treatment, 350 $snags{\cdot}ha^{-1}$ of <30.0 cm DBH remained after the burn treatment (Stephens and Moghaddas 2005*b*), adding to the deadwood pool. At the same time, there was no significant change in snags ≥30 cm DBH resulting from the prescribed fire, leading to little net change in large standing dead trees (Stephens and Moghaddas 2005*b*). Although the overall number of standing dead trees increased dramatically following the fire only treatment, most were small in size and contributed relatively small amounts to the C pool of dead trees. As a result, the C pool in dead trees did not differ significantly among treatments. Meanwhile, C stored in live trees did not differ between the fire-only treatment and control. Carbon storage increases substantially as tree diameter and height increase. Few dominant and codominant trees were killed by the fire-only treatment resulting in only minor changes in the C pool of live trees. Because dominant and codominant trees store the majority of the C, many suppressed and intermediate trees can be removed without great impacts to C stocks.

Carbon pools in surface materials and soil

Because of the dry surface fuel conditions at the time of the prescribed fire, litter, duff, and dead surface wood were significantly reduced in both the fire-only and mechanical plus fire treatments. The prescribed fires consumed >75% of the C stored in litter, duff, and surface wood. Despite these large losses from the organic horizons, C pools in surface mineral soil to 15 cm was not significantly changed by any of the treatments (Fig. 3). Mineral soil C does not represent the entire belowground C pool. Many fine roots were included in the mineral soil samples but sieving removed the coarse root fraction, which is not accounted for here. Coarse roots do contribute small amounts of biomass and C within the surface soil. Page-Dumroese and Jurgensen (2006) measured live roots in surface soils across a range of middle to late successional forest stands at 14 sites in the western United States. Root C (from all roots — fine, medium, and coarse) in the upper 30 cm of soil ranged from about 0.2 to 0.8 $t{\cdot}ha^{-1}$ across sites. This accounted for only about 0.1%–2.3% of the total C in the upper 30 cm of mineral soil at each site. The work presented here and other recent studies (Moghaddas and Stephens 2007) quantify the initial effects of fuel treatments on soil C, but the long-term effects of fuel management actions on soil C are poorly understood and require more research (Misson et al. 2005; Kobziar and Stephens 2006; Woodbury et al. 2007). Although a prescribed fire conducted under moister spring conditions resulted in reduced consumption of surface fuel (Knapp et al. 2004), the prescribed fires in this study were implemented during the historic fire season for this forest (prior to the onset of fall precipitation; Stephens and Collins 2004) and burned the majority of ground and surface fuels. This work did not estimate the C fraction that could be transformed by fire into relatively inert forms (black carbon) that could be a significant C sink (Kuhlbusch and Crutzen 1995).

All active treatments resulted in significantly reduced total C stocks, but the source and extent of the C loss varied by treatment. In the mechanical treatment, C was reduced primarily by the removal of live trees. In the fire-only treatment, the greatest source of C loss was through the combustion of litter, duff, and dead surface wood; together, these accounted for approximately 14% of the total C assessed in these plots. In the mechanical plus fire treatment, similar amounts of C were lost from live tree removal and surface- and ground-fuel consumption from fire. Whereas a portion of the C in the commercially harvested trees will be stored

Fig. 1. Mean total aboveground carbon for (A) live, (B) dead, and (C) all trees combined among the three treatments and controls. Bars with the same letter are not significantly different (ANCOVA, $p > 0.05$). Error bars are SEs. Mech, mechanical treatment.

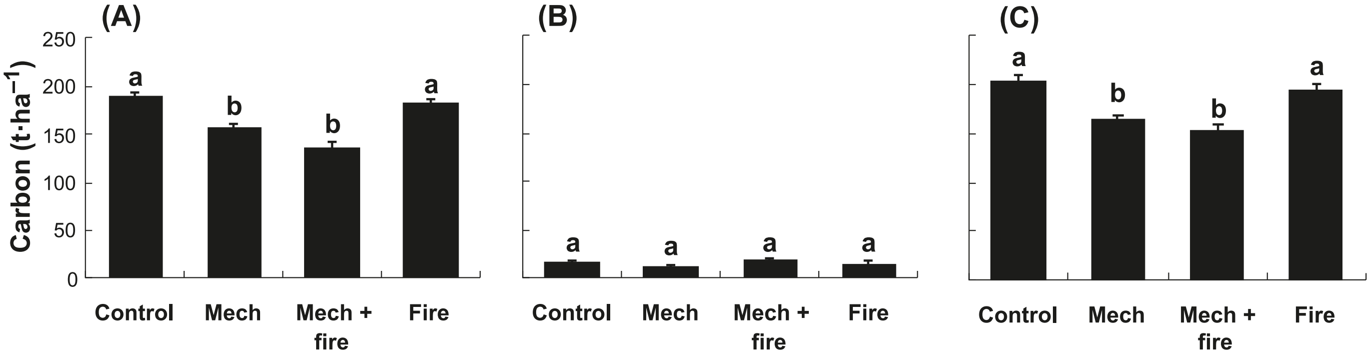


Fig. 2. Mean total carbon for (A) litter and duff, (B) surface deadwood, and (C) all combined. Bars with the same letter are not significantly different ($p > 0.05$). Error bars are SEs. Mech, mechanical treatment.

