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Prescribed fire effects on field-derived and simulated forest carbon stocks over time

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A B S T R A C T

To better understand the impact of prescribed fire on carbon stocks, we quantified aboveground and belowground carbon stocks within five pools (live trees and coarse roots, dead trees and coarse roots, live understory vegetation, down woody debris, and litter and duff) and potential carbon emissions from a simulated wildfire before and up to 8 years after prescribed fire treatments. Total biomass carbon (sum of all the pools) was significantly lower 1 year post-treatment than pre-treatment and returned to 97% of pre-treatment levels by 8 year post-treatment primarily from increases in the tree carbon pool. Prescribed fire reduced predicted wildfire emissions by 45% the first year after treatment and remained reduced through 8 year post-treatment (34%). Net carbon (total biomass minus simulated wildfire emissions) resulted in a source (10.4–15.4 Mg ha^{-1}) when field-derived values were compared to simulated controls for all post-treatment time periods. However, the incidence of potential crown fire in the untreated simulations was at least double for the 2 year and 8 year post-treatment time periods than in the treated plots. We also compared field-derived estimates to simulated values using the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS). In our validation of FFE-FVS to predict carbon stocks, the model performed well for the total biomass carbon (4% difference); however, there was great variability within the individual carbon pools. Live tree carbon had the highest correlation between field-derived and simulated values, and dead tree carbon the lowest correlation and highest percent differences followed by herb and shrub carbon. The lack of trends and variability between the field-derived and simulated carbon pools other than total biomass indicate caution should be used when reporting carbon in the individual pools.

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1. Introduction

Forest ecosystems play an important role in the global carbon cycle; they are both sources and sinks of carbon (Depro et al., 2008; McKinley et al., 2011; Pan et al., 2011). Forests store about 45% of terrestrial carbon with about 60% in trees (Malmsheimer et al., 2011; Ryan et al., 2010). Within forests, the aboveground carbon pools are more dynamic than soil pools and are more affected by near-term management activities and disturbance (Hines et al., 2010). Forest management, land use change, and disturbances such as wildfire, storms, and insects all affect carbon pools. US forests are currently a carbon sink primarily because of afforestation and fire suppression since settlement (Birdsey et al., 2006; Houghton et al., 2000); however, the current sink may decline even under current suppression tactics through the 21st century as woody encroachment reaches its maximum extent and ecosystem recovery slows (Hurtt et al., 2002). Ironically, wildfire is one of the primary threats to carbon storage in dry forests of the Western US due in part to the elevated biomass or fuel levels that create the sink (Malmsheimer et al., 2011). Fire initially releases large amounts of carbon into the atmosphere as a result of the combustion of living vegetation and dead fuels. Additional carbon is released from the decomposition of fire-killed vegetation where carbon was initially stored, which is released over time as it decomposes (Harmon and Marks, 2002; Ryan et al., 2010). Typically, the impact of fire is a short-term phenomenon offset by the uptake of carbon by surviving and new vegetation following the fire (Canadell et al., 2007; Kashian et al., 2006). The recovery time is dependent on the intensity and frequency of fires, and the ability of the system to regenerate post disturbance due to factors such as site quality, soil loss, and seed source (Kashian et al., 2006). High intensity stand-replacing fire in forests adapted to low-severity fire is one of the largest risks to carbon storage because forests may not regenerate afterward resulting in a vegetation-type conversion

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(Ryan et al., 2010). A net loss will occur if the frequency of disturbance is shorter than the recovery period (Campbell et al., 2012; Kashian et al., 2006; Smithwick et al., 2002).

Fuel treatments have been shown to reduce the severity of wildfires (i.e., Lyons-Tinsley and Peterson, 2012; Moghaddas and Craggs, 2007; Pollet and Omi, 2002; Safford et al., 2012) and therefore reduce losses of carbon (North and Hurteau, 2011). However, there is a debate on the role of fuel treatments in the carbon balance of forests. One side of this debate hinges on the likelihood of a wildfire encountering a fuel treatment. Fuel treatments may be applied to areas that do not subsequently experience wildfire resulting in carbon reductions from the treatment without the carbon benefit from reduced wildfire emissions. In the western US, Rhodes and Baker (2008) found an 8% chance that fuel treatments were subsequently burned by wildfire in a 20 year period. Similarly, Campbell et al. (2012) found that ten locations must be treated in order to beneficially impact future fire in just one location. On the other hand, carbon emissions from the fuel treatment plus the reduced emissions from subsequent wildfire may be less than the greater emissions from a more intense wildfire in untreated fuels.

There are three approaches available to explore the impacts of fuel treatments on carbon stocks if a wildfire occurs. The first simulates stand data, treatments, and effects (i.e., Harmon and Marks, 2002; Mitchell et al., 2009). The second uses empirical stand data coupled with simulated fuel treatments and effects (i.e., Hurteau and North, 2009; Reinhardt and Holsinger, 2010). The third uses purely empirical data collected before and after fuel treatments were conducted. To date, the majority of publications that quantify fuel treatment effects on forest carbon stocks use empirical data with pre-treatment and immediate or near immediate post-treatment data (i.e., Finkral and Evans, 2008; North et al., 2009; Sorensen et al., 2011; Stephens et al., 2009, 2012). Currently only three studies go beyond the scope of immediate effects of fuel treatments on carbon stocks with empirical data (Boerner et al., 2008; Hurteau and North, 2010; Hurteau et al., 2011). Simulation modeling permits assessment of the long-term impacts (>20 years) of treatments on carbon stocks. However, more empirically based research is needed to understand the effects of fuel treatments on carbon pools, and to assess the accuracy of simulated outputs over the same time span.

