Project Title: Fuels treatment effects on carbon stocks and insect induced mortality 10-years after treatments

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I. Abstract

Forests play a vital role in regulating climate by sequestering carbon from the atmosphere. Fire results in direct and indirect emissions of carbon to the atmosphere. In historically frequent-fire forests, post-fire tree mortality was considerably lower than current mortality levels following high-severity wildfire in fire-suppressed forests. Treatments to reduce high-severity wildfire risk require carbon removal by thinning and carbon emissions by prescribed burning. Understanding how these treatments influence total ecosystem carbon is necessary for better understanding the tradeoffs between treatment and wildfire severity. We evaluated the 10-year post-treatment carbon dynamics and insect and pathogen induced mortality in a Sierran mixed-conifer forest.

We found that the emissions from prescribed burning were resequestered by subsequent tree growth within the ten year, post-treatment period. Treatments that retained large trees had a positive carbon balance, whereas those that removed large trees had a negative carbon balance ten years after treatment. When investigating post-treatment tree growth, we found that climate sensitivity varied by species. White fir and sugar pine were the most sensitive to drought. However, white fir had the shortest post-drought recovery of the species examined. Insect and pathogen-induced tree mortality was highest in treatments that included both thinning and burning.

II. Background and Purpose

Forests sequester substantial amounts of carbon (C), which in the US equates to approximately 10% of annual anthropogenic emissions (Woodbury et al. 2007). The carbon sequestration potential of forests has garnered the attention of policy makers seeking a solution for changing climatic conditions, resulting in the development of state, regional, national, and international protocols for quantifying C in forests (e.g. Climate Action Reserve, Clean Development Mechanism). While C sequestration by forests presents a climate mitigation benefit, that benefit can be reversed by natural disturbances that release carbon dioxide back to the atmosphere (Galik and Jackson 2009). Fire and insect outbreaks are two of the most significant risks to forest C. Fires directly release substantial amounts of C to the atmosphere (Campbell et al. 2007; Wiedinmyer and Neff 2007) and large scale insect outbreaks have been found to transition forests from C sinks to sources (Kurz et al. 2008). Fuels treatments can reduce fire severity and may reduce insect mortality by reducing moisture stress in leave trees (Fettig et al. 2008). In the dry, temperate forests of the western US, a legacy of fire suppression has led to a change in fire frequency, and severity (McKelvey and Busse 1996; Miller et al. 2009). Fire is also being influenced by changing climatic conditions, leading to an increase in fire size (Westerling et al. 2006), a trend which is expected to increase as climatic conditions continue to change (Westerling et al. 2011). Efforts to reduce the risk of high-severity fire in these dry, temperate forests typically involve mechanical fuels reduction and/or prescribed fire. Fuels reduction treatments have C costs in the form of biomass loss from thinning, emissions from prescribed fire, and equipment emissions (Finkral and Evans 2008; North et al. 2009). However, previous research has found that the C penalty incurred from fuels reduction treatments can be more than offset by avoided emissions when wildfire occurs (Hurteau et al. 2008; Hurteau and North 2009, Earles et al. 2014). The balance of C costs and benefits is dependent on forest type; in temperate rainforests of the Pacific Northwest, fuels treatments can reduce C storage when landscape-scale fire occurrence is factored into the calculation, but in dry temperate forests fuels treatments can yield a net C storage benefit (Mitchell et al. 2009).

That fuels reduction treatments can have a net C benefit is based largely on simulation modeling (e.g. Hurteau and North 2009, Mitchell et al. 2009). These models use growth relationships derived from empirical data that are largely historical in nature. Determining the effects of mechanical thinning and burning treatments on tree growth and thus C sequestration under current climatic conditions is essential for assessing the C costs and benefits of fuels reduction treatments. Furthermore it is necessary to determine how old, large trees respond to thinning, as these trees contain a majority of the live tree-based C in the forest (North et al. 2009). Studies have suggested large tree die-offs may be increasing in some forests (Smith et al. 2005; van Mantgem et al. 2009). The mechanisms behind such die-offs are complex, but in fire-dependent forests often include a cascade of effects linking water stress and bark beetle attacks (Ferrell et al. 1994; McDowell et al. 2008). Drought can produce prolonged periods of stomatal closure preventing CO_2 absorption needed for growth, which may reduce a tree's ability to ward off pests (McDowell et al. 2008).