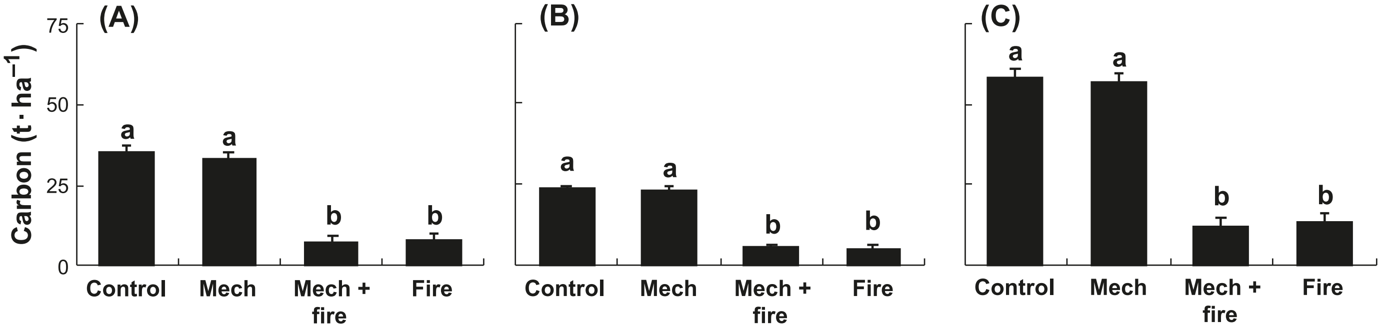


Fig. 3. Mean (A) total soil carbon to 15 cm and (B) total combined carbon for all trees, soil, and surface wood. Bars with the same letter are not significantly different (ANCOVA, $p > 0.05$). Error bars are SEs. Mech, mechanical treatment.

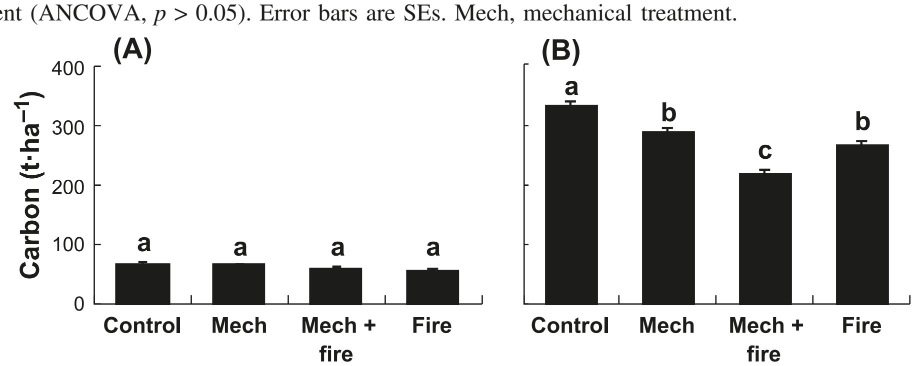


Fig. 4. Mean total CO_2 emissions directly resulting from treatment and milling activity. Bars with the same letter are not significantly different (ANOVA, $p > 0.05$). Error bars are SEs. Mech, mechanical treatment.

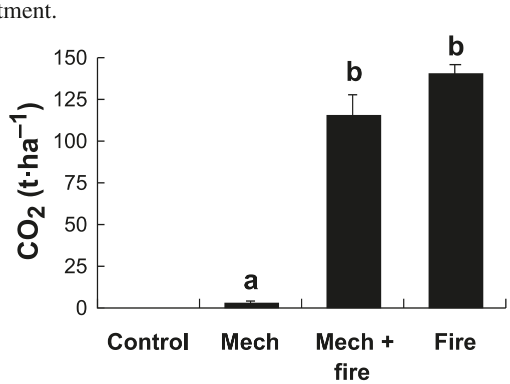
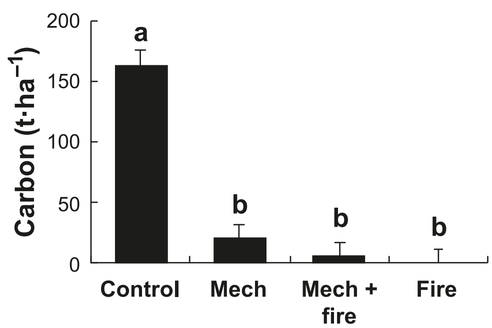


