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Post-fire management regimes affect carbon sequestration and storage in a Sierra Nevada mixed conifer forest

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ABSTRACT

Forests mitigate climate change by sequestering CO_2 from the atmosphere and accumulating it in biomass storage pools. However, in dry conifer forests, fire occasionally returns large quantities of CO_2 to the atmosphere. Both the total amount of carbon stored and its susceptibility to loss may be altered by post-fire land management strategies. Forest managers face a great challenge when asked to manage these lands for C sequestration and simultaneously reduce fire hazard. The objective of our study was to understand how differing post-fire management strategies affect C sequestration and the size of storage pools in the 10 years after a wildfire in a Sierra Nevada mixed-conifer forest. Post-fire management regimes included: (1) salvage-logged, planted, and intensively managed plantation (IM); (2) salvage-logged and planted (SP); (3) no salvage (NS); and (4) green canopy (GC), where fire burned through, but 95% of the overstory trees survived. Carbon sequestration and storage were estimated from measurements of individual ecosystem carbon pools. These pools included: aboveground trees, saplings, snags, stumps, and understory, coarse wood and fine wood, duff, and soil. We found total ecosystem carbon storage was 282 ± 15 Mg ha^{-1} of C in the NS treatment, 206 ± 31 Mg ha^{-1} in the GC, 137 ± 13 Mg ha^{-1} in the SP, and 101 ± 15 Mg ha^{-1} in the IM treatment. There were no significant treatment differences in C storage among the pools that would constitute labile/fine fuels, but there were differences in recalcitrant (coarse fuel) C pools. The greatest C storage in recalcitrant C pools was 258 ± 10 Mg ha^{-1} in the NS and 197 ± 30 Mg ha^{-1} in the GC treatments. Post-fire carbon sequestration rates were 1.6 ± 0.7 Mg ha^{-1} year^{-1} of C in the GC, 0.7 ± 0.3 Mg ha^{-1} year^{-1} in the SP, 0.5 ± 0.1 Mg ha^{-1} year^{-1} in the NS, and 0.5 ± 0.1 Mg ha^{-1} year^{-1} in the IM treatments, but these differences were not statistically significant. Tree carbon sequestration rates were highest in the GC treatment and lowest in the NS treatment. Overall, our results suggest that a mature green-canopy stand provides most benefit in terms of C sequestration, wildfire resilience, and other ecosystem services at a point 10 years after severe wildfire. For forests that suffer high fire mortality, unsalvaged (NS) stands will retain the most carbon onsite.

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1. Introduction

Forests play an important role in the global C cycle because they sequester large quantities of carbon into storage as biomass. Here we will refer to "sequestration" as the rate of annual transfer between atmosphere and biosphere; "storage" will refer to the size of the carbon pool at a point in time. Summed over the land area of the US, forests sequester 148–204 Tg year^{-1} of C (Birdsey et al., 2006), which would offset 10–19% of the annual fossil fuel emissions in the US (Birdsey et al., 2006; Woodbury et al., 2007; Ryan et al., 2010) However, both storage amount and sequestration vary with climate, ecosystem type, disturbance regime, and land use and management practices (Bonan, 2008). Many of these characteristics are changing. For example, the US sequestration rate has increased by 33% from the 1990s to 2000s as a result of increased forest area and recovery from past disturbance, management, and environmental change (Pan et al., 2011).

Several forest management strategies have been proposed to mitigate C emissions. Silvicultural practices seek to increase the density of C in existing forests by promoting carbon sequestration (Canadell and Raupach, 2008). These strategies include afforestation, reforestation, and other silviculture practices (Fahey et al., 2009; Stephens et al., 2009a,b). Afforestation and reforestation increase the amount of land area covered by forest and the amount of carbon stored in biomass.

Climate mitigation efforts may also be achieved by reducing carbon losses from decomposition, which tends to increase carbon

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storage. Carbon storage can be increased through management that favors specific carbon pools that decompose less rapidly (Gaudinski et al., 2000; Kirschbaum, 2000). We will refer to these pools as "recalcitrant." For example, recalcitrant pools include large-diameter trees, coarse wood, and organic matter in mineral soil. In comparison, "labile" C, which decomposes more rapidly, is found in foliage, litter, and understory. Promoting carbon storage in recalcitrant C pools may help managers achieve mitigation goals.

A third way to increase carbon storage is to reduce wildfire occurrence and severity. Managers may achieve these reductions by implementing fuel reduction treatments. Such treatments may seek to reduce surface fuels, increase the height to live crown, decrease crown density, and retain large trees of fire-resistant species (Agee and Skinner, 2005). Fuel reduction may be achieved through prescribed fire, understory removal, tree thinning, whole-tree harvest, or a combination of these approaches (Fule et al., 2001; Peterson et al., 2005; Stephens et al., 2009a,b). These treatments shift carbon storage from pools that favor ignition and increase severity to pools that resist ignition and reduce severity. These pools largely overlap with the ones we have previously termed labile and recalcitrant, because pools that decompose slowly are also less likely to combust. In this study, we will include the following pools as labile/fine fuels: duff, fine wood, and understory. Recalcitrant pools will include coarse wood, stumps, snags, trees, and the portion of soil below the duff layer.

Management of fire-prone forests thus presents a unique challenge. Managers must trade off the promotion of carbon storage against the reduction of fire risk. Managing for carbon storage may increase forest or understory biomass, which may increase fuels (Dore et al., 2008), perhaps propagating crown fires (Agee and Skinner, 2005). On the other hand managing to reduce fire risk often requires a reduction in C storage as fuel loads are reduced (Mitchell et al., 2009). For example, several of our sites have intense post-fire shrub growth (*Ceanothus cordulatus*). This research was initiated in part because, for these shrubs, we did not know the tradeoff between carbon sequestration and their contribution to the labile/fine fuel pool. More generally, we sought to quantify all the major carbon pools and sequestration rates in areas that had been managed differently after fire. The specific objectives of this study were to (1) determine the effects of post-fire management on forest C sequestration, (2) determine how the distribution of C pools varies across management treatments, and (3) compare among treatments the tradeoff between the benefits of C sequestration and the risks of labile/fine fuel accumulation.

2. Materials and methods

2.1. Study site

The study was conducted on the Storrie Fire, which was started at Storrie, CA, USA on August 17, 2000. The fire quickly burned northward through the Sierran mixed-conifer forests of the Plumas and Lassen National Forests in Plumas County, CA. It burned about 21,000 ha on national forest land and 1300 ha on private forest land.

Climate is characterized by hot and dry summers and cold and wet winters. Based on 90-year weather records in Chester, CA, which is located 25 km north of the study plots, annual average maximum temperature was 16.8 °C and minimum temperature was −0.4 °C. Daily extreme high and low temperatures were 40.0 °C and −26.7 °C, respectively. Annual precipitation was 813 mm, with approximately 90% occurring as snow between November and April.