Thinning dense, fire-suppressed stands of conifers reduces the amount of transpiring leaf area in a stand, often reducing evapotranspiration and increasing soil water content (Zou et al. 2008). Even when no difference is detected between soil water content in thinned and control stands, it is common to detect improved hydrological status of the remaining trees. This may be due to increased per tree water availability (Brodribb and Cochard 2009). Fuels reduction thinning has reduced water stress, as measured by pre-dawn water potential, in many ponderosa and Jeffrey pine stands (Sala et al. 2005; Wallin et al. 2006). Prescribed fire can also reduce tree density but some studies have found an immediate, short-term ($\leq$ 2 years) increase in bark beetle damage and tree mortality (Fettig et al. 2008, Maloney et al. 2008). Longer-term tree response to prescribed fire has not been as well studied although in one experiment soil moisture did increase but it was not detected until seven years after treatment (Feeney et al. 1998; Wallin et al. 2004, Kerhoulas et al. 2013).

The objective of our research was 1) to quantify the longer term impacts of fuels treatments on the growth of residual trees and stand-scale C sequestration; 2) determine how growth response in large old trees differs from young small trees and; 3) quantify the effects of fuels treatments on soil moisture, insect damage, tree mortality, and subsequent fuel loading.

III. Study Description and Location

Site Description: This research was conducted at the Teakettle Experimental Forest in California's southern Sierra Nevada Mountains. The site ranges in elevation from 1900-2600 m and has a Mediterranean climate with almost all of the 125 cm of annual precipitation falling as snow (North et al. 2002). Teakettle's mixed-conifer forest is comprised primarily of white fir (*Abies concolor*), red fir (*A. magnifica*), incense-cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), and Jeffrey pine (*P. jeffreyi*) (Rundel et al. 1988). Eighteen permanent four hectare plots were established that represented the range of variable forest conditions at the site. A pre-treatment analysis of stand conditions indicated that there were no significant differences in forest structure (North et al. 2002).

Six treatments form a full factorial design that included three levels of mechanical thinning treatment (no thin, understory thin, overstory thin) crossed with two levels of prescribed burning treatment (no burn, prescribed fire), and were applied to the plots (3 reps of each) from 2000-2001. The understory thin treatment removed all trees between 25 and 75 cm diameter at breast height (dbh), following California spotted owl guidelines (Verner et al. 1992). Although initially designed to minimize impacts on spotted owl habitat, the guidelines have been widely implemented to reduce fuels in Sierran mixed-conifer forests. The overstory thin removed all trees > 25 cm, except 22 large trees ha^{-1} left regularly spaced approximately 20 m apart. In general this produced a sparse canopy with regular, widely separated tree crowns typical of defense zones adjacent to the wildland-urban interface (WUI). The thin and burn plots were mechanically treated in 2000 and burned in 2001. The thin-only plots were treated in 2001.

Data Collection: Pre-treatment data collection methods included mapping, using a surveyor's total station, measuring, and permanently tagging all trees and snags $\geq$ 5 cm DBH. Fuels, fine roots, soil C, soil moisture, and understory plant cover were measured on permanent sample points established on a grid within each plot. Understory plants (herbs and shrubs) were sampled using a 10 m^2 circular plot and re-surveyed each year through 2006. Soil moisture was measured every two weeks at two depths (0-15 cm and 15-40 cm) using time domain reflectometry (TDR). Mass of the fine woody debris (FWD) was estimated before and after treatment, (the controls were only sampled once), using the planar intercept method (Brown 1974), with modifications, at nine sample points within each plot. For the 1000-hour fuels a cut-off was made in the upper range of the fuel size to avoid overlapping with a complete coarse woody debris (CWD) inventory, in which the endpoints of all pieces ≥20 cm diameter were mapped, the diameters recorded and decay class assigned (Maser et al. 1979). Log length was calculated using the mapped coordinates and the volume of each log was estimated as a frustrum paraboloid (Husch et al. 1993). Species was often unidentifiable, as a result mass (Mg ha^{-1}) was estimated using the average specific gravity by decay class for the species at Teakettle from Harmon et al. (1987). A detailed reporting of the pre and post-treatment methods can be found in North et al. (2009), Zald et al. (2008), Wayman and North (2007), and Innes et al. (2006).