In this study we calculated carbon stocks in various aboveground and belowground pools based on field data before and up to 8 years after treatment by prescribed fire in central and northern California. The goals of this study were to better understand how prescribed fire treatments affect forest carbon stocks over time and to assess the accuracy of modeling carbon stocks into the future using the Fire and Fuels Extension (FFE-FVS, Rebain, 2010; Reinhardt and Crookston, 2003) to the Forest Vegetation Simulator (FVS, Crookston and Dixon, 2005). The specific questions addressed are: (1) How do forest carbon stocks change over time? (2) How do potential carbon emissions vary from simulated wildfire over time? and (3) Do forest carbon stocks differ between field-derived and simulated values? This study is unique from existing research (Boerner et al., 2008; Hurteau and North, 2010; Hurteau et al., 2011) because of the regional scope, and it will be a first to assess the accuracy of simulated versus field-derived forest carbon stocks between various carbon pools.

2. Methods and materials

2.1. Study area

California is divided into three broad eco-region divisions based on precipitation amount and patterns as well as temperature (Bailey et al., 1994; Bailey, 1996). All of our plots fall within the Mediterranean division, which is characterized by temperate rainy winters and hot dry summers. Further classification into eco-region domains, provinces, and sections are based on vegetation, natural land covers, and terrain features (Bailey, 1996; Bailey et al., 1994; Miles and Goudey, 1997). Sugihara and Barbour (2006) created nine bio-regions in California by combining the 19 eco-region sections within California (Miles and Goudey, 1997) based on vegetation and fire regime. Our plots were within five of the nine bio-regions (Fig. 1): Sierra Nevada ($n = 9$), North Coast ($n = 4$), Southern Cascade ($n = 7$), Klamath Mountains ($n = 2$), and Northeastern Plateau ($n = 3$). Conifer species present in the plots included: white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), incense cedar (*Calocedrus decurrens* (Torr.) Florin), western juniper (*Juniperus occidentalis* Hook.), Jeffrey pine (*Pinus jeffreyi* Balf.), sugar pine (*Pinus lambertiana* Douglas), ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Hardwood species present in the overstory included: big leaf maple (*Acer macrophyllum* Pursh), tanoak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehder), canyon live oak (*Quercus chrysolepis* Liebm.), and California black oak (*Quercus kelloggii* Newberry). The elevation of the plots ranged from about 700–1650 m on all aspects. Slopes ranged from level ground to 48%.

2.2. Field sampling

The data used in this study were from a larger regional monitoring program to characterize pre- and post-treatment fuels and vegetation as a result of fuel treatments on national forests in California (Vaillant et al., 2009a; Vaillant et al. 2009b). Personnel on each national forest were contacted and asked to provide candidate fuel treatment projects that they expected to treat in the near future. This study includes only prescribed fire treatments that

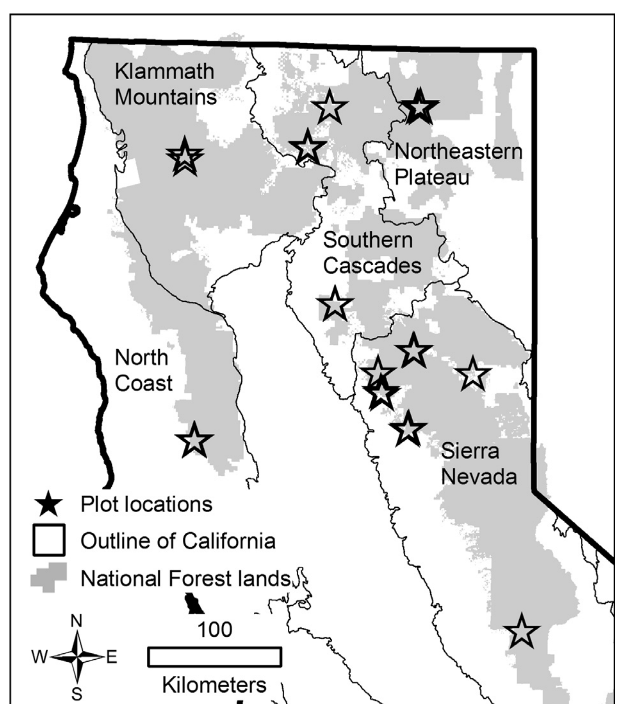

Fig. 1. Study plot locations, national forests, and ecoregions (Sugihara and Barbour, 2006) within California. All plots were established prior to treatment, then revisited 1 year, 2 year, and 8 year after the prescribed fire.

were implemented by the individual national forests treated for the project area, not just the plot locations.

Once the fuel treatment projects were selected, plots were randomly located within each fuel treatment before the treatment occurred and pre-treatment data were collected for a total of 25 plots in 12 fuel treatment projects (Fig. 1). The field sampling protocol was based on the National Park Service monitoring handbook (USDI National Park Service, 2003) with some modifications to optimize sampling efficiency (Vaillant et al., 2009a). Up to six plots were installed in each project prior to the prescribed fire; however, only those affected by the burns were maintained (1–4 plots per project). All plots were sampled prior to treatment, then 1 year, 2 year, and 8 year post-treatment.