Fig. 5. Mean total live tree carbon with a >75% chance of mortality in a wildfire occurring under 97.5th percentile weather conditions. Bars with the same letter are not significantly different (ANCOVA, $p > 0.05$). Error bars are SEs. Mech, mechanical treatment.



in wood products, C lost during the prescribed burns was released to the atmosphere immediately.

Treatment-related CO_2 emissions

Total CO_2 emissions were greatest in the mechanical plus fire and fire-only treatments. This was directly due to the conversion of surface litter, duff, and deadwood to CO_2 via combustion during the prescribed burns. The emissions in the mechanical-only treatment reflect harvest, log loading, transport, and milling-related emissions, which were significantly lower than emissions for both treatments that utilized prescribed fire.

The increased amount of CO_2 emitted from the fire-only units, which were not mechanically manipulated before burning, is most likely due to the presence of an intact, or otherwise less disturbed, duff layer. Burning of this layer can account for a substantial amount of emissions (Clinton et al. 2006). Skid trails in the mechanical plus fire units reduced fuel continuity and resulted in less area burned (Moghaddas and Stephens 2007). Despite the fact that the mechanical plus fire units had a higher amount of large (>7.5 cm diameter) woody debris and other slash, this fuel is of insufficient quantity to offset the effects of disturbance of the duff and fine fuels on the forest floor.

Wood products

Results of previous studies indicate that fractions of the material from logs entering a sawmill in California will be converted into several products with different estimated median lives in the first use of those products (Table 1). These figures indicate that approximately one-third of the C from the logs will be emitted in rather short order, although the fuel fraction will probably have substituted for emissions from fossil sources. Much of the sawn and reconstituted product will sequester C for substantially longer periods, in its first use. Beyond the first use, most of the material will probably be utilized for energy, recycled, or landfilled. Skog (2008) estimated that, in 2005, 14% of discarded wood was burned, 9% was recycled, and 67% was landfilled (minor amounts were composted or disposed of in surface dumps.) With current trends, it is likely that more material will be used in some way rather than being landfilled. If burned for energy, it will offset other C emissions to some extent. If reused or landfilled, it will continue to sequester C. In fact, Skog (2008) estimated that three-quarters of wood products deposited in landfills will not decay because of the anaerobic conditions.

Table 1. Wood products and their median life from logs entering sawmills in California.

Product	Proportion of sawlog used, by mass	Median life (years)
Lumber	0.53	50
Reconstituted*	0.19	33
Fuel	0.22	0
Other*	0.05	5
Unused	0.01	5

Note: Data are from McKeever (2002), Milota et al. (2005), Morgan et al. (2004), Skog (2008), Winistorfer et al. (2005), and Winjum et al. (1998).

*Reconstituted products in California include particleboard, paper, medium-density fiberboard, and hardboard. Other products include landscaping bark, soil additives, and livestock bedding.

Potential fire-related mortality

In this study, the control treatment stored the greatest amount of C but was at greatest risk of losing that C during a severe wildfire, both in the surface dead material and live trees. Modeling results for the control demonstrated that 90% of the live tree C pool had a high (>75%) chance of being killed in severe wildfire. In addition, results from this study indicate that burning treatments alone or those that utilized moderate to heavy thinning from below combined with prescribed fire were most effective at protecting on-site live C from direct mortality due to wildfire; the mechanical-only treatment also reduced the vulnerability of C loss by wildfire. These findings corroborate with those reported by Hurteau and North (2009), who determined that untreated stands stored more C but were at higher risk to high-severity fire than those treated to create a low stand density dominated by large fire-resistant pines. This type of fuel treatment (prescribed fire alone or thinning from below followed by prescribed fire) has been shown to reduce fire severity in modeled (Fulé et al. 2001; Stephens et al. 2009) and real wildfire conditions (Ritchie et al. 2007) and reflects the basic principles of fuel reduction (Agee and Skinner 2005).

