ECOSYSTEM CARBON DYNAMICS FOLLOWING FIRE RISK MITIGATION TREATMENTS IN A MIXED-CONIFER FOREST, SIERRA NEVADA, CALIFORNIA

A Thesis in Ecology by Morgan L. Wiechmann

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Thesis Abstract

Forests can sequester carbon from the atmosphere, helping mitigate climate change. Fire suppression, increasing temperature, and prolonged drought have resulted in increased wildfire frequency and severity in recent decades in the western United States. Large and severe wildfires impact the carbon cycle both through direct emissions and reduced sequestration resulting from tree mortality. Mechanical thinning and prescribed burning can reduce fire severity and carbon loss when wildfire occurs. Ecosystem carbon dynamics altered during treatment implementation include carbon removal (thinning) and carbon emissions (prescribed fire), and following treatment include carbon sequestration (biomass growth) and charcoal production formed from burning vegetation. To accurately assess long-term carbon costs and benefits of thinning and burning these dynamics need to be evaluated.

We quantified the 10-year post-treatment carbon stocks from a full-factorial experiment for three levels of thinning and two levels of burning in a mixed-conifer forest in California’s Sierra Nevada. Our results indicate that (1) treatments that retain large trees quickly recovered the initial carbon loss, (2) the carbon emitted from prescribed fire was recovered within the historical fire return interval, and (3) in treatments that included prescribed fire, fire-tolerant species experienced similar percent change in carbon as fire-intolerant species. Selective thinning to reduce forest density, followed by prescribed burning to reduce surface fuels, can help stabilize forest carbon and restore ecosystem resilience. By retaining large diameter trees and selectively thinning midsize (25-75 cm DBH) fire-intolerant tree species, the remaining tree carbon is aggregated in fewer, larger trees that are resistant to subsequent wildfire-induced mortality.
Additionally we compared charcoal carbon produced from coarse woody debris and fine woody debris charred during prescribed burning. We quantified post-treatment charcoal carbon formation in organic matter and the top 5 cm of mineral soil from downed logs (> 30 cm diameter) that were present prior to treatment. Our results indicate that there was no difference in the amount of charcoal carbon produced from combusted coarse woody debris and combusted fine woody debris. We also compared treatment effects of charcoal production and found that treatments that included burning had significantly more charcoal carbon than the control. Charcoal carbon represented 0.19% of total ecosystem carbon, a relatively small fraction of total ecosystem carbon. However, charcoal carbon is long-lived and will likely continue to accumulate with repeated burning, leading to additional increases in long-term soil carbon storage. Given increasing efforts to reduce high-severity wildfire risk with thinning and burning, our findings help improve our understanding of the effects these treatments have on ecosystem carbon flux.
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Chapter 1: Background

Absorbing billions of tons of carbon dioxide annually, forest ecosystems are an important component of the global carbon (C) cycle and assist in the mitigation of climate change (Canadell and Raupach, 2008; Pan et al., 2009). Over large geographic areas, disturbances can negatively impact this ecosystem service, releasing C to the atmosphere for years to decades (Campbell et al., 2007; Chambers et al., 2007; Dore et al., 2008; Kurz et al., 2008). In many dry western forest types that historically evolved with high frequency, low severity wildfire, recent fire-exclusion and associated changes in forest structure and composition, coupled with prolonged drought have increased the risk of stand-replacing wildfire events. Fire emissions associated with wildfire events represent a large and highly variable component of the United States C budget (Wiedinmyer and Neff, 2007), and have garnered increasing attention because of C emission contribution to climate forcing (Kloster et al., 2010).

Altering the forest structure and decreasing fuel loads through mechanical thinning and prescribed burning can increase forest resistance to stand-replacing wildfires (Agee and Skinner, 2005). Though treatment implementation requires C removal (thinning) and C emissions (prescribed fire), two factors that have received increasing attention because of the role of forests in mitigating climate change. By reducing the risk of stand-replacing wildfires forest managers may also be protecting against potential C losses through reduced direct wildfire C emissions and increased resistance to vegetation type switching following the wildfire event (e.g. grasslands or shrublands) (Hurteau and Brooks, 2011). The tradeoff between the C debt incurred during treatment and C protection from reduced wildfire emissions has been the
focus of recent debate. In Chapter One we provide additional insight to this debate by quantifying forest C dynamics in a Sierra Nevada mixed-conifer forest over the 10-years following fuel reduction treatments and assess the post-treatment forest growth to recover the treatment implementation C debt.

While various studies have investigated forest C losses from fire restoration management (Finkral and Evans, 2008; Stephens et al., 2009; Hurteau and Brooks, 2011), few studies have investigated the potential long-term C storage resulting from prescribed fire through the production of recalcitrant, pyrogenic charcoal. In Chapter Two we improve estimates of charcoal C inputs resulting from prescribed burning by quantifying the effects of pre-fire fuel load and fuel type on charcoal formation resulting from different fuel reduction treatments that included prescribed burning in a California mixed-conifer forest.

1.1 Literature Cited


Chapter 2: Mitigating climate change and restoring ecosystem resilience: An analysis of 10-year post-treatment carbon dynamics in a mixed-conifer forest, Sierra Nevada, California, USA

2.1 Abstract

Forests sequester carbon from the atmosphere, helping mitigate climate change. In fire prone forests, fire events result in direct and indirect emissions of carbon. If fire-induced tree mortality is high, a forest can transition from a carbon sink to source. Mechanical thinning and prescribed burning can reduce fire severity and carbon loss when wildfire occurs. However, treatment implementation requires carbon removal (thinning) and carbon emissions (prescribed fire) to reduce high-severity fire risk. The carbon debt incurred during treatment may be recovered by subsequent tree growth, although there is much uncertainty in this payback time. To assess the long-term carbon costs and benefits of thinning and burning treatments, we quantified the 10-year post-treatment carbon stocks from a full-factorial experiment involving three levels of thinning and two levels of burning in a mixed-conifer forest in California’s Sierra Nevada. Our results indicate that (1) treatments that retain large trees quickly recovered the initial carbon loss, (2) the carbon emitted from prescribed fire was recovered within the historical fire return interval, and (3) in treatments that included prescribed fire, fire-tolerant species experienced similar percent change in carbon as fire-intolerant species. Selective thinning to reduce forest density, followed by prescribed burning to reduce surface fuels, can help stabilize forest carbon and restore ecosystem resilience. By retaining large diameter trees and selectively thinning midsize (25-75cm DBH) fire-intolerant
tree species, the remaining tree carbon is aggregated in fewer, larger trees that are resistant to subsequent wildfire-induced mortality.

### 2.2 Introduction

Forests store substantial amounts of carbon (C) and their contribution to the global terrestrial C sink is significant (Canadell and Raupach, 2008). This ecosystem service provides a climate-regulating benefit of approximately 2.4 Pg C yr\(^{-1}\) (Pan et al., 2011). Yet, disturbances can negatively impact this ecosystem service over large geographic areas, resulting in a transition from C sink to source for years to decades (Campbell et al., 2007; Chambers et al., 2007; Dore et al., 2008; Kurz et al., 2008). Many terrestrial systems have evolved with fire, a globally distributed disturbance, the frequency of which varies across biomes and through time (Bowman et al., 2009). Worldwide, biomass burning is a significant contributor to total C emissions, and this emission source has garnered increasing attention because of its role in climate forcing (Kloster et al., 2012). Average annual global emissions from biomass burning were 2.0 Pg C yr\(^{-1}\) from 1997-2009, with forest fires contributing approximately 15% of total C emissions (van der Werf et al., 2010). While climate change is a global issue, forest-based C sequestration requires local ecological knowledge, including an understanding of disturbance dynamics (Hurteau et al., 2013b).

In the western United States wildfire had a dynamic relationship with climate from 500 CE to the 1800s (Marlon et al., 2008; Marlon et al., 2012). During the 1800s, biomass burning increased with European settlement, causing a peak in fire activity (Marlon et al., 2012). By the 20\(^{th}\) century, forest recovery following logging and human factors (i.e., grazing, fire suppression) led to a large fire deficit, causing substantial ecological changes (Covington and Moore, 1994;
Since 1905, when federal fire suppression policy was enacted, stem density of shade-tolerant, fire-intolerant species has increased and surface fuels have accumulated in many dry forest types (Stephens and Ruth, 2005; Scholl and Taylor, 2010). The decrease in fire activity during much of the 20th century, coupled with the changing climate, has altered fire type and size. In recent decades the probability of large wildfires and wildfire severity has increased in much of the western US (Westerling et al., 2006; Krawchuk et al., 2009; Miller et al., 2012), but is still below the pre-fire exclusion historical average. In the Sierra Nevada mountains of California, mixed-conifer forests developed with low to moderate severity fires approximately every 11-30 years (Taylor and Skinner, 2003; North et al., 2005). Fire-exclusion and associated changes in forest structure and composition have created forests that are more susceptible to stand-replacing wildfires (Miller et al., 2009) that are beyond the historical range of variability. The increase in high-severity, stand-replacing fire events can result in mortality of much of the overstory, triggering changes in structure and composition of vegetation and converting forests from a C sink to a C source (Stephens and Moghaddas, 2005; Dore et al., 2008; Westerling and Bryant, 2008; Hurteau et al., 2009).

In these fire-adapted ecosystems, treatments that reduce tree density and surface fuels are a common management practice used to lower the risk of stand-replacing fire. Fuel treatments include various levels of mechanical thinning, prescribed burning and a combination of both to reduce fuel quantity and fuel continuity (Agee and Skinner, 2005). Fuel treatments can also meet additional management objectives, including restoration of native species (Wayman and North, 2007; Laughlin and Fule, 2008), protection from insect and pathogen outbreaks (Campbell et al., 2012), and improving tree regeneration (Zald et al., 2008).
However, implementing fuel reduction treatments may also pose a threat to certain management objectives. For example, fuel reduction treatments may create favorable conditions for invasive plant establishment (Keeley, 2006; Collins et al., 2007). Additionally, treatment implementation requires C removal (thinning) and C emissions (prescribed fire), two factors that have received increasing attention because of the role of forests in mitigating climate change. The tradeoff between the C debt incurred during treatment and C protection from reduced wildfire emissions has been the focus of recent debate. Some research has found that treatments that reduce wildfire severity yield a net C benefit when wildfire occurs (Hurteau et al., 2008; Hurteau and North, 2009; Stephens et al., 2009; North and Hurteau, 2011). However, other studies suggest that, over an entire disturbance cycle, long-term C storage of a treated forest is similar to a forest that is untreated and subject to a stand-replacing fire based on the idea that fuel reduction treatments do not increase C carrying capacity (Mitchell et al., 2009; Campbell et al., 2012). Largely absent from this discussion have been the effects of thinning and burning on long term post-treatment tree growth. Understanding local ecological conditions following fuel reduction treatments may provide insight into the capacity of post-treatment forest growth to recover the C debt incurred during treatment.