Overstory, pole-size, and seedling tree information were gathered within fixed area nested plots sized 0.1 ha, 0.025 ha, and 0.005 ha, respectively. Oversto ry trees included those equal to or greater than 15 cm diameter at breast height (dbh), pole-sized trees were greater than or equal to 2.5 cm and less than 15 cm dbh, and seedlings were less than 2.5 cm dbh. All live and dead overstory and pole-sized trees were tagged. For all live overstory and pole-sized trees, tag number, species, dbh, height to live crown base, and total height were recorded. For all dead overstory and pole-sized trees, tag number, species, dbh, and total height were measured. Seedlings were tallied by species, vigor (live or dead), and height class (⩽15 cm, >15 cm to ⩽30 cm, >30 cm to ⩽60 cm, >60 cm to ⩽100 cm, >100 cm to ⩽200 cm, etc.).

Understory vegetation was collected along 50 m transect(s). Shrub data included: species, average height, length along the transect, and vigor. Species and cover classes by vigor for non-vascular plants, subshrubs, forbs, and grasses were recorded within five 1 m by 1 m quadrats placed every 10 m along each transect. Cover classes were 0–1%, >1–5%, >5–25%, >25–50%, >50–75%, and >75–100% (Daubenmire, 1959).

Forest floor (litter and duff) and surface fuels (dead and down woody material) were inventoried following the line intercept method (Brown, 1974; Van Wagner, 1968) with 15.24 m transects. Dead and down 1-h (⩽0.64 cm in diameter) and 10-h (0.64 cm to ⩽2.54 cm in diameter) fuels were tallied for the first 1.83 m, and 100-h (2.54 cm to ⩽7.62 cm in diameter) fuels were tallied for the first 3.66 m. Diameter and soundness (rotten or sound) were recorded for all dead and down 1000-h fuels (>7.62 cm in diameter) for the entire transect. Litter and duff depths were recorded at 10 equidistant points along each transect starting at 0.3 m. Fuel bed depth, defined as the maximum height from the bottom of the litter layer to the highest dead and down fuel along a vertical plane extending from the fuel transect (Brown, 1974), was recorded in 10 equidistant intervals.

2.3. Calculating carbon stocks from field data

The Fire and Fuels Extension (FFE-FVS) for the Forest Vegetation Simulator (FVS) calculates and reports carbon stocks for eight pools based on forest stand data. The carbon pools include: aboveground live tree carbon (live tree stems, branches, and foliage), standing dead carbon (dead tree stems, branches, and foliage if present), belowground live (coarse roots of live trees), belowground dead carbon (coarse roots of dead or cut trees), dead down wood (all woody surface material), forest floor (litter and duff), and herbs and shrubs (live plants only), and carbon released from fire (carbon in fuel consumed by simulated fires) (Hoover and Rebain, 2011; Rebain, 2010). The FFE-FVS leverages tree growth from FVS and models non-tree fuel loads (i.e., accumulation and decomposition of dead woody material) over time and potential fire behavior. It also calculates carbon stocks within the stand and potential carbon emissions released by a simulated fire. The FFE-FVS is one of many extensions available for the FVS and is frequently used for vegetation and fire and fuels management. The FVS and FFE-FVS use geographically derived equations called "variants" to model tree growth and fuel accumulation and decomposition over time. Our plots are within four variants: Western Sierras ($n = 9$), Southern Oregon/Northeast California ($n = 8$), Klamath Mountains ($n = 4$), and Inland California/Southern Cascades ($n = 4$).

In order to run FFE-FVS, data characterizing the stand and trees within each stand are input (Dixon, 2002; Rebain, 2010; Reinhardt and Crookston, 2003). Location (region, forest, district, latitude, longitude) and topographic information (elevation, slope) were provided for each stand. Biomass estimates of surface fuel and forest floor loads were calculated from field data with coefficients specific to the Sierra Nevada range (van Wagtendonk et al., 1996, 1998) were included in the stand information for FFE-FVS. Tree data input included: species, dbh, height, crown ratio (calculated from tree height and height to live crown base measurements), and history code (live, recently dead, or dead for a long time). For seedling trees taller than 1.52 m (6 ft) we assigned crown ratio of 75% and dbh of 2.3 cm (0.9 in.) because these values were not gathered in the field. Each plot was run separately within FFE-FVS.

Within FFE-FVS there are two options for calculating aboveground live tree biomass: the variant-specific default equations within FFE-FVS (Hoover and Rebain, 2011; Rebain, 2010) or equations derived by Jenkins et al. (2003). We used the Jenkins et al. (2003) equations, because our plots came from multiple FVS variants (Hoover and Rebain, 2011). All other carbon pools assume and use the FFE-FVS default calculations to convert biomass to carbon stocks. Within FFE-FVS, biomass is assumed to be half carbon (Penman et al., 2003) therefore all values are divided in half to determine carbon stocks with the exception of forest floor (litter and duff), which is assumed to be 37% carbon (Smith and Heath, 2002).