Data collection for this research also included re-measuring all trees that were measured during the immediate post-treatment measurement period and any ingrowth. Each individual tree's tag number, status, dbh, height, and height to live crown were measured. All snags were re-measured and the appropriate decay class recorded. Soil C, fuels, and fine roots were re-measured following the same protocols used in the initial measurement periods, at the same nine gridpoints in each plot (n=162). We re-measured fuels, including CWD, using the same methodology along the same three transects at each of 162 gridpoints sampled before and immediately after treatment. Soil C and fine root sampling followed the same post-treatment protocol presented in Wayman and North (2007) and Soung-Ryoul et al (2009), where at the same grid points three 2 cm diameter soil cores were extracted and aggregated by 0-10 cm and >10-30 cm depths. Total C analysis was be conducted by DANR at University of California, Davis.

Quantifying Climate Sensitivity: We selected 72 trees for dendrochronological measurements. Sample trees were stratified by species and treatment type, with three replicates in each category. This portion of the study excluded red fir, since it occurs primarily in the riparian buffer zone. The remaining four species comprise > 95% of the basal area in the treatment units. From each sample tree we measured diameter and collected increment cores at DBH, the base of the live crown, and the top. Core depth was determined by the distance necessary to obtain 20-years of pre-treatment growth and the post-treatment growth period. We measured annual growth using a WinDendro system. Tree cores were cross-dated following Stokes and Smiley (1986) and detrended by applying a linear regression or negative exponential curve as a function of fit and subtracting growth values from the detrending line/curve.

Assessment of Tree Mortality Agents: We had previously made assessments of all living and dead trees on the Teakettle Forest and their links to biotic mortality agents. The first census of tree mortality agents was conducted in 2000-2002 prior to any treatments at the site (Smith et al, 2005); a follow-up post-treatment census was conducted in 2004-2005 (Maloney et al. 2008). Many studies have been conducted on individual forest insect and pathogen species, their direct effects on trees, and their interactions to cause tree death. Assessment of insects on living trees relied on observations of characteristic signs and symptoms (Furniss and Carolin 2002; Sinclair and Lyon 2005). Bark beetle attacks were confirmed by the presence of boring dust or pitch tubes. Dead trees were dissected to determine apparent mortality agents and contributing factors, including bark removal to identify characteristic galleries of bark beetles (Furniss and Carolin 2002). Trees weakened or killed by other agents (i.e., pathogens) may be later attacked by bark beetles. When this occurs, beetles attacks are generally sparser and there would be little resinosis around the galleries. Stand level characteristics were taken into consideration when diagnosing causes of tree death (Rizzo and Slaughter, 2001). For example, group kills by bark beetles will often result in a cluster of trees that appeared to have died at the same time; i.e., each of the trees will be in the same condition or state of decay. In contrast, root disease centers are characterized by clusters of dead trees that have died at many different times so that a range of decay classes are apparent. In addition, enlarging root disease gaps usually have living, symptomatic trees at their borders indicating spread of the pathogen, while this is not apparent in mortality gaps caused solely by bark beetles. These data were then used to plot spatial distribution.

Carbon Calculations and Data Analysis: We quantified C 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 and duff 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).

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.