2.3 Objectives

The objective of our research was to quantify forest C dynamics in a Sierra Nevada mixed-conifer forest over the 10-years following fuel reduction treatments. We hypothesized that a range of C balance outcomes would exist 10-years following treatments. We predicted 1) post-treatment tree growth would sequester more C than was emitted from prescribed burning treatments; 2) understory thinning treatments with and without prescribed burning would
recover the C removed and emitted during treatment within the ten year period, and overstory thinning treatments would continue to have a C debt; and 3) fire-tolerant species (e.g. Pinus spp.) would have increased C gain relative to fire-intolerant species (e.g. Abies spp.) in treatments that included burning.

2.4 Materials and Methods

Study Site

The Teakettle Experimental Forest is a 1300 ha reserve of old-growth, mixed-conifer forest located approximately 80 km east of Fresno, CA in the Sierra Nevada. The elevation ranges from 1900-2600 m, with an average annual precipitation of 125 cm, falling almost exclusively as snow in this Mediterranean climate (North et al., 2002). Dominant species in the mixed-conifer forest include Abies concolor, A. magnifica, Calocedrus decurrens, Pinus jeffreyi, and P. lambertiana. The Teakettle experiment, located in the Experimental Forest, was established to examine the ecological effects of a range of structural manipulations and prescribed burning. The experiment utilized a full factorial design, crossing three levels of thinning (no thin, understory-thin, overstory-thin) with two levels of burning (no burn, prescribed fire). Three replicates of each treatment were established using four-hectare treatment units. Within each treatment unit, 9-49 monumented grid points were established for sampling understory vegetation and surface fuels. The understory-thin treatment removed all trees 25-75 cm diameter at breast height (DBH). The understory-thin was initially designed to reduce impacts on California spotted owl (Strix occidentalis occidentalis) habitat, although the guidelines now have been primarily used to reduce stand-replacing fire risk in Sierra mixed-
conifer forests. The overstory treatment removed all trees greater than 25 cm DBH, with the exception of 22 large diameter trees ha\(^{-1}\). The thin and burn plots were mechanically treated in 2000 and burned in 2001. The thin-only plots were treated in 2001. Prescribed fires were implemented during fall 2001 (North et al., 2002).

**Data Collection**

Prior to treatment, all trees and standing snags \(\geq 5\) cm DBH were measured, mapped (using a surveyor’s total station) and permanently tagged. The following measurements described in the subsequent paragraphs were repeated immediately following treatment in 2002 and again in 2011 using the same protocols with one exception; during the 2011 re-measurement, logs \(\geq 30\) cm were not remapped and measured. Instead, all coarse woody debris (CWD) was measured using the planar-intercept method (Brown 1974). Three 15 m fuel transects were measured at each gridpoint within each treatment unit. A detailed reporting of the pre- and post-treatment methods can be found in North et al. (2009a), Wayman and North (2007), and Innes et al. (2006).

At the monumented grid points, understory vegetation, fuels, soil C, and fine roots were measured. Understory plants (herbs and shrubs) were sampled using a 10 m\(^2\) circular plot. Fuels were measured using the planar intercept method (Brown 1974) with modifications and fine woody debris (FWD), CWD, litter (Oi soil layer), and duff (Oe soil layer) were sampled on three transects at nine grid points within each treatment unit. Logs with a diameter \(\geq 30\) cm and \(\geq 2\) m length were also mapped with the total station and the mapped coordinates were used to calculate log length. The volume of each log was estimated as a frustrum paraboloid (Husch et al., 1993). Mass (Mg ha\(^{-1}\)) was estimated using genus-specific gravities of Harmon et
by decay class for the dominant species found at Teakettle (Innes et al., 2006). For unidentified CWD, we used average specific gravities for the dominant tree species found at Teakettle by decay class (Harmon et al., 1987). Soil C was measured by collecting three 2 cm diameter cores at each of nine grid points within each treatment unit in 2012. Soil cores were collected at a depth of 0-10 cm and 10-30 cm, air dried to a constant weight, and sieved with a 2mm sieve. Total C analysis was conducted by the ANR analytical lab at the University of California, Davis.

C calculations

We quantified changes in C from the period of 2001 to 2011 from the 10-year post treatment re-measurement (2011 data collection). We used genus-specific allometric equations from Jenkins et al. (2004) to calculate tree and snag biomass. Coarse and fine woody debris biomass were calculated following Brown (1974), assuming C concentration to be 50% of biomass. The C in litter (Oi soil layer) and duff (Oe soil layer) was quantified using a C concentration of 37% (Smith and Heath, 2002). We quantified C in shrubs using a site-specific relationship between percent cover and biomass (Hurteau and North, 2008) and assumed a shrub C concentration of 49% following Campbell et al. (2009). Carbon emissions produced by prescribed burning were obtained from the difference between total pre-treatment live trees, snags, coarse and fine woody debris, and litter values and post-treatment for the same C pools. Additional C losses associated with milling efficiency were calculated by assuming that 60% of each log was converted into lumber, while the remaining 40% was considered a direct emission. Additional sources of C removal and emissions related to treatment were also included and a detailed description can be found in North et al. (2009a).
Data analysis

To draw comparisons across treatments, C values were scaled to per hectare values. Because our hypotheses were based on post-treatment C dynamics, we calculated the percent change in C stock size between the 2011 (10-years post-treatment) and 2002 (immediately post-treatment) measurement periods for each of the measured C pools. We used ANOVA and Tukey’s HSD post hoc analysis to determine if there were significant ($p < 0.05$) differences between treatments. Each variable was evaluated for normality and equal variance; all variables met the ANOVA assumptions.

2.5 Results

Change in total C stock

Post-treatment total C increased in all treatments except the control, with the two largest 10-year C gains occurring in the understory-thin (51.5 Mg C ha$^{-1}$) and burn-only (47.7 Mg C ha$^{-1}$) treatments (Fig. 2.1, Table 2.1). The live tree C gain in the understory-thin (47.3 Mg C ha$^{-1}$) accounted for most of the total stand C gain (Fig. 2.1, Table 2.1). While in the burn only treatments, the live tree C gain (19 Mg C ha$^{-1}$) and snag C gain (20 Mg C ha$^{-1}$) accounted for the majority of total stand C gain (Fig. 2.1, Table 2.1). Over the post-treatment decade total C in the control remained relatively constant, with decreases in live tree C compensated by increases in snag C (Fig. 2.1, Table 2.1). The burn, understory-thin, and understory-thin and burn sequestered more C than was removed and emitted during treatment implementation, resulting in a C surplus of 64, 41, and 4 Mg C ha$^{-1}$, respectively (Fig. 2.1). However, in the understory thin and burn, mortality resulted in a large increase in dead tree C and a small
decrease in live tree C over the 10-year period (Fig. 2.1, Tables 2.1 & 2.2). The overstory-thin, and overstory-thin and burn continued to have a C debt of 6 and 45 Mg C ha\(^{-1}\), respectively (Fig. 2.1).

![Bar graph showing carbon pools](image)

Figure 2.1. Mean and standard error of C pools immediately post-treatment (2002, colored bars) and 10-years post-treatment (2011, colored and hashed bars) in Mg C ha\(^{-1}\). Carbon Balance (solid blue bar) is the 10-year C stock gain minus C removed and emitted during treatment implementation in Mg C ha\(^{-1}\).
Table 2.1 The carbon stock size (Mg C ha\(^{-1}\)) of different pools in 2002 (immediately post-treatment) and 2011 (10-years post-treatment) for each of the six treatments. Standard errors are shown in parentheses.

<table>
<thead>
<tr>
<th></th>
<th>Control</th>
<th>Burn only</th>
<th>Understory-thin</th>
<th>Understory-thin and burn</th>
<th>Overstory-thin</th>
<th>Overstory-thin and burn</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Post 10-yr post</td>
<td>Post 10-yr post</td>
<td>Post 10-yr post</td>
<td>Post 10-yr post</td>
<td>Post 10-yr post</td>
<td>Post 10-yr post</td>
</tr>
<tr>
<td>Live tree</td>
<td>302.4 (40.7)</td>
<td>286.5 (34.4)</td>
<td>240.1 (10.2)</td>
<td>259.0 (14.5)</td>
<td>173.3 (9.35)</td>
<td>169.9 (3.87)</td>
</tr>
<tr>
<td>Snag</td>
<td>20.8 (3.65)</td>
<td>45.6 (10.1)</td>
<td>25.2 (1.26)</td>
<td>34.1 (7.39)</td>
<td>26.4 (11.9)</td>
<td>48.1 (8.36)</td>
</tr>
<tr>
<td>CWD</td>
<td>27.6 (4.81)</td>
<td>15.4 (1.99)</td>
<td>9.4 (0.78)</td>
<td>20.3 (2.85)</td>
<td>20.3 (4.36)</td>
<td>9.0 (1.87)</td>
</tr>
<tr>
<td>FWD</td>
<td>4.17 (0.39)</td>
<td>12.7 (1.45)</td>
<td>4.16 (0.55)</td>
<td>10.2 (0.78)</td>
<td>7.73 (1.04)</td>
<td>5.09 (0.65)</td>
</tr>
<tr>
<td>Litter and duff</td>
<td>7.90 (0.25)</td>
<td>5.20 (0.83)</td>
<td>4.70 (0.15)</td>
<td>4.90 (0.93)</td>
<td>7.40 (0.25)</td>
<td>6.00 (0.69)</td>
</tr>
<tr>
<td>Soil</td>
<td>78.1 (2.91)</td>
<td>75.2 (10.3)</td>
<td>67.6 (3.97)</td>
<td>67.3 (4.71)</td>
<td>103.0 (13.6)</td>
<td>79.3 (1.29)</td>
</tr>
<tr>
<td>Shrub</td>
<td>0.03 (0.01)</td>
<td>0.02 (0.00)</td>
<td>0.02 (0.00)</td>
<td>0.02 (0.00)</td>
<td>0.02 (0.01)</td>
<td>0.03 (0.00)</td>
</tr>
<tr>
<td>Total</td>
<td>441.0 (41.1)</td>
<td>440.7 (38.6)</td>
<td>351.3 (32.9)</td>
<td>398.9 (34.9)</td>
<td>381.6 (28.9)</td>
<td>433.1 (34.1)</td>
</tr>
</tbody>
</table>
Table 2.2 The percent change in carbon in the different carbon pools between 2002 (immediately post-treatment) and 2011 (10-years post-treatment). Values in the same row with different letters are significantly different (p≤0.05).