FFE-FVS was used to calculate carbon stocks for the aboveground live tree, standing dead tree, belowground live, and belowground dead carbon pools. Since the belowground carbon pools are determined based on the above ground tree data, aboveground live tree and belowground live carbon were combined into a single "tree" pool, and aboveground dead tree and belowground dead carbon were also combined into a single "snag" pool for the same reasoning. Because FFE-FVS outputs carbon stocks at the end of a growing season, the SNAGBRK and SNAGFALL keywords (commands that control processes, inputs and outputs for FVS) were used to stop breakage and falling of material, respectively, to better match the data for field-derived snag carbon estimates. It is possible to alter fuel accumulation and decomposition rates but not completely stop these processes, because we were interested in examining the carbon stocks as collected in the field, not after a simulated growing season, values for dead down wood (hereafter "surface fuel") and forest floor (litter and duff) were calculated outside of the program by multiplying the calculated loads by the appropriate conversions (0.5 for surface fuels and 0.37 for forest floor). In addition, herb and shrub data is not an available input for the model. Rather the biomass is determined based on percent canopy closure and dominant tree species for most of the variants used in this analysis (Rebain, 2010). Live herb and shrub biomass was directly calculated from our field data using the FIREMON methodology and bulk density values (Lutes et al., 2006) then divided in half to determine carbon stocks.

This dataset was not from an experiment which included a control, but rather, was a compilation of region-wide treatment monitoring data. In lieu of actual untreated control data we simulated our pre-treatment data forward to the 1 year, 2 year, and 8 year post-treatment time periods using FFE-FVS to determine total biomass carbon.

Table 1
Daily weather and fuel moistures were obtained from the most representative remote automated weather station (RAWS) for each fuel treatment site for the day of each prescribed fire and used in the simulations. Summary statistics for each parameter used are presented below.

Parameter	Mean	Median	Minimum	Maximum
1-h (%)	6	6	3	13
10-h (%)	7	8	4	13
100-h (%)	10	10	6	15
1000-h (%)	18	17	10	29
Live herbaceous (%)	47	30	30	132
Live woody (%)	84	70	60	134
Temperature (°C)	15	16	6	26
Wind speed (km h^{-1})	9	8	2	15

2.4. Estimating potential carbon emissions from simulated wildfire

The FFE-FVS program was used to estimate potential carbon emissions (defined as carbon in fuel consumed from wildfire, Rebain, 2010) from a simulated wildfire at each time period for the field-derived data and simulated untreated controls. FFE-FVS was used to select the most representative fire behavior fuel models (Anderson, 1982; Scott and Burgan, 2005) for the simulation. Fuel moisture values for the "very dry" scenario with FFE-FVS were used for the simulation. Specifically, 1-h, 10-h, 100-h, 1000-h, duff, and live fuel moistures were 3%, 4%, 5%, 10%, 15%, and 70%, respectively. FFE-FVS also requires a 6 m (20 ft) wind speed and temperature for simulating wildfires; the default 32 km h^{-1} wind speed was used with a 27 °C temperature.

2.5. Simulating carbon stocks into the future with field-derived values

The FFE-FVS program was used to project carbon stocks into the future using the pre-treatment data. A prescribed fire treatment was applied to each plot, and plots were simulated forward through 8 year post-treatment to match the field data. Daily weather and fuel moisture data were obtained from the most representative remote automated weather station (RAWS) for each plot on the day of each prescribed fire and were processed using Fire Family Plus (Main et al., 1990). A summary of this data is available in Table 1. If live herbaceous or live woody fuel moisture were below fully cured values (30% and 60%, respectively), fully cured values were used (Scott and Burgan, 2005). The "dry" default (50%) for duff moisture in FFE-FVS was used for each plot. Again the Jenkins et al. (2003) equations were used for calculating aboveground live tree biomass, and the FFE-FVS defaults for all other pools. No further restrictions on fuel accumulation and decomposition or snag changes were used, allowing the program to control these factors. Simulated values for 1 year, 2 year, and 8 year post-treatment were then compared to field-derived estimates.

2.6. Statistical analysis

For the field-derived data, we used a mixed model with repeated measures in SAS (SAS Institutes Inc., Cary, North Carolina, USA) to analyze changes in carbon pools over time. The fuel treatment site was included as the random factor in the model, because plots were not truly independent. Before statistics were run a significance level of $P < 0.1$ was chosen because of the known spatial variability with fuels (Keane et al., 2012). To compare the post-treatment field-derived and simulated carbon pools, Spearman rank-correlation coefficients were calculated for each metric and time period to assess the relatedness between the data sets.

3. Results

3.1. Prescribed fire impacts on field-derived carbon stocks over time

Our field-derived estimates of carbon stocks indicate that trees contributed the largest proportion of the total biomass (sum of tree, snag, herb and shrub, surface fuels, and forest floor pools) carbon (range 74–83%) and herbs and shrubs the least (<1%) for all time periods (Table 2). The prescribed fire treatments reduced total biomass carbon by about 13%. The largest carbon stock reduction was in the forest floor pool (55%) and the smallest the tree pool (2%). With the exception of the tree and herb and shrub carbon pools, all others increased 2 year relative to 1 year post-treatment. By 8 year post-treatment most carbon pools (snag did not) exceeded both the 1 year and 2 year post-treatment stocks and total biomass carbon returned to 97% of the pre-treatment level.