Given that California protocols require that C sold be stored for a period of 100 years (CCAR 2007), there is a relatively high potential for this site to be impacted by a wildfire under extreme weather conditions during this period. When trees are killed by wildfire or other causes, their stored C is not immediately released to the atmosphere; rather, the C is slowly released as they decay over many years. Treatments that directly released the most CO_2 also protected the greatest amount of C in the form of live trees from potential wildfire mortality. However, it is also important to look at the fate of C pools. The mechanical treatment released <1 t C·ha^{-1} in emissions, but 20 t·ha^{-1} of live C were predicted to be killed in a wildfire. The mechanical plus fire treatment released approximately 31 t C·ha^{-1} with very little live C loss to future wildfire mortality. The fire-only treatment released 38 t C·ha^{-1}, and very little loss of live tree C from was projected from a future wildfire.

It is important to note that the longevity of the effect of fuel treatments can vary from 5 to 20 years depending on treatment and vegetation type. The fire-only treatments used in this study are scheduled to be burned for the second time in the fall of 2009; these fires should consume the majority of dead and down fuels created by prescribed fire mortality (2002 fires) and should maintain high fire resistance. An evaluation of the mechanical plus fire treatments resulted in no need for a second fire at this time. Very little tree mortality resulted from prescribed fires because they were thinned (crown thinning and thinning from below) and masticated before fire was applied. It is estimated that these stands will not require reburning for approximately another decade. The evaluation of future entries in all active treatments will continue with the goal of maintaining or increasing C stocks over the long term. Although the fuel treatments applied in this study increased forest resistance to fire, they may not have created conditions for successful regeneration of all mixed-conifer species (Moghaddas et al. 2008) emphasizing that fire hazard reduction should not be the sole management objective.

Efforts to manage forests for C sequestration present a unique and complex opportunity for today's forest managers. There is approximately 1×10^9 t of C accumulated in live tree biomass and 100×10^6 t of C in snags and down deadwood stored in California (Christensen et al. 2008), with nearly 55% of that live biomass found on public lands. Of C on public lands, nearly 24% of that area is managed in reserve status (Christensen et al. 2008) where fuel treatment opportunities are more limited; however, wildland fire use could be used in remote areas to reduce fire hazards and increase forest resiliency (Collins and Stephens 2007; Collins et al. 2009). At the same time, recent studies indicate that such forests managed at high density with high surface fuel loads are susceptible to high-severity fire (Stephens et al. 2009) and that prevalence of high-severity fire is increasing in mixed-conifer forests in the Sierra Nevada (Miller et al. 2008).

Current California methodology for C accounting requires that harvest stock loss be treated as an immediate emission (CCAR 2007). However, accounting for emissions from wildfire is not required, and if a wildfire does occur, the California Climate Change Action Registry requires that only the baseline be recalculated for the disturbed site (Hurteau and North 2009). A more complete accounting would include C released from wildfire events, similar to the Intergovernmental Panel on Climate Change (IPCC 2006) guidelines, and accounting for some harvested material being sequestered for substantial periods as wood products.

Conclusion

Forest managers face an important decision: should C stored on site be maximized to ensure greatest short term benefit of C sequestration and potential C-related revenue, or should some of that C be removed using active treatments, including prescribed fire, mechanical thinning from below, and mastication, thereby reducing total stored C in the short term but increasing fire resistance in the long term? Results from this study indicate that in fire-prone dry coniferous forests of the western United States that once burned frequently, the latter is the more prudent approach to storing C over the long term in these ecosystems. Within the context of historical fire emissions, today's emissions from prescribed fire and wildfire, combined, are substantially less than those reported under the historical fire regime (Stephens et al. 2007). Previously published work by Narayan et al. (2007) suggests that the use of prescribed burning as a mitigation for potential wildfire CO_2 emissions is a valid approach to reducing overall greenhouse gases under the Kyoto Protocol.

Forest C risk to wildfire will vary a great deal in California, with northwestern forests (dominated by coast redwood (*Sequoia sempervirens* (D. Don) Endl.) and mesic Douglas-fir) having relatively low risk of high-severity fire (Mitchell et al. 2009) in comparison with forests in the Sierra Nevada, southern Cascades, Klamath, Coast Ranges, Modoc, and southern California mountains. Therefore, forest C protocols should vary across the state to reflect this difference in fire risk. Stand structure in forests being used to store C should maintain a configuration of live and dead C pools that reduces the risk of high-severity fire. Changing climates will influence all forests in the state (Millar et al. 2007); as outputs from regional general circulation models become available, these should be used to evaluate forest-fire resistance into the future and how management actions can influence C sequestration.

Overall applicability of these results should be directed to dry coniferous forests and, more specifically, to young growth stands of Sierran mixed-conifer forest with high growth productivity (Olson and Helms 1996), which have been actively managed for timber harvest in the past. To facilitate the detection of meaningful patterns in C storage in forests, it is important to measure both changes in C stocks over time as well as total C stocks (Negra et al. 2008). This work focused on the latter question; future work should investigate how C stocks change over time in response to fire hazard reduction treatments.