Soils at the study site are loamy, mixed, mesic Ultic Haploxeralfs. They fall within either the Holland family or Holland–Skalan family with volcanic parent materials. The depth to parent material is at least 1.0 m and slope angles range from 0% to 50%. Before the fire, the vegetation was a typical mixed conifer forest dominated by *Abies concolor*, *Abies magnifica*, *Calocedrus decurrens*, *Quercus kelloggii*, *Pinus ponderosa*, *Pinus lambertiana*, and *Pseudotsuga menziesii*; the understory was dominated by *Arctostaphylos*, *Castanopsis*, *Ceanothus*, and *Purshia.* The site index of the tallest trees in this mixed-conifer forest is 23 m at 50 years. The plots are located near latitude 40°5′N and 121°16′W.

2.2. Experimental treatments

The study plots were established with four differing post-fire forest management regimes as our treatments. We selected these treatment areas based on the location of existing forest cover created by the fire and post-fire management regimes. We were able to locate three 300-ha blocks on soils classified as loamy, mixed, mesic Ultic Haploxeralfs; each block included the four treatments. Live tree basal area was between 0 m^2 ha^{-1} on the salvage-logged treatments and 41 ± 6 m^2 ha^{-1} on the green canopy (control) treatment (Table 1). The green canopy experienced a low severity burn with little mortality; the post-fire managed treatments all experienced a high severity burn with nearly 100% tree mortality.

The intensively managed (IM) treatment was established on existing plantations managed by W.M. Beaty and Associates (Redding, California). Following the fire in August of 2000, salvage logging commenced immediately and was completed by the fall of 2001. Total sawlogs and biomass removal averaged 68.8 Mg ha^{-1} C (based on the company's timber sales records). Site preparation began in 2001 and the soil was mechanically altered by ripping. Approximately 740 one-year-old seedlings per hectare were planted between 2001 and 2004. The plantations consisted of either pure *P. ponderosa* or mixed conifer species including *P. ponderosa*, *P. lambertiana*, *P. menziesii*, *A. concolor*, and *C. decurrens*. Competing understory vegetation was controlled with herbicide applied twice within the first 5 years. Our treatment plots were located in pure ponderosa pine plantations planted in 2002 and 2003.

The salvage and planted (SP) treatment plots were selected at three plantations established by the US Forest Service in 2004. After the fire, the Forest Service proposed to salvage and plant some relatively flat areas, but the project was withheld due to an appeal of the project and subsequent reversal of the decision. Three years later, the Forest Service was able to salvage 62.5 Mg ha^{-1} C of equivalent wood materials from four areas for a total of 42 ha (based on records in Lassen National Forest). The salvaged area was planted with *P. ponderosa* at 1000 trees ha^{-1} in 2004. The competing vegetation was removed from circles of 1.5 m radius for the largest 50% of the planted seedlings in 2005. The soil was not mechanically altered by ripping.

The no salvage (NS) treatment plots were naturally regenerating burned stands. These plots were not managed following the fire. They burned at such high severity that the fire killed all of the trees on the plots; the resulting snags were not removed from the site. Snags, down wood, and a dense stand of thorny shrubs

Table 1
Stand characteristics across forest management treatment. Tree parameters included live trees and saplings at least 1.37 m tall.

	Stand characteristic across treatments			
Parameter	GC	NS	SP	IM
Tree basal area (m^2 ha^{-1})	41 ± 6		0.4 ± 0.1	3.8 ± 1.0
Ave. stem diameter (cm)	35 ± 7		3.2 ± 0.3	6.9 ± 0.8
Tree density (stems ha^{-1})	300 ± 59		360 ± 79	840 ± 120
Snag basal area (m^2 ha^{-1})	5.7 ± 1.3	34 ± 5		0.9 ± 0.4

Table 2
Species specific allometric equations used to calculate aboveground tree carbon storage and aboveground tree carbon sequestration. Allometric equation symbols are DBH (diameter at breast height), *D* (ground level diameter), and variables *a*, *b*, and *c* are constants. Units are biomass units and independent variable units, respectively.

Species	Allometric equation	*a*	*b*	*c*	Component	Units	Source
Abies concolor	$y = a * (D)^2 * H + c$	677.6747		0.03725	AB	kg, m	Pacific Southwest Research Station – Redding, unpublished data
Abies concolor	$\ln(y) = a + b * \ln(\text{DBH})$	4.36982	2.5043		AB	g, cm	Westman (1987)
Abies concolor	$y = a * e^{(b * \text{height})}$	0.0153	0.0369		AB	kg, cm	Pacific Southwest Research Station – Redding, unpublished data
Calocedrus decurrens	$y = a * D^b$	54.932	2.3719		AB	g, cm	Pacific Southwest Research Station – Redding, unpublished data
Calocedrus decurrens	$\ln(y) = a + b * \ln(\text{DBH})$	−11.7031	2.7818		ST and B	Mg, cm	Pacific Southwest Research Station – Redding, unpublished data
Pinus lambertiana	$y = a * (D)^2 * H + c$	474.865		0.03495	AB	kg, m	Pacific Southwest Research Station – Redding, unpublished data
Pinus lambertiana	$y = a * (\text{DBH})^2 + c * (\text{DBH})$	0.1212		0.8268	AB	kg, cm	Pacific Southwest Research Station – Redding, unpublished data
Pinus lambertiana	$y = a * e^{(b * \text{height})}$	0.0165	0.0319		AB	kg, cm	Pacific Southwest Research Station – Redding, unpublished data
Pinus ponderosa	$y = a * (D)^2 * H + c$	278.1443		0.40039	AB	kg, m	Pacific Southwest Research Station – Redding, unpublished data
Pinus ponderosa	$y = a * (\text{DBH})^b$	0.0599	2.4711		AB	kg, cm	Pacific Southwest Research Station – Redding, unpublished data
Pinus ponderosa	$y = a * e^{(b * \text{height})}$	0.1625	0.0147		AB	kg, cm	Pacific Southwest Research Station – Redding, unpublished data
Pseudotsuga menziesii	$y = a * (D)^b$	69.8758	1.62976		AB	g, cm	Brown (1978)
Pseudotsuga menziesii	$y = a * (\text{DBH})^2 + c * (\text{DBH})$	0.1215		1.137	AB	kg, cm	Brown (1978)
Quercus kelloggii	$y = a * (\text{DBH})^b$	0.0945	2.503		AB	kg, cm	Ter-Mikaelian and Korzkhun (1997) and Wiant et al. (1977)

occupied the plots by the time the experimental measurements began.

The green canopy (GC) plots were comprised of mixed conifer stands that experienced a low intensity fire. Following the fire, the dominant and co-dominant trees experienced low or no mortality; we assumed that the understory was consumed by the fire. The stands may have been thinned in the past, but there are no records of such pre-fuel treatment activity. This regime has been considered the managers' model of fire-adapted structure in this forest type. Photographs of the treatments can be obtained upon request to the authors.