Total biomass carbon was significantly ($P < 0.1$) reduced by treatment (1 year post), however in subsequent years (2 year and 8 year post), although still reduced, the change was no longer significant (Table 2). Snag carbon significantly increased between the pre-treatment and 2 year and 8 year post-treatment as well as the 1 year and 2 year post-treatment time periods. The 8 year post-treatment herb and shrub carbon stocks were higher than all other time periods; the difference was significant between 8 year and both 1 year and 2 year post-treatment. Surface fuel carbon was only significantly reduced between pre-treatment and 1 year post-treatment (40% reduction); however, later years were still reduced by about 27%. Forest floor carbon was significantly different between all time periods except 1 year and 2 year post-treatment, with post-treatment 34–55% lower than pre-treatment. The tree carbon pool remained relatively constant through all time periods, with the largest difference (9 Mg ha^{-1}) between 2 and 8 year post-treatment (Table 2).

3.2. Potential carbon emissions from modeled wildfire

Potential carbon emissions from modeled wildfire were reduced as a result of the prescribed fire (45% reduction), and re-

Table 2
Mean and standard error (SE) values of field-derived carbon stocks for various pools and simulated wildfire emissions. Significant differences ($P < 0.1$) among time periods for a single metric are denoted by the same letter. Statistical analysis was not completed for the net carbon.

Carbon pool	Pre Mg ha^{-1}	1 Year post Mg ha^{-1}	2 Year post Mg ha^{-1}	8 Year post Mg ha^{-1}
Tree (live coarse roots, stems, branches, and foliage)	150.2(18.8)a	147.1(18.7)a	145.7(18.8)	154.7(20.5)
Snag (dead coarse roots, stems, branches, and foliage if present)	3.3(1.3)ab	3.7(1.0)c	6.6(1.6)ac	6.1(1.7)b
Herb and shrub	0.7(0.1)ab	0.4(0.1)ac	0.3(0.1)bd	0.8(0.1)cd
Surface fuel (downed woody debris)	26.0 (5.2)a	15.7(3.8)a	18.5(4.5)	19.0(4.3)
Forest floor (litter and duff)	24.8(2.2)abc	11.1(1.3)ad	12.0(1.2)be	16.4(1.1)cde
Total biomass (sum of above)	204.0(21.1)a	177.9(21.2)a	181.8(20.7)	197.1(22.2)
Simulated wildfire emissions	34.7(3.6)abc	19.0(2.4)a	20.5(2.4)b	23.0(2.0)c
Net (total biomass – wildfire)	169.3(19.2)	158.9(20.1)	161.3(18.1)	174.1(21.3)

Table 3
Mean and standard error (SE) values of total biomass carbon, simulated wildfire emissions, and net carbon for simulated untreated controls for the same time periods as the post-treatment field-derived data.

Carbon pool	1 Year post Mg ha^{-1}	2 Year post Mg ha^{-1}	8 Year post Mg ha^{-1}
Simulated untreated total biomass	203.3(20.8)	206.2(20.8)	228.7(21.6)
Simulated wildfire emissions	34.0(3.4)	34.7(3.5)	39.2(3.8)
Net (total biomass – wildfire)	169.3(19.8)	171.6(19.6)	189.5(19.4)

mained significantly lower than the pre-treatment levels through 8 year post-treatment (41% and 34% reduction for 2 year and 8 year post-treatment, respectively) for the field-derived data (Table 2). Potential carbon emissions from modeled wildfire were predominantly from the consumption of carbon in the forest floor (average 48%, range for time periods 44–52%) and surface fuel (average 39%, range for time periods 36–44%), followed by tree (average 9%, range for time periods 8–10%), then herb and shrub (average 4%, range for time periods 2–6%) pools.

Potential carbon emissions from modeled wildfire in the simulated untreated controls (pre-treatment field-derived data simulated forward without treatment) were highest for the 8 year post-treatment time period and lowest for the 1 year post-treatment time period (Table 3). Mean potential carbon emissions from the pre-treatment field-derived data were higher than the 1 year post-treatment control and the same as the 2 year post-treatment control (Tables 2 and 3).

3.3. Comparing field-derived and FFE-FVS simulated carbon stocks

Compared to field-derived values, FFE-FVS over predicted mean carbon stocks for snag for all the time periods, herb and shrub for 1 and 2 year post, and surface fuel for 8 year post and under predicted total, tree, and forest floor for all time periods, herb and shrub for 8 year post, and surface fuel for 1 and 2 year post (Table 4). Field-derived versus simulated total biomass carbon mean difference was 4% (3–6% range) when all years were averaged. Spearman rank order values indicate a high correlation between the field-derived and simulated values for the total biomass and tree carbon pools (ρ 0.91–0.96); the correlation increases slightly in later years for total biomass (Table 4). The FFE-FVS simulated snag carbon has reasonable correlation to the field-derived data 1 year post-treatment (ρ 0.53), but poor correlation 2 and 8 years post-treatment (ρ 0.21 and 0.28). The herb and shrub data show a great deal of variation (−224 to 2% difference) and little correlation (ρ −0.02–0.16) between field-derived and simulated values. The scatter seen in the forest floor and surface fuel carbon pools highlight the variability found in the field (Fig. 2). Although variable, FFE-FVS produces reasonable estimates of forest floor carbon (ρ 0.65–0.69) and surface fuel carbon 1 year post (ρ 0.73) and less so 2 and 8 year post (ρ 0.43 and 0.51).

