Wildfire and fuel treatment effects on carbon storage, eastside Cascade Range, Washington, USA

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A thesis submitted in partial fulfillment

of the requirements for the degree

Master of Science

University of Washington

2012

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TABLE OF CONTENTS

LIST OF TABLES

Section 1:

Section 2:

Section 3:

LIST OF FIGURES

Section 2:

Section 3:

EXECUTIVE SUMMARY

The primary objective of this thesis is to quantify trade-offs in carbon (C) storage between fuel reduction treatments (fuel treatments) and C release from wildfire. Two separate research questions pertaining to trade-offs in C storage were explored to identify: (1) Short-term treatment and wildfire effects on C storage following the 2006 Tripod Complex Fire; and (2) Long-term patterns in C storage across multiple future management and disturbance scenarios. Each question corresponds to a section in the thesis manuscript and is preceded by a comprehensive literature review and synthesis of scientific evidence for treatment and wildfire effects on C storage in dry forest ecosystems.

Section 1:

In this comprehensive literature review, I summarize the scientific evidence for how fuel treatments and wildfire affect long-term C sequestration in dry forest ecosystems, an emergent focus in forest management in the context of climate change. The review is organized by individual elements of C release associated with fuel treatment application (i.e., thinning, prescribed fire emissions, wildfire emissions, equipment usage) and by specific topics within the relevant body of scientific literature. I compare the different approaches used for calculating C budgets, effects of various assumptions used in empirical analysis and modeling, and influence of temporal and spatial scales on inferences.

Section 2:

The literature review revealed that analyses of wildfire and C release in dry forests rely

mostly on simulated wildfire events and rarely examine *in situ* effects of wildfire/treatment interactions. In this study, I use empirical data, in combination with high-resolution (25-m), spatially explicit fuel maps to explore treatment effects on C release from the 2006 Tripod Complex Fire in the Okanogan-Wenatchee National Forest (OWNF), Washington (USA). Sources of C release associated with the application of fuel treatments (i.e., thinning, prescribed fire) are summarized and compared to wildfire C release from within the treatment units.

Section 3:

Uncertainty in future wildfire frequency and extent complicate long-term (decades to centuries) assessments of trade-offs between C release from fuel treatments and wildfire. In this study, I simulate a broad range of treatment intensities, and stochastically model wildfire at multiple frequencies over a 100-year simulation period. Simulated future wildfire frequencies correspond to mean fire-free intervals (MFFI) ranging from complete fire suppression to the pre-settlement MFFI, as estimated from fire scar records at low elevations in the OWNF. Consideration of multiple future management and disturbance scenarios provides an opportunity to assess the capacity at which stands in the study domain sequester and retain C over time, given expected changes to the fire regime as a result of climate change. The objective of this study is to see if patterns in C storage and C release observed in the 2006 Tripod Complex Fire persist at an expanded temporal horizon.

Conclusions:

Analyses of C dynamics from treatment units within the Tripod fire perimeter (Section 2) reveal strong evidence that fuel treatments that include prescribed fire (thin-and-burn)

successfully reduce C emissions from wildfire relative to control treatments (12 to 16 Mg C ha^{-1}). Yet, C release associated with the implementation of treatment units (53 to 90 Mg C ha^{-1}) exceed observed reductions in C emissions from wildfire, reversing any potential C benefit to the system even after consideration of the potential reduction in C release from the decomposition of fire-killed trees (11 to 20 Mg C ha^{-1}). This study improves the resolution and accuracy at which forest fuels are considered for C budgets, but inferences on C dynamics in dry temperate forests are incomplete without consideration of C at an expanded temporal scale (decades to centuries). The simulation of multiple future scenarios, bracketing a broad range of treatment intensities and wildfire frequencies (Section 3), strengthens inferences of wildfire and treatment effects on C storage and facilitates a discussion of C dynamics in the context of expected changes to fire frequency as a result of climate change. Results suggest that C stores will decrease, regardless of treatment level, as wildfire frequency increases. The failure of any treatment intensity in the simulation design to recoup treatment costs via reductions in C emissions from wildfire suggests that patterns in C storage and C release observed in Section 2 may persist over time and that fuel treatments result in a net reduction of C stores at short and long time scales. The primary long-term benefit to C storage from fuel treatments may be to stabilize C stores and reduce the likelihood of extreme negative C outcomes, such as vegetation life-form conversion.