Each treatment plot was approximately 2090 m^2 (ca. 150 ft by 150 ft). The plot corners were determined using a rangefinder and compass. We calculated the exact area (S) of each plot as follows:

$$S = 0.5 \times a \times b \times \sin\alpha + 0.5 \times c \times d \times \sin\beta \tag{1}$$

where a, b, c, d were the lengths of the four sides of the plot, α was the angle between a and b, and β was the angle between c and d. We used the exact plot size to calculate C storage and sequestration. Each plot contained five subplots, each 9.29 m^2 (100 ft^2) in area, which were used for some components of forest carbon storage and sequestration. The subplot and plot level data were summed to determine total ecosystem carbon storage and sequestration, as described below.

2.3. Aboveground tree carbon

We estimated the carbon content of all trees in each ~2020 m^2 treatment plot. Total aboveground tree biomass was calculated using species-specific allometric equations (Table 2) that predict aboveground tree biomass from tree diameter at breast height (DBH: 1.37 m above ground line). These allometric equations describe the biomass of live trees above the height of a short stump (~15 cm). Although stump mass of the green trees would have added slightly to the total biomass in the GC treatment, the biomass of these short stumps was neglected in this analysis. We used separate calculations to determine the aboveground portion of C in tall stumps and snags. Additional allometric equations were used to predict aboveground biomass for trees shorter than 1.37 m using tree diameter at ground level and/or tree height. We measured every single tree on each replicate, regardless of size. We assumed that the carbon content of tree biomass was 50% (Table 3) (Woodbury et al., 2007; Campbell et al., 2009) to convert biomass to carbon content and then scaled the tree carbon content to a per hectare basis (Mg ha^{-1} of C).

2.4. Understory carbon

Understory carbon content was defined as the carbon stored in aboveground understory biomass (Mg ha^{-1} of C), which included all annual and perennial plants. It was measured by destructively sampling all non-tree aboveground biomass in three 9.29 m^2 subplots per treatment plot. We measured the fresh weight in the field and subsampled to determine the water content. Subsamples were dried at 60 °C until they reached constant mass, and then weighed. Understory field biomass was converted to carbon content by subtracting the water content from the field fresh weight and assuming that the dry weight of understory biomass was 50% C (Thornton et al., 2002).

2.5. Snag carbon

We measured the carbon content for all snags within each treatment plot. Snags were defined as any standing dead tree at least 1.3 m in height. Snag carbon (Mg ha^{-1} of C) was calculated as:

$$\text{SN} = V \times D \times C \tag{2}$$

where V was volume, D was density, and C was the carbon concentration of the snag. Volume was estimated from an allometric equation that predicts volume from snag diameter measured 1.37 m from the forest floor (Zhang et al., 2005). The volume of the snags with broken tops was estimated as if they were cylinders. We estimated the density of snags by felling and sub-sampling cookies

Table 3
Carbon concentration of ecosystem components used to convert biomass to carbon.

Component	Carbon (%)	Source
Tree	50	Woodbury et al. (2007) and Hurteau and North (2010)
Snag	50	Hurteau and North (2010)
Stump	50	Janisch and Harmon (2002)
Understory	50	Thornton et al. (2002)
Coarse wood	50	Harmon (2008) and Hurteau and North (2010)
Fine wood	50	Hurteau and North (2010)
Duff	40	Campbell et al. (2009)
Soil coarse fraction	29.2–47.8	This study
Mineral soil	1.7–7.5	This study

Table 4
Specific gravity of snag and coarse wood for each decay class. The parameters were used convert coarse wood and snag biomass to carbon.

Wood component	Specific gravity
Decay class 1	0.38
Decay class 2	0.32
Decay class 3	0.23
Snag	0.41

*This study.

from snags of three size classes. Cookie subsamples were dried at 60 °C until they reached constant weight. Density was determined from the dry mass of the subsample and volume as determined by the volume displacement of water (Table 4). After weighing, samples were sprayed with carburetor sealant to minimize water uptake. They were then forced underwater and the apparent increase in mass was used as an estimate of water displacement (Williamson and Wiemann, 2010) and thus volume.

2.6. Stump carbon

The carbon content of aboveground stump biomass was estimated on three subplots per treatment plot. Stumps were defined as the aboveground tree bole biomass remaining after trees were harvested. Stumps were assumed to be cylindrical and stump volume was determined from field measurements of stump diameter and height. Stump carbon content was calculated as:

$$\mathrm{ST} = \pi \times \left(\frac{D_1}{2}\right)^2 \times H \times D_2 \times C \tag{3}$$

where D_1 was the diameter, H was height, D_2 was density, and C was the carbon concentration. We assumed that the carbon concentration of stumps is 50% (Janisch and Harmon, 2002). We assigned a decay class from 1 to 3. Decay class 1 was defined as stumps with greater than 70% of the biomass remaining. Decay class 2 had 30–70% of the biomass remaining. Decay class 3 had up to 30% of the biomass remaining and was characterized by cubical fragments and crumbling chunks. We took four subsamples from each of the decay classes to determine density following the same methods as described in Section 2.5 above (Table 4).

2.7. Coarse wood

We measured the carbon content of all coarse wood (CW) in three subplots per treatment plot. CW was defined as any downed woody debris over 5 cm in diameter (Mattson et al., 1987). The carbon content of CW was calculated as:

$$\mathrm{CW} = \pi \times \left(\frac{D_1}{2}\right)^2 \times L \times D_2 \times C \tag{4}$$

where D_1 was midpoint diameter, L was length within the plot, D_2 was the density of wood determined for each decay class, and C was the carbon concentration. We assumed that the carbon concentration of CW is 50% (Harmon, 2008). We measured the length and diameter of each piece of CW in the 9.29 m^2 subplots. As above, we assigned a decay class from 1 to 3 to determine the density.

2.8. Fine wood

Fine wood was collected on three subplots per treatment plot using a 2500 cm^2 duff sampler. FW was defined as plant biomass 1–5 cm in diameter. FW samples were dried in the oven at 60 °C until the samples reached constant mass and then weighed. To convert FW biomass to carbon, we assumed a carbon concentration of 50% (Prichard et al., 2000).

2.9. Duff

Duff was collected on three subplots per treatment plot using a 2500 cm^2 duff sampler. Duff was defined as plant biomass less than 1 cm in diameter. Samples were oven-dried at 60 °C until they reached constant mass and then weighed. Duff biomass was converted to carbon by assuming that the carbon concentration of duff was 40% (Campbell et al., 2009).