4. Discussion

4.1. Prescribed fire impacts on carbon stocks over time

In the western US, wildfire is one of the largest threats to dry forest carbon (Malmsheimer et al., 2011). Fuel treatments have been proven to reduce fire intensity and therefore severity when burned by wildfires under most weather conditions (i.e., Lyons-Tinsley and Peterson, 2012; Moghaddas and Craggs, 2007; Pollet and Omi, 2002; Safford et al., 2012). Although only a small percentage of fuel treatments are burned by wildfires (Campbell et al., 2012; Rhodes and Baker, 2008), treatments continue to be implemented due to the known benefits to aiding fire suppression resources (Moghaddas and Craggs, 2007) and reducing potential fire emissions (North and Hurteau, 2011). To date, three studies have quantified the effect of fuel treatments on carbon stocks beyond the first couple of years using empirical data (Boerner et al., 2008; Hurteau and North, 2010; Hurteau et al., 2011). Data from our study is consistent with that of Boerner et al. (2008), Hurteau and North (2010) and Hurteau et al. (2011); low intensity fuel treatments, such as prescribed fire, offset carbon lost through treatment over a relatively short timeframe. We found total biomass carbon to be 97% of the pre-treatment level by 8 year post-treatment.

While limited data is available in Boerner et al. (2008), Hurteau and North (2010), and Hurteau et al. (2011), general comparisons between their research and ours suggests fairly good agreement. Hurteau and North (2010) reported immediate and 7-year post-treatment changes to carbon within trees, snags, and shrubs in a California mixed conifer forest making comparisons between their findings and ours possible. Between the immediate and 7-year post prescribed fire treatment, Hurteau and North (2010) found only a 1% gain in shrub carbon, whereas our understory live carbon more than doubled between 1 year and 8 year post-treatment. Part of our gains could be attributed to the fact that we included other understory vegetation in addition to shrubs and that our values are much higher after treatment. Combining small and large tree and snag carbon from Hurteau and North (2010), tree carbon increased at a rate of 4.62 Mg ha^{-1} year^{-1} and carbon in the snag pool increased by 4.49 Mg ha^{-1} year^{-1}. Our accumulation rates were much lower between the 1 year and 8 year post data; tree carbon increased by only 1.09 Mg ha^{-1} year^{-1} and carbon in the snag pool increased by 0.34 Mg ha^{-1} year^{-1}. The difference in tree carbon accumulation is possibly due to higher site productivity in the plots sampled by Hurteau and North (2010) than our plots across northern California. Lower accumulation rates in snag carbon might be because our plots had a higher percentage of snags falling (69%) between 1 year and 8 year post-treatment resulting in them contributing to the surface fuel instead of the snag pool.

In a meta-analysis of 12 study sites across the US, Boerner et al. (2008) found more intense treatments (mechanical plus fire) to have a larger negative impact on total ecosystem carbon than less intense treatments (fire-only or mechanical only). Although total

Table 4
Percent difference and Spearman rank order (ρ) values between field-derived and FFE-FVS simulated carbon pools 1 year, 2 year, and 8 year post-treatment.

	Percent difference			Spearman rank order (ρ)		
Carbon pool	1 Year post (%)	2 Year post (%)	8 Year post (%)	1 Year post	2 Year post	8 Year post
Tree	15	9	4	0.92	0.93	0.91
Snag	−534	−223	−136	0.53	0.21	0.28
Herb and shrub	−160	−224	2	0.16	0.11	−0.02
Surface fuel	28	29	−1	0.73	0.43	0.51
Forest floor	31	28	47	0.67	0.65	0.69
Total biomass	6	3	3	0.92	0.96	0.96

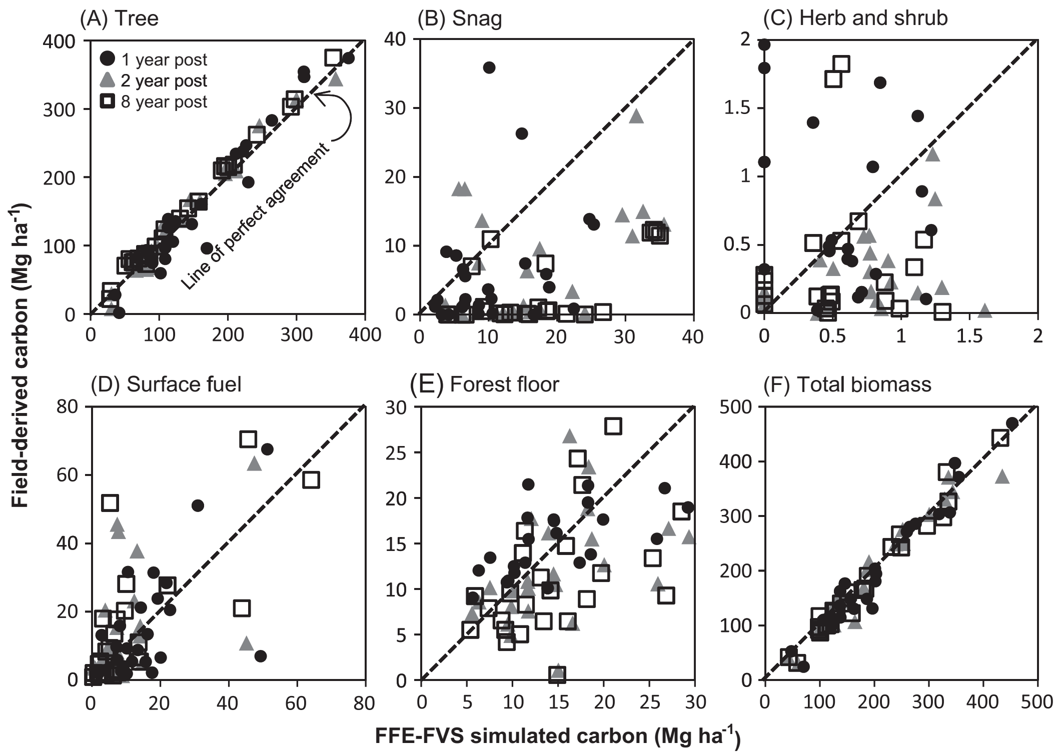

Fig. 2. Field-derived carbon stocks versus FFE-FVS simulated carbon stocks for 1 year, 2 year, and 8 year post-treatment for tree (A), snag (B), herb and shrub (C), surface fuel (D), forest floor (E), and total biomass (F) carbon pools.