Acknowledgements

This project was partially funded by the USDA–USDI Joint Fire Sciences Program. Special thanks to Blodgett Forest Research Station (especially Bob Heald) and their dedicated staff and to all those who helped implement the prescribed burns. We also thank all graduate students and summer field technicians who were integral to this project.

References

Agee, J.K., and Skinner, C.N. 2005. Basic principles of forest fuel reduction treatments. For. Ecol. Manage. **211**(1–2): 83–96. doi:10.1016/j.foreco.2005.01.034.

Amacher, A.J., Barrett, R.H., Moghaddas, J.J., and Stephens, S.L. 2008. Preliminary effects of fire and mechanical fuel treatments on the abundance of small mammals in the mixed-conifer forest of the Sierra Nevada. For. Ecol. Manage. **255**(8–9): 3193–3202. doi:10.1016/j.foreco.2007.10.059.

Apigian, K., Dahlsten, D., and Stephens, S.L. 2006. Fire and fire surrogate treatment effects on leaf litter arthropods in a western Sierra Nevada mixed-conifer forest. For. Ecol. Manage. **221**: 110–122.

Beesley, D. 1996. Reconstructing the landscape: an environmental history, 1820–1960. *In* Sierra Nevada Ecosystem Project: final report to Congress. Vol. II. Assessments and scientific basis for management options. Centers of Water and Wildland Resources, Davis, Calif. pp. 1–24.

Boerner, R.E., Huang, J., and Hart, S.C. 2008. Fire, thinning, and the carbon economy: effects of fire and fire surrogate treatments on estimated carbon storage and sequestration rate. For. Ecol. Manage. **255**(8–9): 3081–3097. doi:10.1016/j.foreco.2007.11.021.

Brinker, R.W., Kinard, J., Rummer, B., and Lanford, B. 2002. Machine rates for selected forest harvesting machines. Alabama Agricultural Experiment Station Auburn, Ala. Circ. 296.

Brown, J.K. 1974. Handbook for inventorying downed woody material. USDA For. Serv. Gen. Tech. Rep. INT-16.

California State Assembly. 2006. California global warming solutions act of 2006. State of California, Sacramento, Calif. Bill 32.

Carlton, D. 2004. Fuels management analyst plus software, version 3.8.19 edition [computer program]. Fire Program Solutions, LLC, Estacada, Ore.

Christensen, G.A., Campbell, S.J., and Fried, J.S. (*Technical Edi-*

tors). 2008. California's forest resources, 2001–2005: five-year forest inventory and analysis report. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-763.

Climate Change Action Registry (CCAR). 2007. California Climate Change Action Registry, forest sector protocol [online]. Available from www.climateregistry.org/tools/protocols/industry-specific-protocols.html [accessed 13 October 2008].

Clinton, N.E., Gong, P., and Scott, K. 2006. Quantification of pollutants emitted from very large wildland fires in southern California, USA. Atmos. Environ. **40**(20): 3686–3695. doi:10.1016/j.atmosenv.2006.02.016.

Collins, B.M., and Stephens, S.L. 2007. Managing natural fires in Sierra Nevada wilderness areas. Front. Ecol. Environ. **5**(10): 523–527. doi:10.1890/070007.

Collins, B.M., Miller, J.D., Thode, A.E., Kelly, M., van Wagtendonk, J., and Stephens, S.L. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems (New York, Print), **12**(1): 114–128. doi:10.1007/s10021-008-9211-7.

Das, A.J., Battles, J.J., Stephenson, N.L., and van Mantgem, P.J. 2007. The relationship between tree growth patterns and likelihood of mortality: a study of two tree species in the Sierra Nevada. Can. J. For. Res. **37**(3): 580–597. doi:10.1139/X06-262.

Davis, S.C., Diegel, S.W., and Boundy, R.G. 2008. Transportation energy data book: edition 27. Table 5.4. Truck statistics by gross vehicle weight class, 2002. Oak Ridge National Laboratory, Oak Ridge, Tenn.

Finkral, A.J., and Evans, A.M. 2008. The effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. For. Ecol. Manage. **255**(7): 2743–2750. doi:10.1016/j.foreco.2008.01.041.

Fried, J.S., Gilless, J.K., Riley, W.J., Moody, T.J., Simon de Blas, C., Hayhoe, K., Moritz, M., Stephens, S., and Torn, M. 2008. Predicting the effect of climate change on wildfire behavior and initial attack success. Clim. Change, **87**(Suppl. 1): 251–264. doi:10.1007/s10584-007-9360-2.