Tree Mortality Data Analysis: A post-treatment census of tree mortality and biotic mortality agents was conducted in 2011-2013 to assess the effects of treatments on levels of pathogens and bark beetles and subsequent tree mortality a decade after treatments. Data were analyzed using ANOVA to compare differences in tree mortality among treatments and Tukey's HSD was used for between treatment comparisons. We used Logistic regression to assess the relationship between tree mortality and tree density at the tree level. Stem map data were used to calculate a density index for each tree using Theissen (i.e., Voroni) polygons (Smith et al. 2005). The area around each tree is bisected by an equidistant line between adjacent stem locations and the lines are connected to form a polygon around each tree location. The polygon's area is a rough approximation of the potential growing space, in square meters, for an individual tree. Polygons were weighted by dividing each polygon's area by the basal area of the individual tree to take into account the greater growing space demands (i.e., light, water, and nutrients) of larger trees. The size and distribution of Theissen polygons has been commonly used to evaluate the impact of density, growing space, and competition of neighboring plants on plant succession. Assumptions were met for each statistical analysis and a p-value of 0.05 was used to evaluate significance.

IV. Key Findings (1-2 paragraph discussion of each)

Carbon Stocks: 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. 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. 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. 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. 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. 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. 1). The overstory-thin, and overstory-thin and burn continued to have a C debt of 6 and 45 Mg C ha^{-1}, respectively (Fig. 1).