biomass carbon losses were similar for fire-only and mechanical only treatments, different pools were affected. In mechanical only treatments the majority of the losses were from removal of live vegetation including trees, whereas for fire the loss was primarily from forest floor carbon. Between 2 and 4 years following treatment, fire-only treatments recovered more carbon than mechanical and mechanical plus fire treatments primarily due to increases in forest floor and dead wood carbon most likely from the fire treatment itself coupled with lower removal of tree carbon (Boerner et al., 2008). We found similar patterns where prescribed fire reduced surface fuels and forest floor; however, even though these pools increased over time, snags remained intact and surviving trees rapidly assimilated enough carbon to offset losses from the prescribed fire.

4.2. Potential carbon emissions from modeled wildfire

To fully understand the effects of fuel treatments on carbon stocks, the effectiveness to limit losses from wildfire should also be considered (Finkral and Evans, 2008; Reinhardt and Holsinger, 2010). Recently, North and Hurteau (2011) found fuel treatments do reduce wildfire emissions by over a half; however, when the wildfire emissions and the fuel treatment emissions are combined they result in a net loss initially compared to wildfire alone. We had similar findings to North and Hurteau (2011). The year following the prescribed fire treatment, simulated wildfire emissions were reduced to about half of the pre-treatment emissions and remained lower through 8 year post-treatment than pre-treatment (Table 2).

In a simulation study, Reinhardt and Holsinger (2010), predicted it would take about 11–50 years, after a simulated wildfire, before stands in multiple forest types treated with fire only in Montana would become a sink. The best comparison to our work was a burn only treatment in warm dry ponderosa pine forest, where it was predicted to take about 11 years after treatment to become a carbon sink (Reinhardt and Holsinger, 2010). When net carbon (total biomass minus wildfire emissions) from the field-derived treated plots was compared to simulated untreated controls for the same time periods, the treated plots were consistently lower with the largest difference at the 8 year post-treatment time period (Tables 2 and 3). Although we found a net loss in carbon after treatment compared to the simulated untreated controls, the potential for crown fire was much higher in the untreated plots (52% and 60% for 2 year and 8 year post-treatment) than the treated plots (16% and 32% for 2 year and 8 year post-treatment). The carbon source in the treated plots is likely to become a benefit over the long-term because of retention of live trees, reduction in decomposition emissions, and resistance to conversion to a lower carbon vegetation type such as a grassland or shrubland from a lower likelihood of uncharacteristic high severity fire relative to the simulated untreated controls.

Additionally, it is likely that our methods overestimate carbon emissions from both wildfire and prescribed fire because we have not estimated the quantity of fuel converted to ash and residual partially burned fuel particles in ash (e.g., black carbon) (North et al., 2009; Smith et al., 2005). Although a small percentage of fuel lost during fire remains on site as black carbon, its role as a long-term carbon pool necessitates more study (DeLuca and Aplet, 2008).

4.3. Comparing field-derived and FFE-FVS simulated carbon stocks

Many studies that assess carbon stocks with respect to fuel treatments and wildfires use the FFE-FVS because of the integration of carbon stock calculations, ability to model fuel treatments, fire simulation capabilities, and the capacity to model the effects of wildfire overtime (i.e., Hurteau and North, 2009; Reinhardt and Holsinger, 2010; Sorensen et al., 2011). Because of the dependence on modeling systems to understand the effects of treatments and disturbance on carbon stocks, we compared our post-treatment field-derived carbon stocks to those simulated in FFE-FVS to assess the accuracy of the model. In our study, simulated and field-derived total biomass carbon stocks only differed by 4% for all time periods and were highly correlated. The largest percent differences were found when FFE-FVS vastly overestimated carbon stored in snags (more so in earlier time periods than later, Table 4). It is possible we did not model the exact conditions for the prescribed fire as were applied in the field resulting in higher tree mortality predicted in FFE-FVS. In addition, the default setting in FFE-FVS initially overestimated snag recruitment and then underestimated it in later years. Snag recruitment continued through 8 year post-treatment because of delayed mortality within our plots. The FFE-FVS simulated snag carbon stocks slightly decreased between 1 year and 2 year post-treatment (12%); by 8 year post-treatment the snag carbon pool was further reduced by about 33% indicating that the majority of trees were assumed to have died within the first year and then start to fall over in later years. How tree mortality (and the resultant snag carbon pools) was modeled in FFE-FVS was different than how tree mortality occurred and was recorded in our plots. Carbon within the snag pool about doubled between 1 and 2 year post-treatment, indicating a slight delay in mortality. Of 801 trees initially sampled, 173 trees had fallen by 8 year post-treatment; but137 had been alive prior to the prescribed burn. The herb and shrub carbon pool had the lowest correlation coefficients between the field-derived and simulated stocks and was also the lowest contributor to total biomass carbon. It was anticipated that the herb and shrub carbon stocks would vary between the two data sets because FFE-FVS does not incorporate understory vegetation as an input into the model (Hoover and Rebain 2011; Rebain, 2010). The FFE-FVS underestimated forest floor carbon stocks all time periods; surface fuel carbon was also initially underestimated, then slightly overestimated by 8 year post-treatment. The actual prescribed fires consumed a higher proportion of the forest floor carbon (55% vs. 41%) and a lower proportion of surface fuel carbon (40% vs. 47%) than what was modeled in the FFE-FVS. This partially explains the difference found between simulated and field-derived carbon in these pools. An average difference of total biomass carbon of about 4% is acceptable if one is only concerned with the sum of the pools. The lack of trends and variability between the field-derived and simulated carbon pools other than total biomass indicate caution should be used when reporting carbon in the individual pools, or perhaps alternative model estimates should be developed for the pools that are very different (snag and herb and shrub). In addition, our simulations only modeled changes to carbon pools eight years into the future; the accuracy beyond this period is unknown.