Fulé, P.Z., McHugh, C., Heinlein, T.A., and Covington, W.W. 2001. Potential fire behavior is reduced following forest restoration treatments. *In* Ponderosa pine ecosystems restoration and conservation: steps towards stewardship. *Edited by* R.K. Vance, C.B. Edminster, W.W. Covington, and J.A. Blake. USDA For. Serv. Rocky Mountain Res. Stn. Proc. RMRS-22. pp. 28–35.

Graham, R.T., McCaffrey, S., and Jain, T.B. (*Technical Editors*). 2004. Science basis for changing forest structure to modify wildfire behavior and severity. USDA For. Serv. Gen. Tech. Rep. RMRS-120.

Hartsough, B.R., Abrams, S., Barbour, R.J., Drews, E.S., McIver, J.D., Moghaddas, J.J., Schwilk, D.W., and Stephens, S.L. 2008. The economics of alternative fuel reduction treatments in western United States dry forests: financial and policy implications from the National Fire and Fire Surrogate Study. Forest Policy Econ. **10**(6): 344–354. doi:10.1016/j.forpol.2008.02.001.

Hurteau, M., and North, M. 2009. Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. Front. Ecol. Environ. **7**: 2–7.

Intergovernmental Panel on Climate Change (IPCC). 2006. IPCC guidelines for national greenhouse gas inventories, prepared by the National Greenhouse Gas Inventories Programme [online]. IPCC, Vienna. Available from www.ipcc-nggip.iges.or.jp/public/2006gl/index.html [accessed 13 October 2008].

Jenkins, J.C., Chojnacky, D.C., Heath, L.S., and Birdsey, R.A. 2004. Comprehensive database of diameter-based biomass regressions for North American tree species. USDA For. Serv. Gen. Tech. Rep. NE-319.

Knapp, E., Keeley, J.E., Ballenger, E.A., and Brennan, T.J. 2005. Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. For. Ecol. Manage. **208**(1–3): 383–397. doi:10.1016/j.foreco.2005.01.016.

Kobziar, L., and Stephens, S.L. 2006. The effects of fuels treatments on soil carbon respiration in a Sierra Nevada pine plantation. Agric. For. Meteorol. **141**(2–4): 161–178. doi:10.1016/j.agrformet.2006.09.008.

Kobziar, L., Moghaddas, J.J., and Stephens, S.L. 2006. Tree mortality patterns following prescribed fires in a mixed conifer forest. Can. J. For. Res. **36**(12): 3222–3238. doi:10.1139/X06-183.

Kuhlbusch, T.A.J., and Crutzen, P.J. 1995. Toward a global estimate of black carbon in residues of vegetation fires representing a sink of atmospheric CO_2 and a source of O_2. Global Biogeochem. Cycles, **9**(4): 491–501. doi:10.1029/95GB02742.

Laudenslayer, W.F., and Darr, H.H. 1990. Historical effects of logging on forests of the Cascade and Sierra Nevada Ranges of California. Trans. West. Sect. Wildl. Soc. **26**: 12–23.

Lim, B., Brown, S., and Schlamadinger, B. 1999. Carbon accounting for forest harvesting and wood products: a review and evaluation of different approaches. Environ. Sci. Policy, **2**(2): 207–216. doi:10.1016/S1462-9011(99)00031-3.

Luyssaert, S., Schulze, E.D., Börner, A., Knohl, A., Hessenmöller, D., Law, B.E., Ciais, P., and Grace, J. 2008. Old-growth forests as global carbon sinks. Nature (Lond.), **455**(7210): 213–215. doi:10.1038/nature07276. PMID:18784722.

McIver, J., Youngblood, A., and Stephens, S.L. 2009. The National Fire and Fire Surrogate study: ecological consequences of fuel reduction methods in seasonally dry forests. Ecol. Appl. **19**(2): 283–284. doi:10.1890/07-1785.1. PMID:19323190.

McKeever, D.B. 2002. Domestic market activity in solid wood products in the United States, 1950–1998. USDA For. Serv. Gen. Tech. Rep. PNW-524.

Millar, C.I., Stephenson, N.L., and Stephens, S.L. 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecol. Appl. **17**(8): 2145–2151. doi:10.1890/06-1715.1. PMID:18213958.

Miller, J.D., Safford, H.D., Crimmins, M., and Thode, A.E. 2008. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems (New York, Print), **12**: 16–32. doi:10.1007/s10021-008-9201-9.

Milota, M., West, C., and Hartley, I. 2005. Gate-to-gate life-cycle inventory of softwood lumber production. Wood Fiber Sci. **37**: 47–57.