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Control</th>
<th>Burn only</th>
<th>Understory-thin</th>
<th>Understory-thin and burn</th>
<th>Overstory-thin</th>
<th>Overstory-thin and burn</th>
</tr>
</thead>
<tbody>
<tr>
<td>Live tree</td>
<td>-4.5 \textsuperscript{a}</td>
<td>7.9 \textsuperscript{a}</td>
<td>25.1 \textsuperscript{a}</td>
<td>-7.6 \textsuperscript{a}</td>
<td>20.3 \textsuperscript{a}</td>
<td>-1.8 \textsuperscript{a}</td>
</tr>
<tr>
<td>Snag</td>
<td>118.2 \textsuperscript{a}</td>
<td>101.9 \textsuperscript{a}</td>
<td>61.2 \textsuperscript{a}</td>
<td>82.0 \textsuperscript{a}</td>
<td>66.0 \textsuperscript{a}</td>
<td>69.4 \textsuperscript{a}</td>
</tr>
<tr>
<td>Coarse woody debris</td>
<td>-40.2 \textsuperscript{a}</td>
<td>31.4 \textsuperscript{ab}</td>
<td>5.4 \textsuperscript{ab}</td>
<td>218.3 \textsuperscript{b}</td>
<td>25.1 \textsuperscript{ab}</td>
<td>133.2 \textsuperscript{ab}</td>
</tr>
<tr>
<td>Fine woody debris</td>
<td>204.9 \textsuperscript{a}</td>
<td>160.7 \textsuperscript{a}</td>
<td>120.9 \textsuperscript{a}</td>
<td>54.7 \textsuperscript{a}</td>
<td>33.2 \textsuperscript{a}</td>
<td>91.0 \textsuperscript{a}</td>
</tr>
<tr>
<td>Litter and duff</td>
<td>-34.6 \textsuperscript{b}</td>
<td>5.1 \textsuperscript{ab}</td>
<td>-18.8 \textsuperscript{ab}</td>
<td>10.4 \textsuperscript{ab}</td>
<td>-57.5 \textsuperscript{b}</td>
<td>78.8 \textsuperscript{a}</td>
</tr>
<tr>
<td>Soil</td>
<td>-2.6 \textsuperscript{a}</td>
<td>0.5 \textsuperscript{a}</td>
<td>-20.8 \textsuperscript{a}</td>
<td>4.6 \textsuperscript{a}</td>
<td>-12.5 \textsuperscript{a}</td>
<td>-17.7 \textsuperscript{a}</td>
</tr>
<tr>
<td>Shrub</td>
<td>-52.1 \textsuperscript{a}</td>
<td>-30.0 \textsuperscript{a}</td>
<td>99.1 \textsuperscript{a}</td>
<td>179.5 \textsuperscript{a}</td>
<td>358.9 \textsuperscript{ab}</td>
<td>787.6 \textsuperscript{b}</td>
</tr>
<tr>
<td>Total</td>
<td>0.5 \textsuperscript{a}</td>
<td>13.8 \textsuperscript{a}</td>
<td>15.8 \textsuperscript{a}</td>
<td>10.8 \textsuperscript{a}</td>
<td>11.8 \textsuperscript{a}</td>
<td>13.8 \textsuperscript{a}</td>
</tr>
</tbody>
</table>
Change in C pools

The live tree C pool increased in half of the treatments, while small declines occurred in the other half of the treatments (Table 2.2). The largest increase, 25%, was in the understory-thin treatment, followed by the overstory thin, at 20% (Table 2.2). The control (-4.5%), understory thin and burn (-1.4%), and overstory-thin and burn (-1.8%) had small declines in live tree C (Table 2.2). The snag C pool increased in all treatments. Percent snag C gain in the control was higher than other treatments (118%), though no statistical difference was detected between treatments (Table 2.2). At 82% C gain, the understory-thin and burn had an approximately 20% larger mean increase in snag C than the understory-thin without prescribed burning. However, the difference between snag C in both overstory treatments was only 3% with prescribed burning (Table 2.2).

Treatments that included prescribed fire experienced a greater C gain in CWD than treatments that did not include prescribed fire. This was likely due to CWD addition by fire-induced tree mortality. At a deficit of 40%, the control was the only treatment that lost C from the CWD pool. Treatments that combined thinning and burning had much larger increases in CWD than burn-only and thin-only treatments (Tables 2.1 & 2.2). CWD C gain in the understory-thin and burn treatment (218%) was significantly greater than the CWD C loss in the control (Table 2.1). The FWD pool had the largest C increase in the control (205%), followed by the burn-only (161%) and understory-thin (121%) treatments (Table 2.2). FWD percent C gain was not significantly different across treatments, due to within treatment unit variability (Table 2.2). With the exception of the overstory-thin and burn treatment having a significantly greater percent C increase in litter and duff than the control and overstory-thin, no statistically
significant differences were observed between treatments (Table 2.2). However, litter and duff C increased in all burn treatments, and decreased in all treatments that were not burned (Table 2.2). The increase in burn treatments is likely due to combined inputs to the litter layer from scorched and fire-killed trees and litter inputs from live trees and understory shrubs over several years following treatment.

Shrub C had the largest increase in treatments that included overstory-thinning, likely due to increased light availability (Tables 2.1 & 2.2). The overstory-thin and burn treatment experienced a 7-fold C increase, significantly greater than any other treatment, with the exception of the overstory-thin treatment (Tables 2.1 & 2.2). Shrub C decreased in the control and burn-only treatments, possibly because of continued in-growths several years following treatment (Table 2.1). Although percent C gain in the shrub pool was greater than other pools for the overstory-thin and overstory-thin and burn, shrub C did not significantly affect total ecosystem C storage since it represents on average less than 0.0003 Mg C ha$^{-1}$ of total forest C (Fig. 2.1, Table 2.2). Soil carbon did not significantly vary between 2002 and 2012 (Tables 2.1 & 2.2).

**Live tree distribution of C**

Ten years following treatment implementation, live trees greater than 130 cm DBH continued to account for the greatest amount of live tree C across all treatments (Fig. 2.2). Live tree C in the largest diameter class increased over the 10-year post-treatment period in the burn-only and overstory-thin treatments (Fig. 2.2). Live tree C in this diameter class also increased significantly in the understory-thin (p<0.05, Student’s paired t-test) (Fig. 2.2). Although not significantly different, the control, understory-thin and burn, and overstory-thin
and burn all had decreases in total live tree C in trees greater than 130 cm DBH (Fig. 2.2). The decrease in total live tree C is likely associated with reduced C stored in trees greater than 130 cm DBH (Table 2.2, Fig. 2.2).

*Abies concolor* accounted for approximately 67% of the basal area prior to treatment (Meyer *et al.*, 2007). Following treatment, in both measurement periods, *A. concolor* accounted for the largest fraction of live tree C (Fig. 2.2). In the combined thin and burn treatments, fire-induced mortality had a similar effect for both fire-intolerant and fire-tolerant species (Fig. 2.2). Percent C gain for *A. concolor* was greatest in the understory-thin (28%) and overstory-thin treatment (26%) (Fig. 2.2).

We hypothesized that growth of fire-tolerant species (e.g. *Pinus* spp.) would have a greater increase than fire-intolerant species (e.g. *Abies* spp. and *C. decurrens*) in treatments that included burning. Our results did not show statistically significant differences in C gain between fire-tolerant and intolerant species (Fig 2.2).
Figure 2.2. Distribution of C among the dominant tree species at Teakettle Experimental Forest, distributed in 20 cm DBH classes for (a) the control and (b-f) for the five treatments. The paired bars in each DBH class present data immediately post treatment (2002) and 10-years post-treatment (hashed bars), respectively. The y-axis scale varies by treatment.
2.6 Discussion

Treating forests to reduce the risk of high-severity wildfire not only requires an immediate reduction in forest C stocks, but also produces emissions from thinning and prescribed burning (Finkral and Evans, 2008; Hurteau et al., 2008; Campbell et al., 2009; North et al., 2009a). By employing simulation methods, Campbell et al. (2012) suggested that treatment application is burdened with a large C debt. Conversely, we found that certain low-intensity treatments (burn-only, understory-thin, and understory-thin and burn) allow the amount of C emitted and removed during treatments to be re-sequestered within 10-years (Fig. 2.1). This 10-year C recovery period is well within the site-specific mean fire return interval (17.3 years, (North et al., 2005)); demonstrating the time required to re-sequester C lost to treatment is within the mean frequency of the historical fire regime (Fig. 2.1). However, we evaluated C debt recovery in terms of direct treatment emissions (prescribed fire, equipment, and hauling emissions) and indirect (milling waste) losses. Excluded from the C debt recovery calculation is the C stored in long-lived wood products, which ranged from 27.5-56.2 Mg C ha\(^{-1}\) (North et al. 2009a) and remains sequestered for a considerable period of time (Skog and Nicholson, 1998). Recovering the C stored in lumber through forest regrowth while maintaining a forest structure that is resistant to high-severity wildfire will likely take a considerably longer period of time as it requires additional growth in the retained trees, rather than in-growth from regeneration.

Some of the variability in C pool size between post-treatment measurement periods was unexpected. Our results show a reduction in coarse woody debris (CWD) in the control over the 10-year post-treatment measurement period. This reduction may be due to decomposition
of CWD or may be an artifact of the change in sampling methodology from 2002 to the 2011 measurement period. In this latter sampling period we used fuel transects to quantify CWD, whereas North et al. (2009a) used a full CWD inventory. Changes in the amount of C contained in shrubs ranged from a 52% reduction in the control to a 788% increase in the overstory-thin and burn. While these relative changes are large, the absolute C pool size for shrubs is quite small in relation to total ecosystem C (Tables 2.1 & 2.2, Fig. 2.1). The changes in shrub cover likely have much larger ecological effects than C balance effects (Hurteau and North, 2008; Hurteau and North, 2010). Differences in the soil C stock between sampling periods are likely explained by the variability that is inherent in soils, coupled with the sampling intensity employed in both measurement periods. More intensive sampling for soil C would likely provide a more accurate representation of treatment effects. Furthermore, black C (formed from the incomplete combustion of biomass) is a recalcitrant form of C storage that was not measured in this study. DeLuca and Aplet (2008) estimated that even small black C inputs from a single fire event (i.e. 0.17-1.7 Mg C ha$^{-1}$) could contribute substantially to total ecosystem C storage because of its relatively stable nature. The effect fuel reduction treatments have on soil C and black C production warrants further investigation.