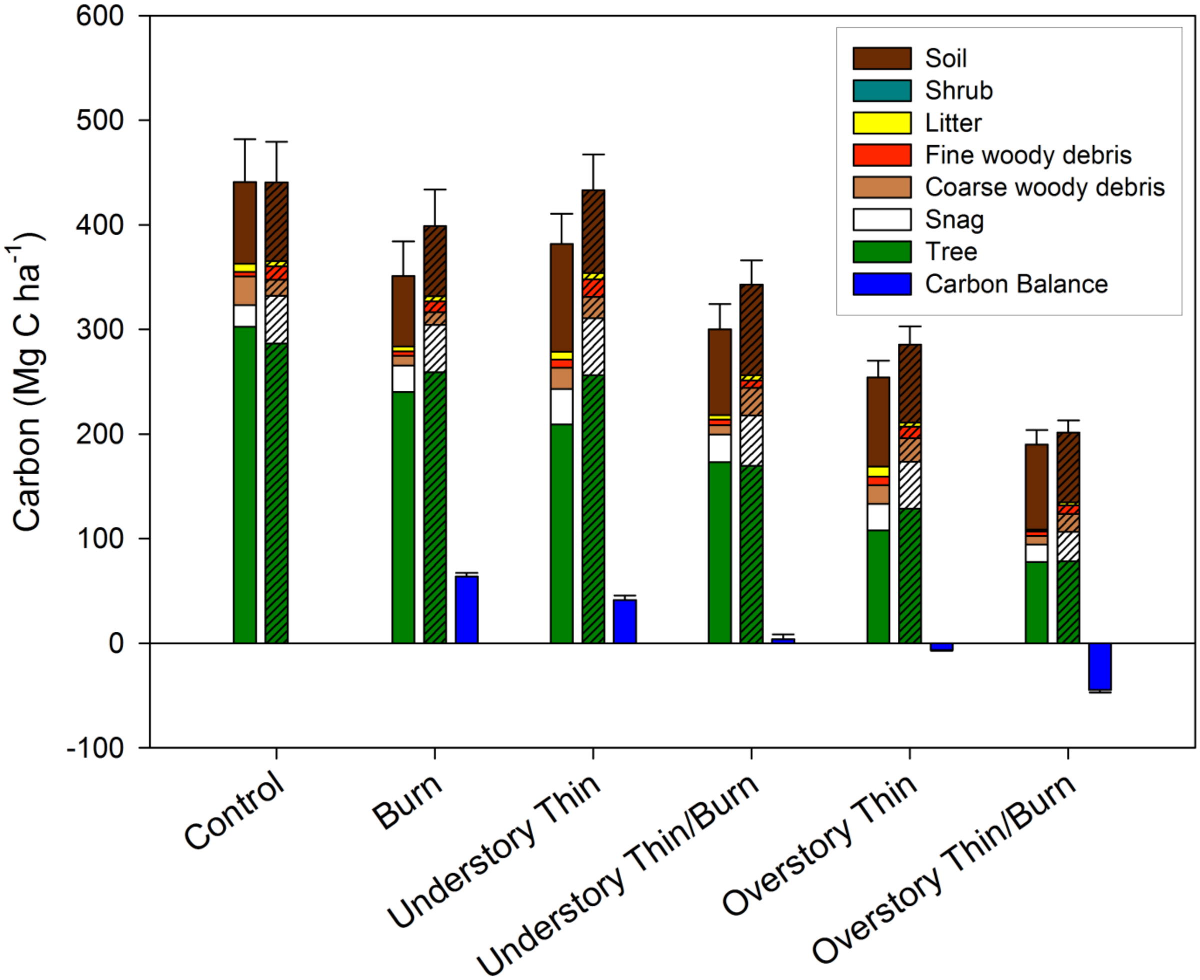

Figure 1: Mean and standard error of C pools immediately post-treatment (2002, colored bars) and 10-years post-treatment (2011, colored and hashed bars) in Mg C ha^{-1}. Carbon Balance (solid blue bar) is the 10-year C stock gain minus C removed and emitted during treatment implementation in Mg C ha^{-1}.

Tree Response to Climate: This study has provided the opportunity to examine the climate sensitivity of a mixed conifer forest, both prior to the fuels reduction treatment and after treatment. During the pre-treatment period (1970 – 2000) there were two severe drought years and 6 very dry years that allowed comparisons among species and across position within tree crowns of growth response to drought. Growth response was assessed in several ways: 1) *resistance* to drought was determined from how much growth was reduced in the drought year compared to the average of the 5 years prior, 2) *recovery* was determined as the rate of return to a stable growth rate in the years after the drought, and, 3) *resilience* was determined as the degree of recovery of the pre-drought growth rate in the years after the drought year. Among the four dominant species, white fir and sugar pine were least resistant to drought years, yet white fir was also the species that recovered most rapidly from drought years. Sugar pine was least resilient, while incense-cedar and Jeffrey pine were highly variable in terms of both impacts of drought and recovery from drought. We hypothesized that fuels reduction treatments would reduce the impacts of drought because these treatments potentially reduce tree competition for water, with this effect being stronger in treatments that reduced more basal area. We are in the process of testing this idea,

We also investigated whether tree response to climate varied with crown position, hypothesizing that drought impacts on radial growth would increase with height in the tree because the upper crown experiences greater water stress than the lower crown (e.g., Kerhoulas and Kane, 2012). Unexpectedly, only one species, sugar pine, exhibited a clear pattern of lower resistance and resilience to drought with increased height. Also contrary to expectation, two species, white fir and incense cedar, showed greater resistance at higher crown positions. The variation in drought response among these co-occurring conifer species reveals a basis for changing species composition in this mid-Sierran forest in response to ongoing climatic change.

<u>Tree Mortality</u>: Tree mortality was significantly different between treatments with the highest levels in treatments that included thinning and burning. Mortality across all treatment ranged from 12% in the understory thin to 45% in the overstory thin and burn. Mortality across all conifer species ranged from 15% for Jeffrey pine to 22% white fir. Percentage mortality for each species was proportional to the percentage of each species present. Mortality was not correlated with stand density at the tree-level. Tree density (Theissen polygon) for each tree was not a good predictor of mortality in our logistic regression analysis.

Biotic agents that contributed prominently to mortality based on the presence of signs and symptoms included the root pathogens *Armillaria gallica* and *Heterobasidion occidentale*, dwarf mistletoe, and bark beetles. *Armillaria gallica* is often considered a weak pathogen of conifers and hardwood hosts but may be a primary pathogen in some circumstances (Baumgartner and Rizzo 2001). In this study dead white fir were often found to be colonized by *A. gallica* without any signs or symptoms of other pathogens or bark beetles suggesting it is causing white fir mortality in this system. The incidence of *A. gallica* on dead trees and stumps across treatment plots ranged from 4% to 35% with no significant difference between treatments. *Heterobasidion occidentale* is one of the most important agents of mortality of white fir. The fungus is common in forests dominated by white fir and can cause mortality to healthy trees. The incidence of *H. occidentale* on dead trees and stumps across treatment plots ranged from 3% to 50% with no significant difference between treatments. Bark beetles are common biotic agents that contribute to mortality of trees in mixed conifer forests. Under most circumstances bark beetles attack weakened trees such as trees infected by pathogens or under drought conditions. In this study most bark beetle attacks occurred on trees already infected by pathogens or on trees that were damaged by fire. Bark beetles occurred on 2% to 20% of live and dead trees across species and from <1% to 38% across plots with no significant difference between treatments. Dwarf mistletoes are angiosperm parasites of conifers that are capable of causing mortality but are most important in predisposing trees to attack by other pathogens and bark beetles. The incidence of dwarf mistletoe across plots ranged from 1% to 42% with no significant difference between treatments.