Misson, L., Tang, J., Xu, M., McKay, M., and Goldstein, A. 2005. Influences of recovery from clear-cut, climate variability, and thinning on the carbon balance of a young ponderosa pine plantation. Agric. For. Meteorol. **130**(3–4): 207–222. doi:10.1016/j.agrformet.2005.04.001.

Mitchell, S.R., Harmon, M.E., and O'Connell, K.E.B. 2009. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. Ecol. Appl. **19**(3): 643–655.

Moghaddas, J.J., and Craggs, L. 2007. A fuel treatment reduces fire severity and increases suppression efficiency in a mixed conifer forest. Int. J. Wildland Fire, **16**(6): 673–678. doi:10.1071/WF06066.

Moghaddas, E.E.Y., and Stephens, S.L. 2007. Thinning, burning, and thin-burn fuel treatment effects on soil properties in a Sierra Nevada mixed-conifer forest. For. Ecol. Manage. **250**(3): 156–166. doi:10.1016/j.foreco.2007.05.011.

Moghaddas, J.J., York, R.A., and Stephens, S.L. 2008. Initial response of conifer and California black oak seedlings following fuel reduction activities in a Sierra Nevada mixed conifer forest. For. Ecol. Manage. **255**(8–9): 3141–3150. doi:10.1016/j.foreco.2007.11.009.

Morgan, T.A., Keegan, C.E., III, Dillon, T., Chase, A.L., Fried, J.S., and Weber, M.N. 2004. California's forest products industry: a descriptive analysis. USDA For. Serv. Gen. Tech. Rep. PNW-615.

Narayan, C., Fernandes, P.M., van Brusselen, J., and Schuck, A. 2007. Potential for CO_2 emissions mitigation in Europe through prescribed burning in the context of the Kyoto Protocol. For. Ecol. Manage. **251**(3): 164–173. doi:10.1016/j.foreco.2007.06.042.

Negra, C., Sweedo, C.C., Cavender-Bares, K., and O'Malley, R. 2008. Indicators of carbon storage in U.S. ecosystems: baseline for terrestrial carbon accounting. J. Environ. Qual. **37**(4): 1376–1382. doi:10.2134/jeq2007.0290. PMID:18574168.

North, M., Oakley, B., Chen, J., Erickson, H., Gray, A., Izzo, A., Johnson, D., Ma, S., Marra, J., Meyer, M., Purcell, K., Rambo, T., Rizzo, D., Roath, B., and Schowalter, T. 2002. Vegetation and ecological characteristics of mixed-conifer and red fir forests at the Teakettle Experimental Forest. USDA For. Serv. Gen. Tech. Rep. PSW-186.

Olson, C., and Helms, J.A. 1996. Forest growth and stand structure at Blodgett Forest Research Station 1933–1995. *In* Sierra Nevada Ecosystem Project: final report to Congress. Vol. III. Assessments and scientific basis for management options. Centers of Water and Wildland Resources, Davis, Calif. pp. 681–732.

Page-Dumroese, D.S., and Jurgensen, M.F. 2006. Soil carbon and nitrogen pools in mid- to late-successional forest stands of the northwestern United States: potential impact of fire. Can. J. For. Res. **36**(9): 2270–2284. doi:10.1139/X06-125.

Peterson, D.L., Johnson, M.C., Agee, J.K., Jain, T.B., McKenzie, D., and Reinhardt, E.D. 2005. Forest structure and fire hazard in dry forests of the western United States. USDA For. Serv. Gen. Tech. Rep. PNW-268.

Ritchie, M.W., Skinner, C.N., and Hamilton, T.A. 2007. Probability of tree survival after wildfire in an interior pine forest of northern California: effects of thinning and prescribed fire. For. Ecol. Manage. **247**(1–3): 200–208. doi:10.1016/j.foreco.2007.04.044.

Saikku, L., Rautiainen, A., and Kauppi, P.E. 2008. The sustainability challenge of meeting carbon dioxide targets in Europe by 2020. Energy Policy, **36**(2): 730–742. doi:10.1016/j.enpol.2007.10.007.

Sall, J., Lehman, A., and Creighton, L. 2001. JMP Start statistics. A guide to statistics and data analysis using JMP and JMPIN software. 2nd ed. Duxbury, Pacific Grove, Calif.

Schwilk, D.W., Keeley, J.E., Knapp, E.E., McIver, J., Bailey, J.D., Fettig, C.J., Fiedler, C.E., Harrod, R.J., Moghaddas, J.J., Outcalt, K.W., Skinner, C.N., Stephens, S.L., Waldrop, T.A., Yaussy, D.A., and Youngblood, A. 2009. The National Fire and Fire Surrogate study: effects of fuel reduction methods on forest vegetation structure and fuels. Ecol. Appl. **19**(2): 285–304. doi:10.1890/07-1747.1. PMID:19323191.