Prior to treatment, aboveground live tree C in the six treatments ranged from 188.0 – 249.8 (±3.82) Mg C ha$^{-1}$ (North et al. 2009a). The overall mean of this pre-treatment C pool was similar to that of the 2011 aboveground portion of the live tree C in the control (236.5 Mg C ha$^{-1}$). The live tree C decline in the control highlights the difficulty in quantifying absolute change in carbon stocks against a pre-treatment baseline, because without treatment the control results suggest that live tree C stocks would have been variable over the post-treatment
decade. In this study we have focused on comparing post-treatment changes. The burn-only, understory-thin, and understory-thin and burn treatments were the only treatments that recovered the initial C debt. In the burn-only and understory thin and burn dead tree C (snag, CWD) increased more than live tree C (Table 2.1, Fig. 2.1). In the understory-thin and burn, the increase in snag C is largely due to the higher mortality in trees > 130 cm diameter (Fig. 2.2d, Fig. 2.1). This finding is especially relevant today, as thinning from below, removing mid-size trees, followed by prescribed burning has become a commonly recommended and implemented management practice for its effectiveness in treating both ladder and surface fuels (Raymond and Peterson, 2005; Stephens and Moghaddas, 2005; Prichard and Kennedy, 2012). The high mortality of large trees following treatment may have resulted from long-term litter build-up at the base of the tree putting these trees at risk of cambial and root injury from smoldering combustion (Swezy and Agee, 1991; Fule et al., 2002; Stephens and Finney, 2002). In the understory-thin and burn, large (> 50cm DBH) fire-intolerant species experienced, on average, a 6.4% reduction, while fire-tolerant species, on average, experienced a 28% reduction in C (Fig. 2.2). Declines in live tree C for both species groups were driven by C reduction in the largest diameter class (> 130 cm DBH) (Fig. 2.2). van Mantgem et al. (2013) found that mortality rates were similar for large (> 50 cm DBH) Pinus (4.6% yr^{-1}) and Abies (4.0% yr^{-1}) tree species 5-years following prescribed fires from 1984-2004. Additionally, evidence has shown that background mortality rates throughout the entire western United States have increased in recent decades, with regional warming and drought stress being the most likely drivers (van Mantgem et al., 2009; van Mantgem et al., 2013). Our results show a similar tendency with an increase of 118% snag C in the control as an example (Table 2.1).
To buffer against mortality associated with combined thinning and burning treatments, we recommend retaining trees that can serve as replacements for large individuals lost to treatment-induced mortality. As an example, the understory-thin and burn treatment removed all trees 50-75 cm DBH. Retaining several of these larger individuals per hectare would help ensure that any large tree mortality following treatment is compensated more quickly than would occur by relying on a 49 cm DBH individual to double in diameter. Additionally, post-treatment competitive release may accelerate growth of retained midsized individuals into larger diameter classes. Furthermore, when selecting mid-sized individuals for retention, the basal area distribution can be moved further toward fire-resistant pine species by retaining *P. jeffreyi* and *P. lambertiana*, similar to findings of the Fire and Fire Surrogate study (Schwilk *et al.*, 2009).

Although additional C costs are incurred when prescribed burning follows thinning, thinning without burning fails to reduce surface fuel accumulation and has the potential to produce relatively high wildfire emissions and not meet society’s objective to protect lives and property when ignition does occur (Stephens *et al.*, 2012). Furthermore, neglecting to restore fire does not promote the ecological benefits associated with low-severity fire in this system (North *et al.*, 2009b; Stephens *et al.*, 2013). While forests do sequester and store C, stabilizing C in frequent fire forests comes with a C cost in the form of periodic emissions from a restored fire regime to avoid high-severity wildfire risk (Hurteau and Brooks, 2011; Hurteau *et al.*, 2013b). Low-intensity fires have the advantage of reintroducing critical ecological processes (Hurteau *et al.*, 2013a). For instance, prescribed fire that emulates historic conditions (low to moderate-severity surface fires) prepares seedbeds for germination of conifer species, recycles
nutrients, and increases soil water availability, all factors that potentially increase C fixation and creates a mosaic of open and closed habitat for threatened owl species (Kilgore, 1973; Sala et al., 2005; Roberts et al., 2011). In a mixed-conifer forest Wayman and North (2007) also found that forests that are not thinned and burned do not sufficiently modify the canopy to increase understory plant cover, suggesting that a combination of thinning and burning are necessary to increase cover of understory vegetation.

Retaining large, fire resistant trees in a more open structure, coupled with the heterogeneity created by prescribed fire (Stephens et al., 2008; North et al., 2009b), will promote ecosystem characteristics that have been identified as important for building system-level resilience for both future wildfire and changing climate (Westerling et al., 2006; Millar et al., 2007; Moritz et al., 2013; Stephens et al., 2013). Shade-tolerant, fire-intolerant species (A. concolor, A. magnifica, C. deccurrens) not only stimulate dense patches of high fire risk, but are also more sensitive to precipitation fluctuations (Hurteau et al., 2007; North et al., 2009b; Collins and Stephens, 2010; Earles et al., In press), suggesting that, as climate becomes more variable, these fire-intolerant species may detract from ecosystem resilience. However, shade tolerant, fire-intolerant species account for a large fraction of live tree C, suggesting that retention of large (≥ 90 cm DBH) individuals of these species can continue to contribute to total C stock. Large fire-tolerant pine species also contribute to post-treatment C gain. The percent C gain of large pines (≥ 90 cm DBH) was greatest in the understory-thin for P. jeffreyi (Fig. 2.2c) and burn-only treatment for P. lambertiana (Fig. 2.2b). However, Zald et al. (2008) found that regeneration of these pine species only occurred in the most intensive treatments. From a climate change resilience perspective, large tree retention in a more open structure may serve
to facilitate reduced water stress during periods of drought. Recent research found that large trees in thinning and burning treatments in a southwestern ponderosa pine forest were less affected by subsequent drought than were smaller individuals of the same species (Kerhoulas et al., 2013).

Post-treatment C dynamics should be considered in the context of treatment effectiveness and longevity in maintaining reduced high-severity wildfire risk. The efficacy of treatments varies as a function treatment intensity, forest type, and site productivity (Stephens et al., 2012). In the Sierra Nevada, treatment longevity ranges from 5-20 years (Stephens et al., 2009; Chiono et al., 2012). Given that fire is a self-limiting process in these fuel limited systems (Collins et al., 2009), repeated burning should help maintain the forest structure and fuels distribution resulting from treatment. One additional consideration regarding post-treatment C dynamics and treatment effectiveness is the projected increase in large wildfire frequency with changing climate (Westerling et al., 2011). Temperature and moisture changes are already causing an increase in the frequency of high and extreme fire weather occurrence (Collins, 2014). If this trend continues, more intensive treatments with higher C costs may be required to maintain treatment efficacy.

In dry, fire-prone western forests, the potential for stand-replacing wildfire is of great concern, not only for its effects on C storage but also for the conservation of mixed-conifer forests. Implementing practices that reduce the risk of wildfire come with initial C costs through direct emissions (prescribed burning) and removal (thinning). However, these C costs are recovered within the mean fire return interval in treatments that do not remove large diameter trees. Initial C costs are outweighed by the benefits that follow fuel reduction
treatments, including reduced stand-replacing wildfire risk and the C emitted from potential severe fire events (Hurteau and North 2009, Hurteau et al., 2013b). Selective thin-from-below followed by prescribed burning may most effectively stabilize C and restore ecological resilience to these forests. The retention of additional midsized trees to buffer against treatment induced mortality and C loss from larger individuals may provide an opportunity to meet wildfire risk mitigation objectives while reducing losses of live tree C. While there are C costs associated with prescribed burning, it is necessary to restore this natural process following thinning to reduce surface fuel loads and maintain ecological processes and ecological resilience in this forest system. Retaining large diameter trees and selectively removing midsize, fire-intolerant tree species (*A. concolor, A. magnifica, C. decurrens*) can contribute a forest ecosystem that is more resilient to changing climate and resistant to stand-replacing wildfires.

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Chapter 3: Formation of charcoal carbon from the incomplete combustion of fuels following prescribed burning in a mixed-conifer forest, Sierra Nevada, California

3.1 Abstract

Fire suppression, increasing temperature, and prolonged drought have resulted in increased wildfire frequency and severity in the western United States. Large and severe wildfires impact the carbon cycle both through direct emissions and reduced sequestration resulting from tree mortality. Forest thinning and prescribed burning reduce high-severity fire risk, but require removal and emissions of carbon. However, during each fire event not all biomass is emitted to the atmosphere. A fraction of the burning vegetation and soil organic matter is converted into charcoal, a stable carbon form. We hypothesized that charcoal carbon deposition from combusted coarse woody debris during prescribed burning would be greater than charcoal carbon deposition from non-combusted fine woody debris. We quantified post-treatment charcoal carbon formation in organic matter and the first 5 cm of the mineral soil from downed logs (> 30 cm diameter) that were present prior to treatment. Our results indicate that there was no difference in the amount of charcoal carbon produced from combusted coarse woody debris and combusted fine woody debris. We also compared treatment effects on charcoal production and found that the burn-only, understory-thin and burn, and overstory-thin and burn treatments had significantly more charcoal carbon than the control. Charcoal carbon represented 0.19% of total ecosystem carbon, a relatively small fraction of total ecosystem carbon. However, charcoal carbon is long-lived and will likely continue to accumulate with repeated burning, leading to additional increases in long-term soil carbon storage. Given increasing efforts to reduce high-severity wildfire risk with thinning and
burning, our results help improve our understanding of the effects these treatments have on ecosystem carbon flux by providing additional information of charcoal carbon formation.