V. Management Implications

<u>Carbon Stocks</u>: 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 risk of stand-replacing wildfire and the C emitted from potential severe fire events (Hurteau and North 2009, Hurteau *et al.*, 2014). Selective thin-from-below followed by prescribed burning may most effectively stabilize C in 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 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 to a forest ecosystem that is more resistant to C loss from stand-replacing wildfires.

<u>Tree Mortality</u>: Thinning and burning treatments are commonly applied to Sierra Nevada forests to reduce fuels, tree densities, and increase the proportion of pine species relative to fir species. In this study treatments that included thinning and burning combined had the highest levels of mortality compared to burning and thinning alone and control plots. Fire damage during burn treatments can lead to elevated bark beetle and pathogen attack presumably from wounding and decreased defensive abilities (Maloney et al 2008). Mortality in the burn-only treatment was not significantly different from the control and understory thin treatments suggesting it is the combination of burning and thinning causing higher mortality. Thinning treatments can increase the incidence of root pathogens, especially *Heterobasidion* sp, by increasing the amount of wood resources (stumps) available for saprotrophic colonization. This can lead to increased mortality of susceptible tree species via root-root transmission from stumps to adjacent trees. Mortality in burn/thin plots is likely due to a combination of bark beetle attack on fire-stressed trees and elevated fir mortality from thinning. Subsequent thinning treatments may further increase fir mortality from the build-up of Heterobaisidion inoculum in stumps. Forests with a regular fire regime may suffer less fire damage and subsequent mortality because of lower surface fuel levels.

VI. Relationship to other recent findings and ongoing work on this topic

<u>Carbon Stocks</u>: 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 declined in the control highlights the difficulty in quantifying absolute change in carbon stocks against a pre-treatment baseline; 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 1, Fig. 1). In the understory-thin and burn, the increase in snag C is largely due to increased mortality in trees > 130 cm diameter (Fig. 2d, Fig. 1). This finding is especially relevant today, as thinning from below 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). Declines in live tree C for both species groups were driven by reductions in the largest diameter class (>130cm DBH) (Fig. 2). van Mantgem *et al.* (2013) found that mortality rates were similar for large (> 50cm 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 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.*, 2013). Low-intensity fires have the advantage of reintroducing critical ecological processes (Hurteau *et al.*, 2014). 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.

<u>Tree Mortality:</u> The range of mortality found in this study is similar to ranges found in other studies of conifer systems in California (Hawkins and Henkel 2011, Das et al 2011, van Mantgem et al 2009). Pre-treatment mortality ranged from 5% to 13%, the first post-treatment mortality ranged from 3% to 56%, and mortality a decade after treatment ranged from 12% to 45% (Smith et al 2005, Maloney et al 2008). In all assessments percentage of dead trees by species was the similar to percentage of live trees by species suggesting mortality is related to abundance of each species in the stand. All the biotic agents of mortality found in the most recent census are well studied pathogens and insects that are known to

cause disease and mortality of conifers (Hawkins and Henkel 2011, Maloney et al 2008, Smith et al 2005, Sinclair and Lyon 2005).

Competition is often invoked to explain non-random patterns of mortality in western conifer forests. Indeed, competition likely contributes to mortality under many conditions. Studies aimed at assessing the importance of competition in determining tree mortality often rely primarily on spatial patterns of live and dead trees and attribute density-dependent mortality to competition (Das et al 2008, Das et al 2011). In this study tree density alone was not sufficient to explain spatial patterns of tree mortality. Although burn/thin treatments were correlated with higher levels of mortality, more data are needed to explain the patterns of mortality in this system. Specifically, environmental data (moisture index, aspect, slope, etc.), and fire intensity data may improve model performance.