2.10. Soil carbon

Three soil samples per treatment plot were collected to determine soil carbon storage at 0–10 cm, 10–20 cm, and 20–30 cm depths. The samples were collected with a soil auger at a randomly selected corner of each of three subplots on each plot. Soil samples were oven dried at 60 °C until they reached constant mass and then separated into rock, coarse fragment (CF), and mineral soil components using a 2 mm mesh sieve. CF was defined as any organic material that would not pass a 2-mm mesh sieve (Page-Dumroese and Jurgensen, 2006). The dry weight of each component was determined and an Elemental Analyzer (CE Instruments, Milan, Italy) connected to a Delta^{plus} isotope ratio mass spectrometer (Finnigan-MAT, Bremen, Germany) was used to determine the carbon concentration for a subset of coarse fraction and mineral soil samples. One set of bulk density samples per treatment plot was taken using a 52.5-mm diameter slide hammer. Bulk density samples were taken at 1–6 cm, 11–16 cm, and 21–26 cm depths. These samples were assumed to represent 0–10, 10–20, and 20–30 cm depths, respectively. The soil coarse fraction and mineral soil carbon pools were estimated from the dry mass and bulk density of soil samples: the product of dry mass carbon concentration, divided by the bulk density at each depth.

$$\text{Soil C pool} = \frac{M}{V} \times D \times \frac{C}{100} \tag{5}$$

where Soil C pool (g cm^{-2} of C) represented either soil coarse fraction (SCF) or mineral soil C in a 10-cm soil layer, M was the component mass (g), V was the soil corer volume (cm^{-3}), D was the soil depth (10 cm), and C was the carbon concentration (%) of the component. The resultant volumetric C content was summed over the appropriate depths, and then scaled up to Mg ha^{-1}.

2.11. Total ecosystem carbon storage

Total ecosystem carbon storage was estimated by summing the individual carbon pools on each treatment plot. Total ecosystem carbon storage (TEC) was defined as:

$$\mathrm{TEC} = \mathrm{TC} + \mathrm{UC} + \mathrm{SN} + \mathrm{ST} + \mathrm{CW} + \mathrm{FW} + D + \mathrm{SCF} + \mathrm{MS} \quad (6)$$

where TEC was the sum of carbon content of individual ecosystem carbon pools (Mg ha^{-1} of C), TC was aboveground live tree, UC was aboveground understory, SN was snag, ST was stump, CW was coarse wood, FW was fine wood, D was duff, SCF was soil coarse fraction, and MS was mineral soil.

2.12. Aboveground carbon sequestration

For mature trees in the GC treatment, aboveground carbon sequestration was estimated for total carbon accumulated since the fire. Aboveground carbon sequestration (CS) was defined as:

$$\mathrm{CS} = \mathrm{TS} + \mathrm{US} \quad (7)$$

(in Mg ha^{-1}year^{-1} of C) where TS was aboveground tree carbon sequestration, US was aboveground understory carbon sequestration. Aboveground tree carbon sequestration (TS) was calculated as the average annual change in live aboveground tree carbon (TC) for individual trees (Jenkins et al., 2001).

$$\mathrm{TS}_m = \frac{(\mathrm{TC})_{t_1} - (\mathrm{TC})_{t_0}}{(t_1 - t_0)} \quad (8)$$

where TS_m was tree carbon sequestration of mature trees on the GC treatment, TC was the aboveground tree carbon (Mg tree^{-1} of C) and $t_1 - t_0$ was the 10 year increment (Hudiberg et al., 2009) between 2001 and 2010. We calculated the average annual change in diameter from the 10 year tree ring width between 2001 and 2010 because we wanted to represent the growth of trees as it recovered from disturbance. We expected that growth patterns in the first few years immediately following the fire and management installation would be low as the trees recovered from the initial disturbance, so the average annual growth would be more representative of the tree response in the 10 years following the fire. TC was calculated using the same species-specific allometric equations that were used to calculate aboveground tree carbon from the tree bole diameter (D) measurements (Table 2). The 2010 diameter, $D_{(t1)}$, was measured in the field as described in the section labeled aboveground tree carbon.

The 2001 diameter, $D_{(t0)}$, was determined from radial tree cores. The radial tree core increment was determined by sampling tree cores from three GC treatment plots. We selected 30 trees from each plot to represent the range of tree size and species. We measured the diameter and took two tree core samples at perpendicular directions at 1.37 m from the forest floor. We also measured the 10-year radial increments (or fresh length, L_w) of the 2001–2010 rings using a caliper to 0.1 mm in the field. The cores were dried and the total dry length (L_d) of the 2001 to 2010 tree rings was measured using a caliper. The shrinkage rate (S) was determined using samples that had been measured outdoors as follows:

$$S = \frac{L_w - L_d}{L_w} \quad (9)$$

The shrinkage was determined to be 3.02%. The 10 year radial increment (r) was calculated using the dry tree ring length measurements for tree ring lengths that were not measured in the field:

$$r = L_w = \frac{L_d}{(1 - S)} \quad (10)$$

Finally, the 2001 diameter on the subsampled trees, $D_{(t0)}$, was calculated as:

$$D_{(t0)} = D_{(t1)} - [2 \times k \times r] \quad (11)$$

where k was the bark correction factor (Wykoff et al., 1982).

In order to estimate the annual tree carbon sequestration for all of the trees in each plot, we developed a regression equation using the 2010 diameter measurements and tree rings from the subsampled trees to predict 2001 tree diameter from the 2010 measured diameter:

$$D_{(t0)} = -4.172 + 0.998 \times D_{(t1)} \quad (12)$$

where the $R^2 = 0.996$.

We used a separate TS calculation for the seedlings planted on the IM and SP treatments because they were planted up to 4 years after the fire:

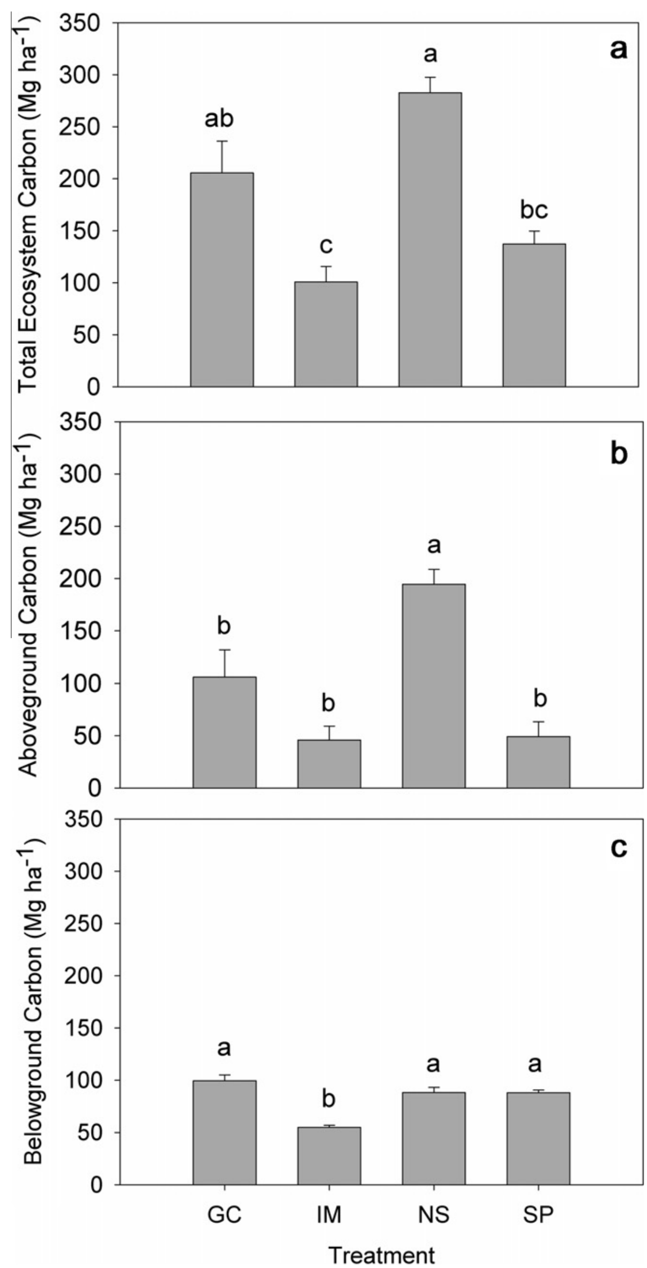

Fig. 1. Carbon storage across management treatment (Mg ha^{-1}) for (a) total ecosystem carbon (aboveground and belowground C storage); (b) total aboveground C (tree, sapling, snag, understory, FW, CW); (c) soil carbon (soil coarse fraction and MS C from 0 to 30 cm depth). Forest management treatments include: green canopy (GC), intensive management (IM), no salvage (NS), and salvage and planted (SP). Bars that are not connected by the same letter are significantly different ($\alpha = 0.05$).

Table 5
Aboveground carbon storage (Mg ha^{-1} of C) for each component across forest management treatments. Letters not connected by the same letter horizontally are significantly different ($\alpha = 0.05$).

Component	Carbon pool (Mg ha^{-1})			
	GC	IM	NS	SP
Tree	81 ± 30*a*	4.1 ± 1.2*b*	0*b*	0.4 ± 0.1*b*
Snag	6.5 ± 1.8*a*	0.5 ± 0.2*a*	82 ± 14*b*	0*a*
Stump	0.6 ± 0.4*a*	11 ± 3*a*	8.2 ± 4.9*a*	2.8 ± 0.7*a*
Understory	1.3 ± 0.6*a*	0.2 ± 0.1*a*	5.2 ± 1.3*a*	5.6 ± 2.4*a*
Coarse wood	9.9 ± 5.7*a*	19 ± 11*a*	81 ± 8*b*	23 ± 9*a*
Fine wood	1.4 ± 0.2*a*	9.5 ± 1.2*a*	13 ± 3*a*	13 ± 5*a*
Duff	5.5 ± 0.2*a*	1.8 ± 0.5*a*	5.7 ± 2.6*a*	4.6 ± 0.9*a*

$$TS_s = \frac{TC_{(t1)}}{(t_1 - t_0)} \quad (13)$$

where TS_s was seedling tree carbon sequestration (Mg ha^{-1} year^{-1}), $TC_{(t1)}$ was the seedling carbon (Mg ha^{-1}) in 2010, t_1 was the year 2010, and t_0 was the year of the start of the growth interval. On the IM and SP treatments, t_0 was the year when the seedlings were planted (see Section 2.2). The individual TS estimates from seedlings and trees were summed on each plot and scaled to an area basis to determine annual tree carbon sequestration.

Aboveground understory carbon sequestration was determined as:

$$US = \frac{UC}{(t_{2010} - t_{2001})} \quad (14)$$

where US was understory carbon sequestration (Mg ha^{-1} year^{-1}), UC was understory carbon storage (Mg ha^{-1}), and t was the year, on the GC, IM, and NS treatments. On the SP treatment, we accounted for the partial understory removal in 2005. We estimated the proportion of the plot that fell within the treated area, which was comprised of circles of 1.5 m radius around the largest 50% of the planted trees:

$$US = p \times \frac{UC}{(t_{2010} - t_{2001})} + (1 - p) \times \frac{UC}{(t_{2010} - t_{2001})} \quad (15)$$

where t was the year, and p was the proportion of plot area with radial understory removal:

$$p = \pi r^2 \times \frac{t}{a} \times 0.5 \quad (16)$$

where r was the1.5 m radius, t was the number of trees per plot, and a was the plot area (m^2).

2.13. Data analysis

Treatment differences in TEC, aboveground C storage, and soil C storage were evaluated using the Tukey–Kramer honest significant difference (HSD) test to make multiple comparisons among treatment means. The HSD was also used to determine treatment differences in less combustible coarse fuels, fine fuels, and individual C storage pools. Statistical analysis was performed using JMP 9 statistical software (SAS, Cary, NC, USA).

To scale plot level estimates of TEC to a land area basis, carbon pool estimates were required for each plot. Due to errors in sample labeling and loss, we were left with 14 missing soil C data points. We used the average of the plot measurements to fill in these missing data. This allowed us to sum each individual ecosystem component of TEC for each replicate. We also excluded one of the coarse wood plots on the NS treatment because satellite imagery (Google earth, http://earth.google.com) showed an anomalously high density of coarse wood when this plot was compared to the surrounding landscape.

3. Results

3.1. Carbon storage across forest management regimes

Total ecosystem carbon storage differed among post-fire management regimes. Total ecosystem carbon storage was 283 ± 15 Mg ha^{-1} in the no salvage (NS) and 206 ± 31 Mg ha^{-1} in the green canopy (GC) treatment (Fig. 1a). These values were significantly higher than those in the plots that were salvaged. The salvage planted (SP) and intensively managed (IM) treatments had values of 137 ± 13 Mg ha^{-1} and 101 ± 15 Mg ha^{-1}, respectively. There were also management differences in aboveground and soil carbon storage (Fig. 1b and c). NS stored significantly more carbon aboveground (194 ± 14 Mg ha^{-1}) than all the other treatments (Fig. 1b); aboveground storage was 106 ± 26 Mg ha^{-1} on the GC, 49 ± 14 Mg ha^{-1} on the SP, and 46 ± 13 Mg ha^{-1} on the IM. The IM treatment stored the least C in the soil (55 ± 2 Mg ha^{-1}); this was significantly lower than the other treatments. The other treatments held 100 ± 6 Mg ha^{-1} in the GC, 88 ± 5 Mg ha^{-1} in NS, and 88 ± 2 Mg ha^{-1} in the SP treatment.

3.2. Distribution of component C storage among treatments

The distribution of carbon among aboveground pools also varied with post-fire management. The largest pool in GC was live tree biomass, with 81 ± 30 Mg ha^{-1} of C (Table 5). The largest pool in IM and SP was coarse wood, which stored 19 ± 11 Mg ha^{-1} and 23 ± 9 Mg ha^{-1}, respectively. The NS treatment stored nearly equal amounts in coarse wood and snag biomass, with 81 ± 7 Mg ha^{-1} and 82 ± 14 Mg ha^{-1}, respectively.