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Abstract

SECTION 1:

Wildfire and fuel treatment effects on forest carbon dynamics: A synthesis of concepts and data

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Sequestration of carbon in forests has the potential to mitigate effects of global climate change by offsetting future emissions and greenhouse gas concentrations in the atmosphere. In dry temperate forests, however, wildfire is a natural disturbance agent with the potential to release large fluxes of carbon into the atmosphere. Climate-driven increases in wildfire frequency, extent, and severity are expected to increase the risks of reversal to carbon stores and affect the potential of dry forests to sequester carbon. Fuel treatments that successfully reduce surface fuels in dry forests can mitigate the spread and severity of wildfire, while reducing both tree mortality and emissions from wildfire. However, heterogeneous burn environments, site-specific variability in post-fire ecosystem response, and uncertainty in future fire frequency and extent complicate assessments of long-term (decades to centuries) carbon dynamics across large landscapes. Results of studies on the effects of fuel treatments and wildfires on long-term carbon

retention across large landscapes are limited and equivocal. Current stand-scale studies, empirical and modeled, describe a wide range of (1) total treatment costs (12 – 116 Mg C ha^{-1}) and (2) reductions in wildfire emissions between treated and untreated stands (1 – 40 Mg C ha^{-1}). Conclusions suggest the direction (source, sink) and magnitude of net carbon effects from fuel treatments are similarly variable (-33 Mg C ha^{-1} to +3 Mg C ha^{-1}). Studies at large spatial scales have shown that the low probability of high-severity wildfire events for any given treated stand can negate any expected carbon benefit.

1.1 *Introduction*

Forest biomes play a key role in the global carbon (C) cycle. C is fixed through photosynthesis from atmospheric carbon dioxide (CO_2) and can be stored for centuries in live biomass, detritus, and soil organic matter in forested ecosystems. As a result, the sequestration of C in forests has the potential to mitigate effects of global climate change by offsetting future emissions and greenhouse gas (GHG) concentrations in the atmosphere. In the United States (US), forests annually sequester 149 - 330 million Mg C, or the equivalent of approximately 10% of C emissions in the US (Woodbury et al., 2007).

Political and market-based efforts to reduce or offset GHG emissions and mitigate the effects of climate change, such as the Clean Development Mechanism, Voluntary Carbon Standard, and Regional Greenhouse Gas Initiative, emerged after the adoption of the Kyoto Protocol in 1997 (Metcalf 2009; Fahey et al., 2010). In 2001, the US Department of Energy sponsored the First National Conference on Carbon Sequestration. By the time the Kyoto Protocol went into effect in 2005, institutions such as the Chicago Climate Exchange (CCX), California Climate Action Registry (CCAR), Verified Carbon Standard (VCS), and other voluntary trading programs had been established to measure, monitor, and trade GHG emissions. Policy frameworks and market-based programs aimed to encourage C sequestration in forests have since expanded in the US and internationally.

The effects of wildfires and management on forest C storage, however, were poorly addressed by original C sequestration policy in the US (Stephens et al., 2009). Forest thinning was considered a C source to the atmosphere regardless of reduced risks to future wildfire and associated C losses, according to guidelines established in the Kyoto Protocol and CCAR (2007) (Hurteau et al., 2008). Attention to contributions from wildfire emissions and the treatment of

hazardous fuels in the C budget is critical in dry forest ecosystems historically maintained by fire. A discussion of treatment and wildfire effects on C storage emerged in response to the failure of policy to recognize the magnitude of C release in wildfire events (Hurteau et al., 2008; Stephens et al., 2009).

In this synthesis, we summarize the scientific evidence for how fuel treatments and wildfire affect long-term C sequestration in dry forest ecosystems, an emergent focus in forest management in the context of climate change. The synthesis is organized by individual elements of C release associated with fuel treatment application (i.e., thinning, prescribed fire emissions, wildfire emissions) and by specific topics within the relevant body of scientific literature (i.e., thinning and wildfire effects on carbon flux). We compare the different approaches used for calculating C budgets, effects of various assumptions used in empirical analysis and modeling, and influence of temporal and spatial scales on inferences.