3.2 Introduction

Anthropogenic climate change is projected to result in regional warming and drying in the western United States (Seager et al., 2007; Solomon et al., 2009; Seager and Vecchi, 2010). These changes in climate are projected to increase the frequency of large wildfires across the region (Pechony and Shindell, 2010; Westerling et al., 2011). Additionally, 20th century fire suppression has increased tree density and the accumulation of forest floor biomass, making forests that evolved with a high-frequency, low-severity fire regime more prone to high-severity, stand-replacing fire events (Stephens and Ruth, 2005; Scholl and Taylor, 2010).

Fire emissions from these wildfire events represent a large and highly variable component of the United States carbon (C) budget (Wiedinmyer and Neff, 2007). Studies have shown that forests in the western United States can remain a net source of C for years to decades following a stand replacing wildfire event (Dore et al., 2008; Meigs et al., 2009). Since forests are valued globally for mitigating atmospheric carbon dioxide, concern about climate change has made mitigation of greenhouse gas emissions and increased C storage a priority for forest managers (Pan et al., 2009).

Altering the forest structure and decreasing fuel loads through mechanical thinning and prescribed burning can increase forest resistance to stand-replacing wildfires (Agee and Skinner, 2005). By reducing the risk of stand-replacing wildfires forest managers may also be protecting against potential C losses, through reduced direct wildfire C emissions and increased resistance to biome switching following the wildfire event (e.g. grasslands or shrublands).
(Hurteau et al., 2008; Hurteau and Brooks, 2011). This is assuming that the C losses associated with the management practices are less than the avoided C losses associated with the wildfire or alternate vegetation state (Hurteau et al., 2008; Hurteau and Brooks, 2011).

Understanding the influence that prescribed fire has on C accumulation rates is essential for quantifying total ecosystem C flux to the atmosphere, and thus accurate C accounting. Various studies have investigated forest C losses from fire restoration management (Kaye et al., 2005; Finkral and Evans, 2008; Hurteau and North, 2009; Stephens et al., 2009; Hurteau and Brooks, 2011; Mitchell et al., 2009), but few studies have investigated the potential long-term C storage resulting from prescribed fire. Fire contributes to long-term C storage by producing highly recalcitrant, pyrogenic charcoal, a C sink that has often been overlooked by fire scientists (DeLuca and Aplet, 2008; Donato et al., 2009).

Pyrogenic charcoal is formed from the incomplete combustion of biomass, in the absence of oxygen. It is represented by a continuum of forms ranging from partly charred plant material to soot and graphite particles (Schmidt and Noack, 2000). Charcoal is C-rich and nitrogen-depleted, and has a highly aromatic molecular structure that contributes to increased resistance to microbial degradation, making charcoal a stable soil C component (Schmidt and Noack, 2000). Charcoal in forest soils has been documented to date more than 10,000 years back in time (Baldock and Smernik, 2002; Preston and Schmidt, 2006). This characteristically long mean residence time makes charcoal a significant contributor of global C sequestration in soils (Forbes et al., 2006; Lehmann et al., 2008; Preston, 2009).

Little direct field evidence has been collected to quantify soil charcoal content in forests that experience low-severity fire, such as prescribed burning. DeLuca and Aplet (2008)
suggested that fuel reduction treatments that do not include prescribed burning may reduce soil charcoal content and thus, long-term C storage in mineral soils. A study in a Florida scrub-oak forest, found that a quarter of litter-fall was converted into charcoal following prescribed burning (Alexis et al., 2006). In conifer forests located in the western US, potential combustible debris on the ground in the form of coarse woody debris (CWD) may provide a source for charcoal formation. In Sierra Nevada mixed-conifer forests CWD represents 3-8% of total ecosystem C, storing 12-27 Mg C ha\(^{-1}\) (Hurteau and North, 2010; Wiechmann et al., In Review). Without directly quantifying potential charcoal C additions from combusted CWD, there may be an unaccounted long-term C storage pool. Furthermore, even if charcoal formation rates are relatively low, the long residence time of charcoal in soil may still have a substantial effect on estimated C flux from prescribed fire (DeLuca and Aplet, 2008).

### 3.3 Objectives

The purpose of this study was to quantify the effects of pre-fire fuel load and fuel type on charcoal formation resulting from different fuel reduction treatments that included prescribed burning in a mixed-conifer forest. We tested the following hypotheses: (1) charcoal C formation is greater from the combustion of CWD than when combustion of smaller fuels occurs, (2) all forest stands that were burned would contain more charcoal C than forest stands that were not burned, and (3) CWD in treatments that were thinned and burned would produce more charcoal C than treatments that were only burned. To improve estimates of C contained in charcoal resulting from prescribed fire, we developed linear regression models that could be used to predict charcoal C formation from combusted CWD. Lastly, using our empirical results we expanded on the forest C sequence provided by DeLuca and Aplet (2008) to make century
scale estimates of potential charcoal C storage in a Sierran mixed-conifer forest that was either restored to the historical fire regime or the fire regime was not restored and experienced two wildfires.

### 3.4 Materials and Methods

#### Study Site

This study was conducted in an old-growth mixed-conifer forest within the Teakettle Experimental forest, a 1300 ha reserve located approximately 80 km east of Fresno, CA in the Sierra Nevada. The climate is characterized as Mediterranean, with average annual precipitation of 125 cm, falling almost exclusively as snow (North et al., 2002). The elevation ranges from 1900-2600 m. This mixed-conifer forest ecosystem is dominated by five tree species: *Abies concolor, A. magnifica, Calocedrus decurrens, Pinus jeffreyi,* and *P. lambertiana.*

The Teakettle experiment, located in the Experimental Forest, was established to examine the ecological effects of a range of structural manipulations and prescribed burning. The experiment utilized a full factorial design, crossing three levels of thinning (no thin, understory-thin, overstory-thin) with two levels of burning (no burn, prescribed fire). Three replicates of each treatment were established using four-hectare treatment units. The understory-thin treatment removed all trees 25-75 cm diameter at breast height (DBH). The understory-thin was initially designed to reduce impacts on California spotted owl (*Strix occidentalis occidentalis*) habitat, although the guidelines now have been primarily used to reduce stand-replacing fire risk in Sierra mixed-conifer forests. The overstory treatment removed all trees greater than 25 cm DBH, with the exception of 22 large diameter trees ha\(^{-1}\). The thin and burn
plots were mechanically treated in 2000 and burned in 2001. The thin-only plots were treated in 2001. Prescribed fires were implemented during fall 2001 (North et al., 2002). Within each treatment unit, 9-49 sub-plots were established for sampling understory vegetation and surface fuels.

**Data Collection**

Prior to treatment implementation, all trees were mapped, tagged, and measured. Permanent 10 m² circular plots were established at either 9 or 49 (intensive) points within the four-hectare treatments units. In each subplot, vegetation, woody debris, and litter depth were measured (North et al., 2002). Using a modified version of the Brown’s planar intercept method fine woody debris (FWD) was quantified at the hectare level (Brown, 1974). Following treatment implementation, the experimental units were re-sampled following the same protocol. Additional measurements within the circular plots of the burn treatments included visually estimating percent ash and percent char on all coarse woody debris (CWD) falling within the circular plots (Wayman and North, 2007). Following treatment implementation, all CWD was mapped and measured (Innes et al., 2006).

**Field Sampling**

We focused our sampling efforts on charcoal formation from CWD (logs ≥ 30 cm diameter) combusted during the 2001 prescribed fire. Sampled logs were chosen from a pre-existing database that included logs that were mapped prior to the prescribed burn treatment. We used this spatially explicit database of log measurements to identify individual logs that were within 15 m of the monumented grid points within treatments that included fire. From
the candidate population, we randomly selected 100 individual logs (10 logs from each treatment unit that included burning and 10 from the control). We collected seven soil cores at each log that encompassed organic matter and the top 5 cm of mineral soil. We constrained the mineral soil depth to 5 cm in an effort to limit sampling to charcoal particles formed during the 2001 prescribed fire. Organic matter was removed and separated prior to collecting the mineral soil. Three cores were taken directly adjacent to the log using 50 cm spacing on the down slope side of the log. Four additional cores were collected using a transect that ran perpendicular to the log from the mid-point. Off-log cores were spaced at 5 cm, 15 cm, 30 cm, and 60 cm down slope of the log. Soil cores were collected using a 10.16 cm diameter metal core. In addition to soil core samples, we sampled char depth at three locations on each log following the methodology of Donato et al. (2009). We used these measurements to quantify the amount of char mass on the log (Donato et al., 2009).

**Lab Analysis**

We used 1 mm and 2 mm sieves to isolate charcoal macro-particles and collected 1 mm and 2 mm charcoal pieces from 536 of the subsamples to obtain the C contribution of each size class. Because the 2 mm size fraction captured the majority of charcoal C (Table 3.1), we focused the remainder of our sampling efforts exclusively on charcoal particles > 2 mm. In addition, Nocentini et al. (2010) found that over half of all C in charcoal was contained in pieces > 2 mm in size. Following methods similar to Brimmer (2006) and Alexis et al. (2006), charcoal separation was performed on white trays under supplemental light using the unaided eye for the 2 mm size fraction and magnifying lamp with an enlargement factor of 175% for the 1 mm size fraction.
After charcoal was picked from the organic matter or soil, the charcoal samples were dried at 65°C, weighed and ground. We used the EA 1110 CHNS-O (CE Instruments 1996) elemental analyzer to obtain the percent carbon and nitrogen in all samples. The CHNS(O) Analyzer determines the percentages of C, H, N, S & O of organic compounds, based on the principle of "Dumas method" which involves the complete and instantaneous oxidation of the sample by "flash combustion" (Matejovic, 1996). We further analyzed for percent hydrogen in the mineral soil charcoal samples using the same elemental analyzer. Oxygen content in the mineral soil was calculated by the difference (Nocentini et al., 2010).

Data analysis

To test the hypothesis that more charcoal is produced from CWD than FWD, we used a linear regression analysis to compare the charcoal weight (g) of samples collected at the different distances. Charcoal in organic matter and charcoal in mineral soil were analyzed separately. We detected no significant differences between charcoal mass (g) collected adjacent to or at varying distances from the logs within each treatment unit. This finding was consistent for both organic matter and mineral soil. Since there was no effect of distance from log, all samples were used to determine charcoal C production at individual logs.

A nested analysis of variance (ANOVA) was used to compare treatment effects on charcoal C production at logs. Charcoal C samples from each log were averaged and converted to an areal scale. C concentrations were scaled to an areal basis (g m⁻²) using the volume and depth of the soil corer, data on log length, the perpendicular transect length (60 cm) and previously derived bulk density (BD) measurements at Teakettle experimental forest (mineral BD: 0.95 g cm⁻³, organic matter: 0.1 g cm⁻³). Cores were treated as subsamples with the
sampling unit being the log (n=30 for treatments that burned, n=10 for un-burned treatments).