Skog, K.E. 2008. Sequestration of carbon in harvested wood products for the United States. For. Prod. J. **58**: 56–72.

Stephens, S.L. 2000. Mixed conifer and upper montane forest structure and uses in 1899 from the central and northern Sierra Nevada, Calif. Madrono, **47**: 43–52.

Stephens, S.L. 2001. Fire history of adjacent Jeffrey pine and upper montane forests in the eastern Sierra Nevada. Int. J. Wildland Fire, **10**(2): 161–176. doi:10.1071/WF01008.

Stephens, S.L., and Collins, B.M. 2004. Fire regimes of mixed conifer forests in the north-central Sierra Nevada at multiple spatial scales. Northwest Sci. **78**: 12–23.

Stephens, S.L., and Moghaddas, J.J. 2005*a*. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. For. Ecol. Manage. **215**(1–3): 21–36. doi:10.1016/j.foreco.2005.03.070.

Stephens, S.L., and Moghaddas, J.J. 2005*b*. Fuel treatment effects on snags and coarse woody debris in a Sierra Nevada mixed conifer forest. For. Ecol. Manage. **214**(1–3): 53–64. doi:10.1016/j.foreco.2005.03.055.

Stephens, S.L., and Moghaddas, J.J. 2005*c*. Silvicultural and reserve impacts on potential fire behavior and forest conservation: twenty-five years of experience from Sierra Nevada mixed conifer forests. Biol. Conserv. **125**(3): 369–379. doi:10.1016/j.biocon.2005.04.007.

Stephens, S.L., Martin, R.E., and Clinton, N.E. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands and grasslands. For. Ecol. Manage. **251**(3): 205–216. doi:10.1016/j.foreco.2007.06.005.

Stephens, S.L., Moghaddas, J.J., Edminster, C., Fiedler, C.E., Haase, S., Harrington, M., Keeley, J.E., Knapp, E.E., McIver, J.D., Metlen, K., Skinner, C.N., and Youngblood, A. 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecol. Appl. **19**(2): 305–320. doi:10.1890/07-1755.1. PMID:19323192.

US Envrionmental Protection Agency (EPA). 2005. Emission facts: average carbon dioxide emissions resulting from gasoline and diesel fuel. U.S. EPA, Office of Transportation and Air Quality, Washington, D.C. Publ. EPA420-F-05-001.

van Mantgem, P.J., Stephenson, N.L., Byrne, J.C., Daniels, L.D., Franklin, J.F., Fulé, P.Z., Harmon, M.E., Larson, A.J., Smith, J.M., Taylor, A.H., and Veblen, T.T. 2009. Widespread increase of tree mortality rates in the western United States. Science (Washington, D.C.), **323**(5913): 521–524. doi:10.1126/science.1165000. PMID:19164752.

van Wagtendonk, J.W., Benedict, J.M., and Sydoriak, W.M. 1996. Physical properties of woody fuel particles of Sierra Nevada conifers. Int. J. Wildland Fire, **6**(3): 117–123. doi:10.1071/WF9960117.

van Wagtendonk, J.W., Benedict, J.M., and Sydoriak, W.M. 1998. Fuel bed characteristics of Sierra Nevada conifers. West. J. Appl. For. **13**: 1145–1157.

Ward, D.E., and Hardy, C.C. 1991. Smoke emissions from wildland fires. Environ. Int. **17**(2–3): 117–134. doi:10.1016/0160-4120(91)90095-8.

Westerling, A.L., Hidalgo, H.G., Cayan, D.R., and Swetnam, T.W. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science (Washington, D.C.), **313**(5789): 940–943. doi:10.1126/science.1128834. PMID:16825536.

Winistorfer, P., Chen, Z., Lippke, B., and Stevens, N. 2005. Energy consumption and greenhouse gas emissions related to the use, maintenance and disposal of a residential structure. Wood Fiber Sci. **37**: 128–139.

Winjum, J.K., Brown, S., and Schlamadinger, B. 1998. Forest harvest and wood products: sources and sinks of atmospheric carbon dioxide. For. Sci. **44**: 272–284.

Woodbury, P.B., Smith, J.E., and Heath, L.S. 2007. Carbon sequestration in the U.S. forest sector from 1990 to 2010. For. Ecol. Manage. **241**(1–3): 14–27. doi:10.1016/j.foreco.2006.12.008.