1.2 *Fuel treatments: Background and objectives*

Forest structure, fuel characteristics, and fire regimes in millions of hectares of dry forest ecosystems in the western US have been significantly altered in the past century (Graham et al., 2004). Legacies of fire suppression, fire exclusion, grazing, and timber harvest have resulted in widespread accumulation of surface and canopy fuels and, in turn, have increased the probability of severe and extensive wildfires compared with pre-settlement forests under natural disturbance regimes (Stephens 1998). Alteration of fire regimes is greatest in arid to semiarid forests, primarily forests dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) or both, which formerly had more frequent and lower severity wildfires than today (Agee 1993). Fire exclusion and fire suppression have contributed

to increases in overstory stem density and growth of shade-tolerant, fire-intolerant vegetation and the quantity, vertical arrangement, and horizontal continuity of forest fuels (Agee 1993).

In addition to contributions to fuel accumulation from past management practices, wildfire is substantially controlled by climate and is expected to continue to increase under global warming scenarios (McKenzie et al., 2004; Littell et al., 2009). Climatic conditions favorable to large wildfire events, such as seasonal water shortages and longer fire seasons, are also expected to become more widespread (Westerling et al., 2006). Climate-driven increases in wildfire frequency, extent, and severity (McKenzie et al., 2004, Westerling et al., 2006) are, in turn, expected to affect the potential of forest ecosystems to sequester C (Deal et al., 2010). Changes in disturbance regimes affect forest age-class structure as well as forest dynamics, and are expected to strongly affect C budgets (Kurz et al., 1995). Moreover, wildfires emit additional CO_2 into the atmosphere and act as a positive feedback that may exacerbate effects of climate change (IPCC 2007).

Hazardous wildfire conditions are now widespread across the western US as a result of changes forest composition, fuel structure, and fire regimes. The rapid expansion of the wildland-urban interface (WUI) has motivated policy and actions focused on reducing wildfire risks to people and homes (Youngblood 2005). Furthermore, implementation of the National Fire Plan (2001), Healthy Forests Initiative (2003) and the Healthy Forests Restoration Act (2003) has directed new resources to achieve objectives in reducing hazardous fuels and restoring fire-adapted ecosystems (Winter et al., 2004). More recently, the Federal Land Assistance, Management and Enhancement Act (FLAME) of 2009 requires the US Forest Service and the US Department of Interior to submit to Congress a report that contains a cohesive wildfire management strategy. Included in the cohesive management strategy is a framework for the

allocation of hazardous fuels reduction funds based on the priority of hazardous fuels reduction projects, and the identification of the most cost-effective means for allocating fire management budget resources (Calkin et al., 2011). Consequently, forest managers are challenged to reduce potential fire hazard in dry forests through expanded and effective fuel treatment programs.

Federal land management agencies and private land owners treat hazardous forest fuels using a combination of mechanical thinning, prescribed fire, and other mechanical strategies such as mastication. The primary objective of fuel treatments is to reduce fuel loads (i.e., the quantity of fuel) and change the fuel profile (i.e., the spatial arrangement of fuels) to minimize the risk of severe, high-intensity wildfires. Specifically, the goals of fuel treatments include the reduction of surface fuels, ladder fuels, and crown density (Agee and Skinner, 2005). The foundation of this objective is that the alteration of the fuel load will protect and sustain natural resources, particularly vegetation, wildlife habitat, and watershed integrity, increase the safety of wildland firefighters and people living in the WUI, and reduce suppression costs associated with high-intensity wildland fires (Busby 2002). Also, significant changes in forest structure and composition have contributed, in some areas, to decreased forest ecosystem integrity and resilience to disturbance (Harrod et al., 2007). As such, fuel treatments employ practices (i.e., thinning, prescribed fire) that restore structural and functional ecosystem components in dry temperate forests while simultaneously reducing risks to wildfire.

1.3 *Carbon and wildfire*

Forests in western North America sequester large quantities of atmospheric CO_2, and thus have the potential to offset anthropogenic emissions and mitigate effects of climate change. However, wildfire poses a significant and increasing risk of reversal to C stores in dry forest

ecosystems, undermining the permanence of C sequestration strategies. Wildfire releases substantial quantities of C to the atmosphere through combustion and the decay of fire-killed vegetation (Law et al., 2004; Campbell et al., 2007; Wiedinmyer and Neff, 2007; Meigs et al., 2009, Mitchell et al., 2009). Fire events are characterized by considerable spatial and temporal variability, and estimates of surface fuel consumption at coarse scales are often uncertain. Fuel treatments have been shown to reduce both tree mortality and emissions from wildfire, but heterogeneous burn environments, site-specific variability in post-fire ecosystem response, and uncertainty in future fire frequency and extent complicate assessments of long-term (decades to centuries) C dynamics across large landscapes. The capacity of forests to sequester and retain C is critical to the benefit of climate change mitigation strategies. Thus, assessing reversal risks to C stores (i.e., wildfire) is critical for accurate C accounting in dry temperate forests.