VII. Future Work Needed

Carbon Stocks: 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. Obtaining a better understanding of treatment longevity from initial entry (thinning and burning) and subsequent maintenance burning is essential for improving our ability to project how forests will respond to climate-driven changes in wildfire size and frequency.

Tree Mortality: Fire suppression has excluded fires from most areas of mixed conifer forest in the Sierra Nevada over the last century (McKelvey et al 1996). Almost all studies investigating forest pathogens and their effects on tree mortality have been conducted during this era of fire suppression. As a result tree mortality and pathogen dynamics under a restored fire regime are not well understood. It will be necessary to monitor pathogens and tree mortality after repeated burning in order to design future fuels and restoration treatments. Lower density, wider tree spacing, and greater tree diversity fostered by frequent fires may reduce pathogen incidence and spread (Parker et al 2006). Many forest pathogens are host specific and are dispersed most efficiently over short distances. For example, dwarf mistletoe species are often specific to one or two conifer species and are typically dispersed within 10 meters of infected branches (Hawksworth and Wiens 1996). Greater tree diversity and wider tree spacing can potentially lower the probability of successful dispersal to an adjacent susceptible tree (Parker et al 2006). In addition, fuels treatments may reduce coarse woody debris that can harbor root pathogenic fungi such as *Armillaria* and *Heterobasidion*. Root pathogenic fungi, such as *Armillaria mellea*, may be able to live saprotrophically for centuries in coarse woody debris (Rizzo et al 1998). Understanding long term pathogen dynamics and tree mortality trends in the context of repeated burning is critical to designing fuels and restoration treatments to meet specific management goals.

VIII. Deliverables Crosswalk

Proposed	Delivered	Status
Workshop/Field trip	1) Presentation at the Southern Sierra Fire and Hydroclimate Workshop, 22-24 April 2014 2) 2012 Sierra Nevada fire field trip for Scientists and Managers 3) Dinkey Landscape Restoration Project stakeholder field trip, June 2011 4) Presentation at the Community Scale Bioenergy from Woody Biomass: Policies and Technologies. Blodgett Experimental Forest, CA. 5) Presentation at Forest Carbon and Woody Biomass Utilization: PUDs and SB1122. CA Energy Commission, Sacramento, CA	1) Complete 2) Complete 3) Complete 4) Complete 5) Complete
Research Brief	10-year post-treatment carbon dynamics www.hurteaulab.org/outreach.html	Complete
Website	http://www.hurteaulab.org/jfsp.html	Regularly updated
Conference Presentations	1) Wiechmann, M.L., M.D. Hurteau. The effect of thinning and burning on charcoal formation and carbon storage in a mixed-conifer forest, Sierra Nevada, California. 2014 meeting of the Ecological Society of America. 2) Zald, H., M.D. Hurteau, G.W. Koch, M.P. North. Growth responses of old growth trees to climate along a vertical canopy gradient. 2014 meeting of the Ecological Society of America. 3) Wiechmann, M.L., K.L. Martin, M.P. North, M.D. Hurteau. Carbon recovered following different fuel reduction treatments in a Sierra Nevada mixed-conifer forest. 2013 meeting of the Ecological Society of America. 4) Hawkins, A.E., D.M. Rizzo. Pathogen. Pathogen and pest responses to forest management in the southern Sierra Nevada. 2014 meeting of the American Phytopathological Society	Complete
Peer-reviewed Publications (3)	1) Wiechmann, M.L., M.D. Hurteau, M.P. North, G.W. Koch, L. Jerabkova. 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) Koch, G.W., Zald, H.J.S., M.P. North, M.D. Hurteau. Climate sensitivity variation with tree height. 3)Hawkins, A.E., D.M Rizzo, M.D. Hurteau, M.P. North Pathogens, bark beetles, and tree mortality a decade after burning and thinning treatments in a Sierra Nevada mixed-conifer forest	1) In review 2) In progress 3) In progress

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