Data were log-transformed to meet all assumptions for the ANOVA. We used Tukey’s HSD post hoc analysis to determine if there were significant ($p < 0.05$) differences between treatments.

To estimate charcoal production in each treatment unit we obtained information on log length for all logs in the treatment units using the pre-treatment log dataset. Using measured values for each log within each treatment unit, we estimated charcoal C produced at mapped logs in each treatment unit to obtain an estimate of total C production in each 4 hectare treatment unit (Mg charcoal C ha$^{-1}$). We again used ANOVA to draw comparisons of charcoal C production among the different treatments at the hectare scale. Based on this estimate we were able to compare charcoal C amount with different C pools (e.g. total soil, live tree, etc) on a per hectare basis. Again, data were log-transformed to meet the ANOVA assumptions of normality and equal variance and Tukey’s HSD post hoc analysis was used to determine if there were significant ($p < 0.05$) differences between treatments.

To quantify the effect of prescribed fire on charcoal formation, we used linear regression models in an information theoretic framework (Burnham and Anderson, 2002). The variables included in the analysis were pre-treatment fine woody debris (Mg C ha$^{-1}$); pre-treatment litter depth (cm), 2002 percent coarse woody debris charred, 2002 percent ash cover, and 2013 char mass on sampled CWD (Mg C). From these predictor variables we developed a candidate set of models that included charcoal C (g C m$^2$) as the response variable. We ranked all possible model combinations for each treatment using the Akaike Information Criterion for small samples (AIC$_C$) (Burnham and Anderson, 2002). We chose the best model (lowest AIC$_C$ value) to estimate charcoal C as a function of the fire effect predictors for each
treatment (burn-only, understory-thin and burn, and overstory-thin and burn). Data were log-transformed to meet the assumptions of normality and equal variance.

200-year forest C sequence

Following DeLuca and Aplet (2008) we created a hypothetical scenario to estimate the accumulation of aboveground biomass (live trees and snags), CWD, charcoal C, and fire emissions extended 200 years into the future. Using previous data collected at the Teakettle experimental forest, the empirical results of this study, and additional assumptions gathered from the literature, we projected forest C stocks in an old-growth mixed-conifer forest under two hypothetical scenarios where the forest was thinned from below, then subject to (1) prescribed fire, followed by 9 additional prescribed burns over a 200-year period, or (2) no prescribed fire, followed by two high-severity fires over a 200-year period.

In the first forest C sequence (understory thin and burn followed by 9 prescribed fires at 20-year intervals) aboveground biomass was calculated using site-specific growth rates (Mg C ha\(^{-1}\) yr\(^{-1}\)) for the 10-years following the initial understory thin and burn and the 9 repeated prescribed fires. Wiechmann et al. (In Review) quantified a 10-year growth release of 1.8 Mg C ha\(^{-1}\) yr\(^{-1}\) following the initial thin and burn and an aboveground biomass accumulation rate of 3.9 Mg C ha\(^{-1}\) yr\(^{-1}\). For this scenario we assumed the accumulation rate reported by Wiechmann et al. (In Review) for the 10-years following each prescribed burn over the 200 year sequence. During the second decade following each burn, aboveground biomass was assumed to accumulate at 0.5 Mg C ha\(^{-1}\) yr\(^{-1}\) (Houghton et al., 2000; Law et al., 2001; Hicke et al., 2006). CWD was assumed to increase 1.95 Mg C ha\(^{-1}\) yr\(^{-1}\) during the 10-years following the initial thin and burn, and 0.27 Mg C ha\(^{-1}\) yr\(^{-1}\) during the second decade after each repeated burn based on
the thin and burn and burn only treatment results presented by Wiechmann et al. (In Review).

Annually, 1.55% of CWD was assumed to decompose during years that did not follow treatments (Harmon et al., 1987). Prescribed fires were assumed to consume 6.8% of the aboveground biomass, and 6.1% of CWD (North et al., 2009). Charcoal C content was 0.66 Mg C ha\(^{-1}\) following the initial understory-thin and burn treatment (Table 3, understory-thin and burn). Additional charcoal C inputs from the repeated prescribed fire were assumed to be 0.68 Mg C ha\(^{-1}\) per fire (Table 3, burn-only). We assumed that half of the charcoal was consumed in each subsequent fire following the initial understory-thin and burn following DeLuca and Aplet (2008).

In the second sequence (understory-thin only, followed by two high-severity wildfires), we projected aboveground biomass from site-specific simulations from Hurteau and North (2009). CWD C was projected from results of Wiechmann et al. (In Review). Values and assumptions from the literature and empirical results from this study were used to predict charcoal C. Hurteau and North (2009) simulated one mid-century wildfire occurring over a 100 year period after an initial understory-thin treatment; we expanded this to include a second mid-century wildfire. CWD accumulated 0 Mg C ha\(^{-1}\) 10 years following the understory thin treatment (understory thin in Wiechmann et al. (In Review)). Equivalent to the first forest sequence, a background decay rate of 1.55% yr\(^{-1}\) was assumed for CWD (Harmon et al., 1987). Fifty percent of CWD was assumed to be consumed by wildfires, with a 5-fold increase in CWD 10-years following the wildfire event (Dore et al., 2008). Before the wildfire, background charcoal C was assumed to be 0.03 Mg C ha\(^{-1}\) (Table 3, control). Additional charcoal inputs from
the wildfires was assumed to be 2% of CWD per fire (DeLuca and Aplet, 2008), with half of the charcoal being consumed during each fire event (DeLuca and Aplet, 2008).

Understory thin and burn C releases were quantified from the site-specific fire emissions, 23.4 Mg C ha\(^{-1}\) (North et al., 2009). For this study we concentrated on emissions that were biologically related (fire emissions) and did not include C releases from other sources such as milling waste, equipment use, long-lived wood products, and transportation to the mill. The repeated prescribed burn scenerio emitted 14.8 Mg C ha\(^{-1}\) per fire event (North et al., 2009). Wildfire emissions were assumed to be 25.8 Mg C ha\(^{-1}\) per fire, based on fire emissions from the understory thin simulation from Hurteau and North (2009).

3.5 Results

Chemical properties

With the exception of charcoal C concentration in organic matter in the control (15% C), C concentrations were relatively similar (50-60%) between treatments and soil profiles (organic matter or mineral soil) (Table 3.2). H/C was similar across all treatments (Table 3.2). Charcoal C represented a small fraction of total soil organic carbon (SOC), ranging from 0.94-1.48% (Table 3.2).

Table 3.1. Charcoal carbon distribution (and standard error) of two macro-charcoal fraction classes in the mineral soil (n=247) and organic matter (n=287).

<table>
<thead>
<tr>
<th>Soil profile</th>
<th>Fraction</th>
<th>C distribution (% of total)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mineral soil</td>
<td>&gt;2 mm</td>
<td>92 (0.54)</td>
</tr>
<tr>
<td></td>
<td>1-2 mm</td>
<td>8.0 (0.54)</td>
</tr>
<tr>
<td>Organic matter</td>
<td>&gt;2 mm</td>
<td>86 (1.67)</td>
</tr>
<tr>
<td></td>
<td>1-2 mm</td>
<td>14 (1.67)</td>
</tr>
</tbody>
</table>
Table 3.2. Elemental ratios and percent (%) charcoal C of total soil organic carbon (SOC). Values in the parentheses are standard errors of the mean.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>C concentration (%)</th>
<th>Mineral soil charcoal</th>
<th>C/N</th>
<th>H/C</th>
<th>O/C</th>
<th>Charcoal C % SOC</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Organic matter</td>
<td>Mineral soil</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Control</td>
<td>15</td>
<td>54</td>
<td>76.9 (1.6)</td>
<td>0.06 (0.0)</td>
<td>0.76 (0.0)</td>
<td>1.36 (0.01)</td>
</tr>
<tr>
<td>Burn only</td>
<td>58</td>
<td>60</td>
<td>80.4 (0.6)</td>
<td>0.06 (0.0)</td>
<td>0.63 (0.0)</td>
<td>0.94 (0.01)</td>
</tr>
<tr>
<td>Understory</td>
<td>59</td>
<td>60</td>
<td>73.5 (0.5)</td>
<td>0.06 (0.0)</td>
<td>0.61 (0.0)</td>
<td>1.48 (0.02)</td>
</tr>
<tr>
<td>thin/burn</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Overstory thin/burn</td>
<td>54</td>
<td>60</td>
<td>81.8 (0.8)</td>
<td>0.05 (0.0)</td>
<td>0.66 (0.0)</td>
<td>0.18 (0.01)</td>
</tr>
</tbody>
</table>

CWD and charcoal C production

We hypothesized that charcoal produced from CWD would be greater than charcoal produced from fine fuels; however no significant differences were detected between charcoal C adjacent to the logs and sampled perpendicular to the logs. To test our hypothesis that treatments that were thinned and burned would produce more charcoal than treatments that were only burned we compared the charcoal C from CWD in each treatment. We found that in the burn-only and understory-thin and burn treatments, charcoal C in both mineral soil (10.7 and 10.7 g C m\(^2\), respectively) and organic matter (9.4 and 6.0 g C m\(^2\), respectively) was significantly greater than charcoal C in both the organic matter and soil in the control (organic matter: 0.1 g C m\(^2\), mineral: 2.7 g C m\(^2\)) (Fig. 3.1). Charcoal C in the overstory-thin and burn (organic matter: 1.9 g C m\(^2\), mineral: 5.12 g C m\(^2\)) was not significantly different from the control and charcoal C in the mineral layer was not significantly different from the understory-thin and burn (Fig.3.1).
Figure 3.1 Mean charcoal C (g C m\(^2\)) and standard error for logs within treatments that include prescribed burning (n=30, logs per treatment unit) and the control (n=10, logs per treatment unit), with charcoal in organic matter (red) and mineral soil (blue). Carbon stock comparisons were made across treatments in the same soil layer. Same colored bars with different letters are significantly different (p≤0.05). Charcoal C was log transformed to meet ANOVA assumptions.