The magnitude of the effect of wildfire on net biome productivity depends on three major processes: direct combustion, decomposition, and re-growth of post-fire vegetation (Flannigan et al., 2009). CO_2, carbon monoxide, methane, particulate matter, and other GHGs are directly released to the atmosphere during wildfire. Wiedinmyer and Neff (2007) determined that C emissions from wildfire in the US are equivalent to 4 to 6% of annual anthropogenic emissions. However, over several decades, delayed mortality after wildfire and the decomposition from fire-killed trees may release up to three times the amount of C to the atmosphere than the fire itself (AuClair and Carter, 1993). Coarse fuels and other material not directly consumed by fire decompose at rates dependent on several environmental factors, including climate, soil microflora, disturbance severity, and quality of substrate (Agee 1993). Decomposition may also exceed assimilation of C by post-fire vegetation for decades following wildfire. Vegetation response to wildfire is characterized by enormous variability, but rapid re-colonization of post-

fire, non-tree vegetation has been shown to significantly affect net primary productivity (NPP) after wildfire and harvest (Law et al., 2004; Meigs et al., 2009). Vegetation life-form conversion (e.g., forestland to shrubland) following insufficient and unsuccessful tree regeneration following wildfire also have implications for the long-term C balance of dry forest ecosystems (Kashian et al., 2006; Deal et al., 2010; Dore et al., 2010; Hurteau and Brooks, 2011).

The frequency of wildfire events in dry forests of western North America has increased since the mid-20th century (NIFC, 2010). During this same time period, the total wildfire area burned (WFAB) has also increased (NIFC, 2010). As a result, forest managers are challenged to reduce potential fire hazard in dry forests through expanded and effective fuel treatment programs. Fuel reduction treatments in dry forests can potentially mitigate the spread and severity of wildfire (Agee and Skinner, 2005; Omi et al., 2006; Prichard et al., 2010), although the effects of fuel treatments on long-term (decades to centuries) C retention across large landscapes are equivocal (Finkral and Evans, 2008; Hurteau and North, 2009; Mitchell et al., 2009).

1.4 *Fuel treatments and initial carbon loss*

Mechanical thinning and surface fuel treatments (including prescribed fire) are commonly implemented to reduce fuel loadings and reduce the severity of wildfire and risk of crown fire (Agee and Skinner, 2005). However, fuel reduction treatments coincide with an initial net C loss to the treatment area. Thinning treatments reduce standing C stocks via whole-tree removal and also release C through the combustion of fuel in logging machinery, transportation of stems and logging residue, burning of slash, and decay of logging slash and wood products (Finkral and Evans, 2008; Stephens et al., 2009). Fossil fuel combustion associated with thinning

treatments is equivalent to approximately 0.5 - 3% of the total aboveground C stock (Finkral and Evans, 2008; Stephens et al., 2009; North et al., 2009; Hurteau and Brooks, 2011). Milling waste and emissions from prescribed fire are the largest source of C released in fuel treatment applications, and though highly variable, can each exceed 20% of the post-treatment aboveground C stock (North et al., 2009). Additional C costs vary considerably depending on the fate of the wood products, transport distance to the processing facility, and type of processing facility (Finkral and Evans, 2008, North et al., 2009; Oneil and Lippke, 2010).

To effectively maintain reduced fire hazard in a given forest, repetition of fuel treatments may be necessary (Peterson et al., 2005, Johnson et al., 2007) and thus must also be included in the analysis of C sequestration strategies. The temporal efficacy of fuel treatments is poorly documented and probably varies considerably across forest ecosystems (Agee and Skinner, 2005). Fuels accumulate on treatment areas, and fire hazard increases over time; after 10-15 years or more, treated areas can be overwhelmed by intense fires burning in adjacent areas (Agee and Skinner, 2005). Thus, multiple prescribed fires and mechanical thinning entries may be necessary to meet long-term fuels management objectives aimed at mitigating effects of future wildfires in dry forests.