The distribution of carbon among belowground pools was less variable among treatments compared to the aboveground components. The greatest differences in soil C storage were in the top 0–10 cm of soil. In this layer the IM stored 22 ± 1 Mg ha^{-1} in the sieved mineral soil and 1.8 ± 0.8 Mg ha^{-1} in the soil coarse fraction (Table 6). In contrast, the GC stored 47 ± 6 Mg ha^{-1} in the sieved mineral soil from this layer and 1.2 ± 0.7 Mg ha^{-1} in the soil coarse fraction. There were no post-fire management differences in the mineral or soil coarse fraction carbon storage at 10–20 cm and 20–30 cm depths.

There were also shifts in the storage of carbon between labile/fine-fuel and recalcitrant carbon pools with fire and post-fire management. For example, the SP treatment removed all the carbon sequestered in snags and the IM treatment removed over 99% of the snag C compared to the NS treatment. In the NS treatment, fire shifted the carbon stored in large-diameter trees to the snag C pool. The shift in carbon storage among individual ecosystem C pools resulted in treatment differences in carbon storage in fire resistant C pools (Fig. 2a). The NS stored 258 ± 10 Mg ha^{-1} and the IM stored 89 ± 15 Mg ha^{-1} in recalcitrant C pools. There were no significant management differences in labile/fine fuel C storage (Fig. 2b) or individual components of labile/fine fuels, which include the understory, fine wood or duff C pools (Table 5).

3.3. Carbon sequestration

Post-fire management affected components of carbon sequestration, but not total carbon sequestration. Total aboveground carbon sequestration was 1.6 ± 0.7 Mg ha^{-1} year^{-1} of C on the green canopy, 0.7 ± 0.3 Mg ha^{-1} year^{-1} on the SP, 0.5 ± 0.1 Mg ha^{-1} year^{-1} on the NS, and 0.5 ± 0.1 Mg ha^{-1} year^{-1} on the IM treatment (Fig. 3a). There were no treatment differences in understory carbon sequestration. Understory carbon sequestration was 0.1 ± 0.1 Mg ha^{-1} year^{-1} of C on the green canopy, 0.7 ± 0.3 Mg ha^{-1} year^{-1} on the SP, 0.5 ± 0.1 Mg ha^{-1} year^{-1} on the NS, and

Table 6
Components of belowground carbon storage with soil depth and forest management treatments. Values that are not connected by the same letter horizontally are significantly different (α = 0.05).

Component	Carbon pool (Mg ha^{-1})			
	GC	NS	SP	IM
Soil coarse fraction				
0–10 cm	1.2 ± 0.7*a*	3.7 ± 0.4*a*	2.0*a*	1.8 ± 0.8*a*
10–20 cm	7.1 ± 2.7*a*	1.9*a*	6.7 ± 0.6*a*	1.1 ± 0.2*a*
20–30 cm	1.3*a*	4.3 ± 2.2*a*	0.9 ± 0.3*a*	0.2 ± 0*a*
Mineral soil				
0–10 cm	47 ± 6*a*	36 ± 3*ab*	35*ab*	22 ± 1*b*
10–20 cm	26 ± 2*a*	26 ± 1*a*	22 ± 1*ab*	16 ± 1*b*
20–30 cm	17*a*	16 ± 2*a*	21 ± 2*a*	14 ± 1*a*

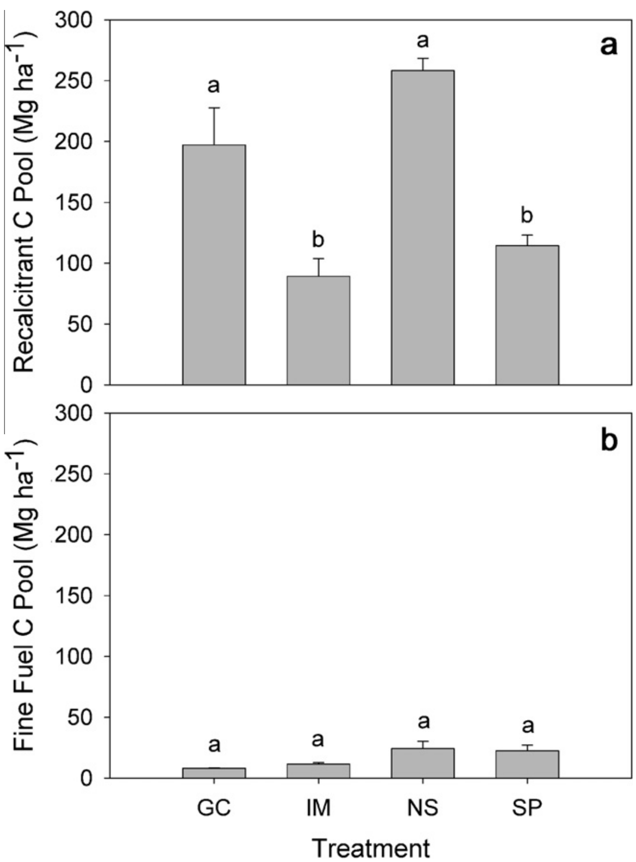

Fig. 2. Ecosystem carbon storage in (a) recalcitrant C pools including tree, snag, stump, coarse wood, and all soil carbon and (b) fine fuels including understory, fine wood, and duff. Forest management treatments include: green canopy (GC), intensive management (IM), no salvage (NS), and salvage and planted (SP). Bars that are not connected by the same letter are significantly different (α = 0.05).

0.02 ± 0.02 Mg ha^{-1} year^{-1} on the IM treatment (Fig. 3b). Aboveground tree carbon sequestration was significantly greater on the GC than the NS treatment; values were 1.5 ± 0.6 Mg ha^{-1} year^{-1} on the GC, 0 Mg ha^{-1} year^{-1} on the NS, 0.07 ± 0.0 Mg ha^{-1} year^{-1} on the SP, and 0.5 ± 0.1 Mg ha^{-1} year^{-1} on the IM (Fig. 3c).