4.4. FVS/FFE-FVS limitations

As with all modeling systems, limitations and weaknesses exist with FVS and FFE-FVS. Only limitations related to tree growth, fuel accumulation and decomposition, and carbon reports are discussed. Although FVS includes variant specific equations for tree growth, calibration with local site information (topography and site productivity) is highly encouraged (Hoover and Rebain, 2011). In addition, tree regeneration is not automatically modeled in most variants and needs to be initiated by the user (Crookston and Dixon, 2005); none of the variants in this study included regeneration. FVS models non-tree vegetation for a very limited geographic area using the COVER extension to FVS (Moeur, 1985). For most variants (including those used in this study), it relies on tree canopy cover and habitat type to assign fuel loading constants for live herb and shrub fuels. Non-tree vegetation is not an input for the model (Crookston and Dixon, 2005). Fuel accumulation is only from live and dead trees including harvesting activities; contributions from non-tree vegetation are not considered (Rebain, 2010). Decay of fuel pools over time is based on a limited set of empirically derived relationships as described in Rebain (2010) and is simulated over time using a simple constant proportional loss model. The above factors can all affect the carbon reports from FFE-FVS for this study. Finally, FFE-FVS was not designed for full entity-wide carbon accounting; currently carbon within soil and fine roots is not included, nor are carbon sources associated with harvesting and transporting wood products, application of fertilizer, etc. (Rebain, 2010; Hoover and Rebain, 2011).

In addition, each variant in FVS is calibrated to a default cycle length (the number of years between projections). For the variants used in the study the default cycle length is 5 years for Klamath Mountains and 10 years for the others. Although it is possible to change the cycle length; for example, 1 year cycles were used in this study, it is discouraged because the default cycle length may result in under or over prediction of stand values (Hoover and Rebain, 2011; Wykoff et al., 1982). However, Hoover and Rebain (2011) note that a few 1 year cycles will not significantly affect results, but completing simulations with 1-year cycles is discouraged. To be able to answer the questions we set forth 1 year cycles were necessary and since the simulation period was short (a maximum of 8 years) we assumed the under and over prediction is minimal and does not impact our results.

Finally, FFE-FVS was used to simulate untreated controls for the 1 year, 2 year, and 8 year post-treatment time periods in lieu of actual data. Although this seems like a valid way to more directly compare differences in net carbon, our one concern with this method is that by introducing an untreated dataset modeled through time and comparing it to field-derived data, our 'untreated' dataset will be under the influence of modeling error, whereas our 'treated' dataset was derived from field data. This is evident when looking at the simulated wildfire emissions in the untreated dataset. The emissions are lower 1 year post-treatment and identical 2 year post-treatment to the pre-treatment field-derived values. However, because the total biomass carbon was highly correlated between modeled and field-derived dataset, we decided to include the simulated untreated controls.

5. Conclusions

Carbon stocks within trees encompass the majority of the total biomass carbon (80%) within our study. Similar to other research (Boerner et al., 2008; Hurteau and North, 2010; Hurteau et al., 2011), we reported rapid tree carbon recovery in low intensity prescribed fire treatments. Although the prescribed fire treatments reduced total biomass carbon by 13%, tree carbon was reduced by only 2% enabling stand carbon to recover to near pre-treatment levels by 8 year post-treatment. Prescribed fire treatments initially reduced potential losses from modeled wildfire emission by about half; however, when the treatment removals are included, a net source of carbon results for the first 2 years following treatment. Although treatment resulted in a net source of carbon when compared to simulated untreated controls, this initial loss is likely to become a benefit over the long-term. The incidence of potential crown fire was much lower in the treated stands further confirming the benefit of prescribed fire treatments to reduce carbon loss in the likelihood a wildfire will eventually occur.

When carbon stocks were simulated into the future using FFE-FVS estimates of total biomass, carbon stocks were on average only 4% different than the field-derived values. The overall deviation is acceptable given the time span addressed, but additional comparisons across longer time periods are needed for the individual carbon pools, which varied up to 534%. Additionally, one of the least correlated carbon pools (herb and shrub) was the only one that FFE-FVS does not include as an input, which is a known weakness of the program (Hoover and Rebain, 2011).

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