1.4.1 *Emissions from equipment usage*

Forest operations produce C emissions by burning fossil fuels used to power vehicles and machinery. The overall amount of fossil fuel consumption for a given site depends on the intensity and frequency of silvicultural treatments (Markewitz 2006). Estimation of C emissions from equipment usage requires knowledge of the volume per hour efficiency of a machine (i.e., liters of fuel/hour), the number of hours per unit of area for a particular activity, and the C

emitted per volume of fuel consumed (Markewitz 2006). Data sources for equipment usage include logging contractor records, electronic activity recorders on logging equipment, and average consumption rates from company sources (e.g., Stihl [Virginia Beach, Virginia, USA], Caterpillar [Peoria, Illinois, USA]). Fuel consumption rates are converted to CO_2 emissions using the mean C contents of gasoline and diesel (EPA 2005) as well as the molecular mass ratio of CO_2 to C (Stephens et al., 2009).

Detailed summaries of treatment-related emissions in dry temperate forests are presented in four recent studies (Finkral and Evans, 2008; North et al., 2009; Stephens et al., 2009; Sorensen et al., 2011). Emissions released from equipment used to harvest, load, and transport logs during fuel reduction treatments are within a range of 0.05 – 1.20 Mg C ha^{-1}, approximately 0.5 -3.0% of total post-treatment C storage (Table 1).

Stephens et al. (2009) examined 12 different experimental treatment units ranging in size from 14 to 29 ha in mixed-conifer forest in the north-central Sierra Nevada. Units were replicated across four different treatment types (mechanical only, burn only, mechanical and burn, and control). Study units were comparable in terms of stand structure and species composition. Mechanical thinning was conducted as a moderate to heavy thin from below. Following harvest, approximately 90% of the remaining understory conifers and hardwoods were masticated using an excavator mounted with a rotary masticator (Stephens et al., 2009). Altogether, the authors reported that total equipment release were equivalent to less than 0.91 Mg C ha^{-1}, less than 1% of post-treatment C storage. The additional mastication treatment was not conducted in any other studies, and potentially explains why emissions from equipment usage surpassed estimates in comparable understory thin units from North et al (2009).

Six different fuel treatments (burn only, understory thin, understory thin and burn,

overstory thin, overstory thin and burn, control) were applied in a full-factorial experimental design to 18 permanent 4-ha fuel plots in a Sierra Nevada mixed-conifer forest (North et al., 2009). The authors utilized US Forest Service records to calculate emission contributions from logging machinery and personnel transport. This study reported the longest haul distance (235 km) for log transport to the nearest sawmill. Overall, equipment-related emissions (0.64 – 1.20 Mg C ha^{-1}) are comparable to those estimated by Stephens et al. (2009).

Equipment emissions from a 90-ha ponderosa pine restoration thinning treatment in northern Arizona (Finkral and Evans, 2008) were far less than treatments conducted in the Sierra Nevada, but are consistent with another study conducted in northern Arizona (Sorensen et al., 2011). Finkral and Evans (2008) report equipment emissions equivalent to 0.07 – 0.20 Mg C ha^{-1}, less than 1% of total post-treatment C storage. The authors suggest that flat ground, openness of the stand, and generally easy working conditions may explain their relatively low estimate of C equivalent used in logging.

Sorensen et al (2011) examined 5 different treatment units on federal and state forestland in northern Arizona. Silvicultural prescriptions include low thinning and crown thinning of varying intensities consistent with regional standards for fire hazard reduction. The authors worked with logging contractors to estimate diesel and gasoline consumption of all logging machinery used during the harvest, including chainsaws, skidders, feller bunchers, processors, and forwarders. The authors also tracked the fuel consumption associated with the employee and equipment transport to and from the study site. C equivalent emissions were found to range from 0.05 to 0.28 Mg C ha^{-1}, less than 1% of total post-treatment C storage and the smallest component of the overall C budget in the thinning operations.

Emissions from equipment usage during fuel treatments across all four studies amount to

a small percentage (0.5 - 3%) of the total aboveground C stock. There is far greater variability and magnitude in treatment-related C emissions from prescribed fire, harvested C (whole-tree removal), waste associated with milling operations, and the decay of wood products.