**Treatment effect on charcoal C production**

To estimate the impact fuel reduction treatments have on charcoal C formation as a function of CWD availability, we scaled the amount of charcoal formed from the combustion of logs to an areal extent (Fig. 3.2). All treatments that included burning, in both soil layers, had
significantly more C than the control (control, organic matter: 3.7 g C ha\(^{-1}\), mineral soil: 30.3 g C ha\(^{-1}\)) (Fig. 3.2). The only statistical differences detected between treatments that were burned were in organic matter charcoal C, where charcoal C formed in the overstory-thin and burn treatment (93.7 g C ha\(^{-1}\)) was less than the burn only and understory-thin and burn treatments (352.2 g C ha\(^{-1}\), 242.8 g C ha\(^{-1}\), respectively) (Fig. 3.2). Overall, more charcoal C was contained in the mineral soil layer than in the organic matter, suggesting that there is vertical movement of charcoal C down the soil profile, increasing the probability charcoal will remain sequestered and reside on site for a longer duration (Fig. 3.2).
Figure 3.2. Using the pre-fire log dataset, we estimated the mean charcoal C (g C ha⁻¹) produced in each treatment unit that included prescribed burning and the control for charcoal found in organic matter (red) and mineral soil (blue). Carbon stock comparisons were made across treatments for charcoal in the same soil profile. Same colored bars with different letters are significantly different ($p \leq 0.05$) (n=3). Charcoal C was log transformed to meet ANOVA assumptions.

To determine the relative contribution of charcoal to total ecosystem C within treatments, we compared the results from this research to a previous study. Wiechmann et al. (In Review) found that total ecosystem C among treatments ranged from 201-441 Mg C ha⁻¹, with live tree C (78-287 Mg C ha⁻¹) having the largest stock. Of the treatments that included
prescribed burning total ecosystem C ranged from 201-398 Mg C ha\(^{-1}\), with live tree C (78-259 Mg C ha\(^{-1}\)) having the largest stock in burn treatments (Wiechmann et al., In Review). Charcoal contributed 0.40-0.68% of total ecosystem C storage, a relatively small amount (Table 3.3).

Table 3.3. The 2013 charcoal C stock size and the C stock size (Mg C ha\(^{-1}\)) of different pools 10-years post-treatment (2011) for the control and three burn treatments. Values in parentheses are standard errors from mean.

<table>
<thead>
<tr>
<th></th>
<th>Control</th>
<th>Burn only</th>
<th>Understory thin burn</th>
<th>Overstory thin burn</th>
</tr>
</thead>
<tbody>
<tr>
<td>Live tree</td>
<td>286.5 (34.4)</td>
<td>259.1 (14.5)</td>
<td>169.9 (3.87)</td>
<td>78.4 (19.1)</td>
</tr>
<tr>
<td>Snag</td>
<td>45.6 (10.1)</td>
<td>45.2 (1.26)</td>
<td>48.1 (8.36)</td>
<td>28.0 (2.50)</td>
</tr>
<tr>
<td>CWD</td>
<td>15.4 (1.99)</td>
<td>12.1 (4.44)</td>
<td>26.6 (6.35)</td>
<td>17.1 (3.59)</td>
</tr>
<tr>
<td>FWD</td>
<td>12.7 (1.45)</td>
<td>10.2 (0.78)</td>
<td>7.03 (2.79)</td>
<td>8.19 (0.89)</td>
</tr>
<tr>
<td>Litter and duff</td>
<td>5.20 (0.83)</td>
<td>4.90 (0.93)</td>
<td>4.83 (1.47)</td>
<td>2.91 (0.53)</td>
</tr>
<tr>
<td>Soil</td>
<td>75.2 (10.3)</td>
<td>67.3 (4.71)</td>
<td>84.8 (5.80)</td>
<td>66.7 (1.45)</td>
</tr>
<tr>
<td>Shrub</td>
<td>0.02 (0.00)</td>
<td>0.02 (0.00)</td>
<td>0.04 (0.01)</td>
<td>0.07 (0.01)</td>
</tr>
<tr>
<td>Charcoal</td>
<td>0.03 (0.003)</td>
<td>0.68 (0.02)</td>
<td>0.66 (0.03)</td>
<td>0.40 (0.01)</td>
</tr>
<tr>
<td>Total</td>
<td>440.7 (38.6)</td>
<td>398.9 (34.9)</td>
<td>331.3 (21.9)</td>
<td>201.3 (11.9)</td>
</tr>
<tr>
<td>Charcoal % of total</td>
<td>0.007%</td>
<td>0.17%</td>
<td>0.20%</td>
<td>0.20%</td>
</tr>
</tbody>
</table>

Predicting charcoal C

Of the candidate set of predictors, across treatments and soil layers, log char was a very influential predictor of charcoal C production from combusted CWD of the candidate set of predictor variables (Table 3.4 and Table 3.5, Fig. 3.3). In several cases, models with log char as the predictor were ranked as the second best model, based on AIC\(_C\) values (Table 3.4 and Table 3.5). However, in these cases AIC\(_C\) values were within two of the best model, indicating equal support. Log char explained between 18 and 35% of the charcoal C variance in the organic matter layer and 29-49% of the variance in the mineral soil layer (Table 3.4 and Table 3.5).
While charring on logs is a contributing factor to charcoal production, other predictor variables, such as those relating to fire behavior, may improve the ability to estimate charcoal production from prescribed burning.

Table 3.4. Top two models for each treatment with the smallest AIC values for modeling charcoal C (g C m$^{-2}$) production in organic matter. Δ AIC is the difference in AIC values from the model with the lowest AIC.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Predictor(s)</th>
<th>AIC</th>
<th>Δ AIC</th>
<th>R$^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burn only</td>
<td>Fine woody debris (Mg C ha$^{-1}$)</td>
<td>5.50</td>
<td>0</td>
<td>0.136</td>
</tr>
<tr>
<td></td>
<td>Log char (Mg C)</td>
<td>5.55</td>
<td>0.018</td>
<td>0.184</td>
</tr>
<tr>
<td></td>
<td>Fine woody debris (Mg C ha$^{-1}$) + percent ash (%)</td>
<td>9.81</td>
<td>4.30</td>
<td>0.137</td>
</tr>
<tr>
<td>Understory thin and burn</td>
<td>Fine woody debris (Mg C ha$^{-1}$)</td>
<td>5.84</td>
<td>0</td>
<td>0.09</td>
</tr>
<tr>
<td></td>
<td>Log char (Mg C)</td>
<td>6.01</td>
<td>0.176</td>
<td>0.257</td>
</tr>
<tr>
<td>Overstory thin and burn</td>
<td>Log char (Mg C)</td>
<td>6.71</td>
<td>0</td>
<td>0.354</td>
</tr>
<tr>
<td></td>
<td>CWD charred (%) + log char (Mg C)</td>
<td>11.03</td>
<td>4.32</td>
<td>0.365</td>
</tr>
</tbody>
</table>

Table 3.5. Top two models for each treatment with the smallest AIC values for modeling charcoal C (g C m$^{-2}$) production in the first 5cm of the mineral soil. Δ AIC is the difference in AIC values from the model with the lowest AIC.

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Predictor(s)</th>
<th>AIC</th>
<th>Δ AIC</th>
<th>R$^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burn only</td>
<td>CWD charred (%)</td>
<td>6.06</td>
<td>0</td>
<td>0.161</td>
</tr>
<tr>
<td></td>
<td>Log char (Mg C)</td>
<td>6.49</td>
<td>0.04</td>
<td>0.487</td>
</tr>
<tr>
<td>Understory thin and burn</td>
<td>Log char (Mg C)</td>
<td>6.20</td>
<td>0</td>
<td>0.449</td>
</tr>
<tr>
<td></td>
<td>CWD charred (%) + percent ash (%)</td>
<td>10.1</td>
<td>3.89</td>
<td>0.116</td>
</tr>
<tr>
<td>Overstory thin and burn</td>
<td>Percent ash (%)</td>
<td>8.6</td>
<td>0</td>
<td>0.247</td>
</tr>
<tr>
<td></td>
<td>Log char (Mg C)</td>
<td>8.63</td>
<td>0.05</td>
<td>0.289</td>
</tr>
</tbody>
</table>
Figure 3.3 Regression results for charred mass on sampled CWD as a predictor for charcoal carbon. Figures in the left column are results for the organic matter layer. Figures in the right column are results for the mineral soil layer. The first row (a-b) is results for the burn-only, the second row (c-d) is results for the understory-thin and burn, and the third row (e-f) is results for the overstory-thin and burn. Both char mass and charcoal mass were log transformed to meet the assumption of normality.
Implications for forest management and century-scale C storage

We provide a 200-year comparison of future development of two forests to illustrate the long-term potential of charcoal C storage from different restoration practices (Fig. 3.4). In this 200-year hypothetical forest sequence, adapted from DeLuca and Aplet (2008), a forest that was thinned once and experienced two severe wildfires stored less total C (379 Mg C ha\(^{-1}\)) than a forest system that was fire-maintained by prescribed surface fires (437 Mg C ha\(^{-1}\)) (Fig. 3.4). Total C emissions were greater in the fire-maintained system (156 Mg C ha\(^{-1}\)), than the forest stand that was not burned with prescribed fire (52 Mg C ha\(^{-1}\) released) (Fig. 3.4). The repeated prescribed burn sequence accumulated approximately 1.4 times as much live tree and snag biomass (418 Mg C ha\(^{-1}\)) than the wildfire sequence (299 Mg C ha\(^{-1}\)) (Fig. 3.4). C contribution from aboveground biomass (snags and live trees) in the understory thin and burn treatment (418 Mg C ha\(^{-1}\)) greatly increased total C storage (Fig. 3.4). Long-term C storage in the form of charcoal was 2.8 times greater in the forest stand that was restored (3.72 Mg C ha\(^{-1}\)) than in the wildfire sequence (1.34 Mg C ha\(^{-1}\)) (Fig 3.4). There was an additional 2.72 Mg charcoal C ha\(^{-1}\) produced in the restored sequence than in the wildfire forest sequence (Fig. 3.4). At year 2200, charcoal C storage was equivalent to 2.4% of emissions in the fire maintained forest sequence; similarly charcoal C storage was equivalent to 2.6% of total emissions in the stand-replacing fire sequence.
Discussion

Changing climatic conditions and a century of fire exclusion have increased the risk of large stand-replacing wildfire events in Sierran forests, a trend that is projected to continue (Westerling et al., 2006; North et al., 2007; Westerling and Bryant, 2008; Miller et al., 2009).

Figure 3.4. Carbon stocks and emissions from a hypothetical 200-year forest sequences. The two hypothetical sequences: (a) an initial understory-thin followed by repeated prescribed burning at 20-year intervals, and (b) an initial understory-thin followed by two severe wildfires. Note the different y-axis scales below the break.