4. Discussion

4.1. Carbon storage and sequestration among treatments

Post-fire management affected total C storage and the distribution of C among ecosystem pools. Of the burned and treated sites,

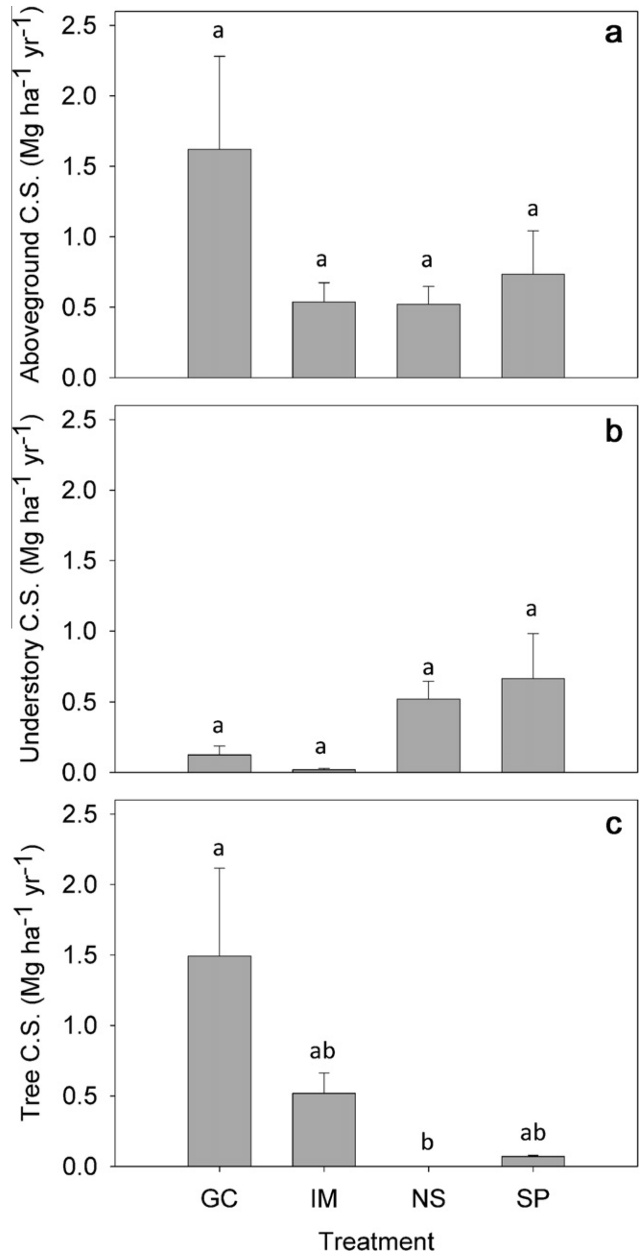

Fig. 3. Carbon sequestration (CS) across forest management treatment averaged over 10 years following fire, (a) total aboveground carbon sequestration (tree and understory), (b) aboveground understory carbon sequestration, and (c) aboveground tree carbon sequestration. Forest management treatments include: green canopy (GC), intensive management (IM), no salvage (NS), and salvage and planted (SP). Bars that are not connected by the same letter are significantly different (α = 0.05).

the NS treatment had the highest total C storage due to the large snag and coarse wood pools. The least total ecosystem C was stored in IM and SP treatments due to the reduction of the tree C by the fire and snag C by the logging. The removal of sawlogs and biomass from post-fire salvage logging averaged 68.8 Mg ha^{-1} of C on the IM and 62.5 Mg ha^{-1} of equivalent wood materials on the SP. In addition, soil C was reduced in the 0–10 cm layer on the IM treatment. The IM removed 44. 7 Mg ha^{-1} of soil C from 0 to 30 cm depth compared to the GC (Fig. 1c). Of the 44.7 Mg ha^{-1} removed on the IM, approximately half, 22 Mg ha^{-1}, was lost in the top 0–10 cm of mineral soil (Table 6).

Soil C pools are often incompletely measured or ignored in estimates of carbon storage although soil carbon has been estimated to represent approximately 40% of terrestrial carbon (Dixon et al., 1994; Kindermann et al., 2008). In our study, soil C contributed between 31% and 64% to total ecosystem carbon storage. Our estimates agree with a previous model of C dynamics throughout the state of California, which estimated that approximately half of total C was in the soil (Turner et al., 1995). The potential change in storage following disturbance is variable (Johnson, 1992). We observed a 45% decrease in total soil C storage in the IM treatment compared to the GC. These reductions were attributed to a decrease in the mineral soil fraction. Two meta-analyses (Johnson, 1992; Johnson and Curtis, 2001) show similar reductions in soil C storage, however the responses were dependent upon disturbance severity and management. For example, soil C storage generally increased after sawlog harvest, decreased following whole tree harvest or prescribed fire, and increased after wildfire (Johnson and Curtis, 2001; Powers et al., 2005). A more recent meta-analysis concluded that harvesting generally reduced soil C by 8%, with the reductions occurring primarily in the forest floor (organic horizon) and not the deeper mineral soil layers (Nave et al., 2010). Other studies observed stable to slight increases in mineral soil C following harvest, however the responses also varied with management intensity and ecosystem type (Sanchez et al., 2006).

Snags also stored a large proportion of total ecosystem carbon in the NS treatment. On the SP and IM treatments, carbon storage had been substantially reduced because the fire-killed trees had been removed during salvage-logging. Snag C and soil C represent large and variable carbon storage pools that should be included in carbon budget estimates.

Carbon sequestration was expected to differ among treatments. However, post-fire management did not affect rates of total aboveground carbon sequestration 10 years after fire. We expected to observe higher total carbon sequestration on the GC treatment resulting from greater stand level tree carbon sequestration rates. In fact, we did observe greater tree carbon sequestration rates on the GC treatment compared to the NS, which had no trees. In contrast, the higher aboveground C in the GC treatment was not different from the IM and SP. This lack of difference may have resulted from the high variability among plots.

It is important to recognize that this storage occurred in the 10 years immediately after the fire. After 10 years, stands have not recovered from disturbance in terms of either site occupation (Pretzsch, 2009) or C sequestration (Fig. 3). We expect that stand sequestration rates will reach GC rates at a later successional stage. The amount of time it takes for the seedlings to recover from disturbance depends upon disturbance type and severity, species composition, post-fire management, and stand density. Intensive management will accelerate this process. For example, it took only 36 years for an intensively managed plantation to carry as much aboveground carbon as pre-fire stands that were approximately 70 years old (Zhang et al., 2008). This leads us to speculate that the C losses on the IM treatment would be offset by the acceleration of stand establishment. It would require a modeling exercise to determine whether and when the offset would occur.

Differences in pre-fire conditions could have influenced our measurements of post-fire C. The study was initiated after the fire, so it was not possible to determine pre-fire carbon stocks. However, our study plots were chosen in similar landscapes with similar soils to minimize pre-fire differences. Further, if we compare live tree basal area on the green canopy treatments to snag basal area on the no salvage treatment, the snag basal area is only a little lower (41 vs. 34 m^2/ha in Table 1). We would not expect them to be exactly the same because the fire may have consumed some of the trees or they may have fallen following fire. Nonetheless, the similarity in these values supports our assumption that the sites were similar before they were treated.

4.2. Error analysis

Sources of error in the carbon measurements include inaccurate and incomplete measurements. One possible source of inaccuracy was the use of inappropriate allometric equations. We minimized inaccuracy by using species-, and where possible, region-specific allometric equations. However, sometimes this was impossible. For example, for *Q. kelloggii*, we applied allometric equations developed for *Quercus velutina* Lam. located in West Virginia. Where possible, we also applied different allometric equations, developed from saplings, to predict C storage in saplings. However, we were forced to use an allometric equation developed for *A. concolor* saplings to estimate C for four individual *P. menziesii* saplings. Error may also have resulted from our snag C estimates, which were derived by assuming that snags are cylinders (Kueppers et al., 2004). This may not be accurate because although broken snags approximate cylinders, they do taper with height and some snags had branches remaining. Future studies could reduce these errors by generating allometric equations for large trees of all species and improving estimates of snag C.

Rates of decomposition were accounted for in our estimates of carbon pools by assigning a decay class to snags and stumps, which we related to wood density and used to reduce our volume to mass conversions. When the sequestration rates were calculated from these pools, the influence of decomposition had thus already been accounted for. Whether this was sufficient is not known. We explored the possibility of further accounting for this flux and concluded that it was too variable to include in the current analysis. Decomposition rates would vary among species, among pools, and probably among treatments given the variable temperature and moisture conditions induced by the treatments. Future work will be needed to address this issue more quantitatively on these sites.

Finally, errors due to incomplete measurements were mostly belowground. We did not, for example, include the carbon in coarse roots or the belowground portions of stumps and snags. We explored the use of coarse-root allometric equations (Jenkins et al., 2003) to live trees and snags and found that coarse root C would increase TEC by 0.4–7% depending upon treatment. We did not include these estimates because the equations were not species–specific. Worse, the coarse roots had probably decayed since the fire and we did not measure the decline in their density; moreover, we would expect that decay rates would differ among treatments. Given these limitations, we decided not to try to estimate coarse root C. Our soil C estimates would also have been more accurate if it had not been necessary to fill in data for the missing samples. Finally, our soil sampling was limited to three soil samples per replicate, which would increase the likelihood of a type II error. We recognize that soil C is often variable (Page-Dumroese et al., 2006) and that we would have benefited from more replication.

4.3. Managing for carbon sequestration and storage

Managing forests for carbon storage and sequestration has become an increasingly important priority to help offset anthropogenic CO_2 emissions. Forest carbon management strategies may include maximizing total ecosystem carbon storage, retaining carbon in recalcitrant pools, or promoting carbon sequestration. The GC and NS treatments had the highest total carbon storage. They also stored the highest proportion of total carbon in recalcitrant C pools. This is important because recalcitrant pools such as large diameter trees, soil, and snags retain C for longer periods of time than labile pools.

This study underlines the importance of soil carbon storage. Previous studies have shown that the reduction of soil C following forest management is not ubiquitous; the effects vary with time since disturbance, soil type, and forest management (Black and Harden, 1995; Breshears and Allen, 2002; Page-Dumroese and Jurgensen, 2006; Boerner et al., 2008). Because this site underwent mechanical site preparation, some of the reduction may have been due to the mixing of the deep soil materials into the surface horizons. However, the reductions occurred in the 0–20 cm depth without an increase in the 20–30 cm depth, so it is unlikely that downward mixing of organic matter was sufficient to explain the reduction.

4.4. Managing for carbon sequestration and fire

Managing for carbon sequestration may increase fire hazard. Although carbon storage is often favored by climate mitigation efforts, fire hazard reduction often seeks to reduce fuel accumulation by lowering forest carbon density. Managing for both climate mitigation and fire hazard reduction may be achieved by managing to favor recalcitrant carbon storage pools and minimize storage in labile/fine fuels. As noted earlier, recalcitrant pools include large diameter trees, snags, coarse wood, and soil. Labile/fine fuels include understory, fine wood, and duff.

Reducing C storage in labile/fine fuels may be achieved by controlling understory C that burns readily and may propagate crown fires. In this study, the contribution of understory C is relatively low compared to total ecosystem carbon storage, so loss of the understory would have little effect on carbon sequestration. We therefore recommend thinning understory C as a means to reduce labile fuel loading while maximizing C storage. Post-fire strategies favoring recalcitrant carbon may reduce fuels that contribute to increased fire risk and do not contribute a large proportion of C to long term storage. Recalcitrant C is also favored by reducing soil disturbance (Lal, 2004; Page-Dumroese and Jurgensen, 2006).

A complete accounting for C storage would consider forest products (Johnson et al., 2005; Mitchell et al., 2009). For example, partial or complete removal of snags from the forest with post-fire planting could theoretically increase overall long-term carbon storage by promoting forest regeneration and storage of C in recalcitrant forest products offsite. Accounting for off-site storage is beyond the scope of our study. The objectives of our study were to determine C storage and sequestration in previously burned forests, not a full carbon accounting in forest products. An accurate accounting for the carbon removed from the site is complex and would require additional information about the C storage in sawlogs, the post-harvest use, and an accounting of the C required for processing and shipment.

While this study provides management opportunities to balance carbon sequestration efforts with fire, these opportunities should consider management implications besides fire hazard and carbon sequestration. Post-fire salvage should consider the effects on ecosystem function in terms of vegetation regeneration, animal and plant diversity, hydrology, erosion, and nutrient cycling (Serrano-Ortiz et al., 2011; Donato et al., 2006; Castro et al., 2010). For example, fuel reduction treatments that remove shrub cover should consider the ecological significance of *C. cordulatus* as a nitrogen fixer. The presence of nitrogen-fixing shrubs could influence soil C and N dynamics (Boerner et al., 2008) because a significant proportion of N lost from fire disturbance may be offset by nitrogen fixation (Agee, 1993). It remains uncertain how much plant available nitrogen *C. cordulatus* may be adding to the soil and what are its effects on soil fertility and total plant productivity. Managers must also balance management goals with the cost of implementing post-fire treatments at the landscape and regional scale. These management considerations are becoming increasingly important as various lines of evidence suggest that frequency, extent, and – in some cases – the severity of fire is increasing across the western US (Miller et al., 2009; Dillon et al., 2011).

5. Conclusion

Forest managers are faced with complex decisions about how best to manage forest systems for carbon sequestration, fire risk, forest products, and ecosystem processes. Differing management strategies may be implemented depending upon management goals. The results of our study suggest that in fire-prone ecosystems, the green canopy forest and no salvage logging treatments store the most carbon 10 years after a wildfire. Long-term management goals should account for how stand structural changes may shift the distribution of carbon between labile/fine fuels and recalcitrant pools as the stands continue to recover.

To manage for carbon sequestration while still reducing fire hazard, management efforts should maximize carbon storage and carbon sequestration rates in large diameter live trees and other recalcitrant pools, including soils and dead trees, if possible. They should minimize carbon storage in saplings, understory, and surface fuels. The results of our case study provide managers with important background information to make decisions about how to mitigate climate change and reduce future fire risk in a fire-prone Sierra Nevada forest ecosystem.

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