3.6 Discussion

Changing climatic conditions and a century of fire exclusion have increased the risk of large stand-replacing wildfire events in Sierran forests, a trend that is projected to continue (Westerling et al., 2006; North et al., 2007; Westerling and Bryant, 2008; Miller et al., 2009).
Large wildfire events are releasing substantial amounts of C to the atmosphere and with projected increases in large wildfire frequency, emissions from fire are projected to increase in the most C dense areas of California (Wiedinmyer and Neff, 2007; Hurteau et al., 2014). Previous research has demonstrated that restoring natural fire regimes to frequent-fire forests, such as Sierran mixed-conifer forests, reduces the risk of stand-replacing wildfire, but is not without costs in terms of C emissions to the atmosphere (Wiedinmyer and Hurteau, 2010). However, since charcoal is common in soils worldwide as a result of wildfires and prescribed burning and its chemical properties make it a recalcitrant form of C (Certini, 2005), a full accounting of the effects restoring fire regimes has on C dynamics is incomplete without the quantification of charcoal production from biomass burning. The charcoal collected in this study was found to have a high C/N ratio and low H/C and O/C ratios, providing support that the material is highly stable (Table 3.2) (Pietikäinen et al., 2000; Nocentini et al., 2010). In addition, charcoal C from this system was found in the control treatments that have not experienced fire since 1865 (North et al., 2005).

Our results indicate that prescribed burning is a significant source of charcoal production in this mixed-conifer forest. We had hypothesized that charcoal production would vary as a function of fuel size class, but found no significant difference between CWD and FWD. There are two potential causes of this finding, 1) either there is in fact no difference in charcoal production between the different fuel size classes or 2) the result is an artifact of our sampling design and we did not sample at a great enough distance from the logs to exclude the influence of charcoal produced from CWD. Charcoal pieces that were formed from the combustion of CWD and shed from the sampled log may have been transported down slope of the log either
by gravity or erosion. Given that MacKenzie et al. (2008) found little to no spatial auto-correlation in charcoal content across the forest floor, this finding warrants additional investigation because if the there is no differential effect of fuel size on charcoal formation, our treatment-level estimates of charcoal production are an under-estimate of the actual charcoal produced.

Results by treatment for organic matter and mineral soil show significantly more charcoal in the burn only and understory thin and burn than in the control (Figure 3.1). While charcoal in both the overstory-thin and burn organic matter and mineral soil was not significantly different from the control, when scaled to the treatment level the overstory-thin and burn had a significantly higher amount of charcoal than the control (Figure 3.2). These results demonstrate the effect of pre-fire CWD load on charcoal production. North et al. (2009) reported that CWD in the burn only (24.1 Mg C ha\(^{-1}\)), understory-thin and burn (23.0 Mg C ha\(^{-1}\)), and overstory-thin and burn (18.6 Mg C ha\(^{-1}\)) was reduced by 61, 61, and 58%, respectively as a result of treatment. The post-burn charcoal mass is coincident with pre-burn CWD C mass, demonstrating that pre-burn fuel load is one determinant of charcoal production during burning (Figure 3.2). The other factor, for which we did not collect data that may have influenced charcoal differences between treatments is the effect of fire behavior. Canopy cover can influence microclimatic conditions (Zald et al., 2008), such as relative humidity and temperature, which influence fire behavior. Additionally, FWD fuel loads have been found to drive fire patterns (Fulé et al., 2002; Fule et al., 2004), more FWD in the form of smaller downed branches and debris that had fallen during mechanical thinning may have led to the more active fire behavior, converting more surface biomass into ash or soot, as opposed to
charcoal in soils in the overstory-thin and burn treatment. Similarly, Buma et al. (2014) found that more downed overstory biomass (resulting from wind disturbance) led to higher fire intensity (increased fire temperature and duration), thereby reducing black C formation. Additionally, higher C/N ratios for wood charcoal generally result from burn conditions of higher temperature and duration (Briggs et al., 2012). Although relatively similar, charcoal in the overstory-thin and burn treatment had the highest C/N ratio (Table 2). Data on fire behavior, such as duration, temperature, proportion of smoldering combustion, may provide additional insight into variation in charcoal production as a function of forest overstory structure and fuel loads.

The results of our regression analysis found that log char mass explained 18-35% of the variation in organic matter charcoal and 28-48% of the variation in mineral soil charcoal (Table 3.3 and Table 3.4). Licata and Sanford (2012) reported that the most important influences on soil charcoal C formation are available biomass sources and the fire regime. Lignin-based surface fuels (woody debris) and deep duff layers are two large charcoal C sources that result in relatively higher charcoal C per fire (Licata and Sanford, 2012). Although models that included litter depth were not among the best models to predict charcoal C in this study, pre-fire litter depth was, on average, greatest in the burn only treatments (1.75 cm) followed by the understory-thin and burn (1.3 cm) and overstory-thin and burn litter depths (0.58 cm). Similar to the findings of treatment effects on charcoal production, we think that predictor variables relating to fire behavior may improve the ability to estimate charcoal production from more easily obtained measurements. Fire intensity measurements such as fire-line intensity may help better predict charcoal formation; oxygen availability, and fire duration have also been
cited as possible causes of differences in charcoal production (Carvalho et al., 2011). In addition, fire temperature may also be an important charcoal formation variable, because of its direct effect on charcoal structure and reflectivity (Cerda, 2010). Obtaining a mechanistic understanding of the relationship between biomass amount and type, fire behavior, and charcoal production may require a laboratory-based study that tightly controls variability in these factors.

The charcoal quantities obtained in this study most likely represent a conservative estimate of macroscopic charcoal production from prescribed burning because we confined our sampling to particle sizes greater than 2 mm. However, our subsampling for smaller macro-particles suggest that we captured the majority (86-92%) of the charcoal C produced during the burn treatments (Table 3.1). Also, over time, larger particles may succumb to physical breakdown and eventually contribute to smaller particle size classes.

Overall, more charcoal was found in the top 5 cm of the mineral soil than in the organic matter (Fig. 3.1 and Fig. 3.2). The movement of charcoal vertically down the soil profile could be attributed to leaching of particles carried downward through suspension (lessivage) or bioturbation (reworking of soils and sediments by animals or plants). Movement of charcoal to deeper depths may lead to increased C sequestration over longer time scales, since charcoal would be protected from erosion off site or combustion by subsequent fire events, suggesting that mineral soils reflect an important pool of C influenced by the long-term input of charcoal (Pingree et al., 2012). MacKenzie et al. (2008) found charcoal C content in Sierran mixed-conifer forests to range from 2.0-4.5 Mg C ha⁻¹ in the surface 6 cm of mineral soil, and exhibited a constant charcoal C amount with increased soil depth up to 60 cm. Our study quantified
considerably less charcoal C in the same ecosystem (0.03-0.68 Mg C ha$^{-1}$, Table 3). One possible explanation for the discrepancy in results could be methodological. We used visual separation to quantify charcoal and focused on macro-particles, while MacKenzie et al. (2008) used a chemical extraction method that was not constrained to macro-particles. Additionally, the average C concentration value in this study fell below the range of charcoal C concentration values following wildfires found in the literature (70-80%; (Tinker and Knight, 2000; DeLuca and Aplet, 2008)). In this study (with the exception of the control) all charcoal size fractions had a C concentration of approximately 60%, similar to another prescribed burn experiment (65 % C; Briggs et al. (2012)). The low C concentration (15%) from charcoal pieces in the control organic layer may be the result of possible soil contamination.

While our results from this study indicate that the charcoal contribution from one prescribed fire is small relative to the distribution of total ecosystem C (Table 3.3), the long-lived nature of this C pool could result in a larger contribution to total ecosystem C with repeated burning, on an extended temporal scale. Unlike nearly all plant residue C that enters the soil, decomposes, and returns to the atmosphere or is leached from soils within a few decades, charcoal persists for hundreds of years and remains onsite or immediately nearby (Lynch et al., 2004; Trumbore and Czimczik, 2008). Based on our model of C dynamics under scenarios with and without repeated burning, restoring a natural fire regime in these historically frequent-fire forests could result in a much larger fraction of total ecosystem C stored in this recalcitrant C form in comparison to a scenario that only includes burning by wildfire (Fig. 3.4). Supporting our findings, prior investigations relating the effects of fire
intensity and charcoal amount, found that accumulation was higher following low-intensity fires relative to high-intensity fires (Knicker et al., 2006; Cerda, 2010).

While our results demonstrate that prescribed burning increases charcoal production and the rate at which varies with fuel loads and forest structure, uncertainty remains regarding a mechanistic understanding of charcoal production and subsequent chemical properties, which warrants further investigation. Similar to Deluca and Aplet (2008), our estimates of C accumulation in charcoal with repeated burning suggest that a simple mass-balance approach to estimating emissions from fire may yield an overestimate of emissions and an underestimate of post-fire total ecosystem C. Improving our understanding of the effects of restoring natural fire regimes on charcoal C will improve our ability to quantify the effects of restoration treatments on forest C dynamics.

3.7 Literature Cited


Chapter 4: Conclusion

Following wildfire risk-mitigation techniques we found that selective thin-from-below followed by prescribed burning may most effectively stabilize C and restore ecological resilience to Sierran mixed-conifer forests. The retention of additional midsized trees to buffer against treatment induced mortality and C loss from larger individuals may provide an opportunity to meet wildfire risk mitigation objectives while reducing losses of live tree C. While there are C costs associated with prescribed burning, it is necessary to restore this natural process following thinning to reduce surface fuel loads and maintain ecological processes and ecological resilience in this forest system. Retaining large diameter trees and selectively removing midsize, fire-intolerant tree species (A. concolor, A. magnifica, C. decurrens) can contribute a forest ecosystem that is more resilient to changing climate and resistant to stand-replacing wildfires.

Our results also demonstrate that prescribed burning increases charcoal production and that the rate of charcoal production varies with fuel loads and forest structure. There is still uncertainty regarding a mechanistic understanding of charcoal production and subsequent chemical properties, which warrants further investigation. Our estimates of C accumulation in charcoal with repeated burning suggest that a simple mass-balance approach to estimating emissions from fire may yield an overestimate of emissions and an underestimate of post-fire total ecosystem C. Improving our understanding of the effects of restoring natural fire regimes on charcoal C will improve our ability to quantify the effects of restoration treatments on forest C dynamics.