1.4.2 *Emissions from milling waste and decay of wood products*

Emissions from milling waste have been shown to be one of the largest sources of C emissions related to fuel treatment application (North et al., 2009). From the harvest of trees to the disposal of wood products generated from harvested trees, C is lost at each step of the processing chain due to the physical break down of wood (Ingerson 2009). Estimates from a sawmill in California suggest that approximately 60% of logs are converted to lumber, and the remaining 40% result in milling waste (North et al., 2009). This is consistent with published reports from Skog and Nicholson (2000), Ingerson (2009), Oneil and Lippke (2010), and data summarized by Stephens et al (2009). However, large fractions of milling waste in California are converted to energy at cogeneration plants (14 – 47%) or reconstituted as wood products (e.g., particleboard, hardboard, and fiberboard) with longer median life spans for C storage (Skog 2008; Stephens et al., 2009). Many reconstituted wood products sequester C for decades and comprise roughly 40% of milling waste in California (from Stephens et al. 2009). The manufacturing of reconstituted wood products, however, converts potential milling waste to longer lasting forest products at the cost of increased energy inputs and resins (Ingerson 2009). Milling waste may also be recycled (9%) or composted into mulch (8%) (Skog 2008). The majority of milling waste (67%) is sent to landfills, although anaerobic conditions coupled with high lignin concentrations in wood and paper make most wood products in landfills highly resistant to decay (Gower 2003; Skog 2008). Alternatively, milling waste is burned as fuel and

results in the immediate release of emissions to the atmosphere, but still provides substantial C benefits in energy substitution for emissions generated from fossil fuels (Chen et al., 2000; Perez-Garcia et al., 2006; Oneil and Lippke, 2010). Current trends indicate that less material is reaching landfills each year (Stephens et al. 2009), yet variation in regional market demands and differences in technology across processing facilities complicate comparisons of woody biomass utilization.

Forest harvesting is generally considered to result in a net release of C to the atmosphere without the inclusion of C stored in commercial timber and wood-fiber products (Houghton et al., 1983; Perez-Garcia et al., 2006). However, consideration of the end use of wood products is critical to the assessment of C budgets in managed forests, because decay rates and life spans of forest products vary substantially. Lumber used in home construction can store C for 50 – 100 years, whereas wood used to construct pallets has a median life span of 6 years (Skog and Nicholson, 1998). Wood material that is composted as mulch or used as landscaping bark has a median life span of 5 years (Table 1, Stephens et al., 2009). Paper and shipping materials, including cardboard and pallets, decay rapidly in comparison to solid wood (lumber) and reconstituted wood products (Skog and Nicholson, 1998; Finkral and Evans, 2008). In general, wood products comprise a significant C pool that offsets a fraction of the initial C costs of tree harvest. Finkral and Evans (2008) report that had wood removed from the site been used in longer-lasting products, the fuel treatment would have resulted in a net gain in C storage even after considering emissions projected from future wildfire. In the inland Northwest, Oneil and Lippke (2010) report that C storage in long-lived (>80 years) wood products exceeds simulated C losses from wildfire, even when accounting for a doubling of the historic fire rate as a result of climate change.

There is significant variation in published estimates of milling waste in studies conducted in northern Arizona and the Sierra Nevada (Finkral and Evans, 2008; North et al., 2009; Stephens et al., 2009). The highest magnitude of milling waste associated with thinning treatments (18.3 – 38.2 Mg C ha^{-1}) is reported by North et al (2009). In this study, material harvested during fuel treatment application was processed at the only remaining sawmill in the southern Sierra Nevada, 235 km from the study site. Communication from mill personnel, reported by North et al (2009), indicates that approximately 40% of logs delivered to the mill result in waste (i.e., chips, bark, and sawdust). Milling waste generated at this sawmill is primarily burned as fuel for electricity (cogeneration) and, to a lesser extent, sold as landscaping bark or mulch (North et al., 2009). The authors considered all milling waste as direct emissions to the atmosphere. Bark and mulch by-products generally decompose within 5 years (Stephens et al., 2009) and burned fuel is released immediately to the atmosphere as emissions. Overstory thinning treatments are unique to North et al (2009) and estimates far exceed waste generated from other fuel treatments and are a function of a higher volume of logs processed per treatment unit.

Stephens et al (2009) report milling waste from fuel treatments conducted in the central Sierra Nevada. Three different sawmills received logs generated from this study. Emissions from milling waste were estimated from a coefficient derived from mill operations in the Pacific Northwest, 0.05 Mg CO_2 m^{-3}, and applied to actual harvest volumes from the treatments. A conversion factor (3.67:1) was used to convert C dioxide equivalent to C. Fractions of log mass were converted to various wood products from sawmill surveys in California, Oregon, and Washington (Morgan et al., 2004; Milota et al., 2005; Stephens et al., 2009). The decay rates for the wood products were then estimated from national historic use data (Winjum et al., 1998;

McKeever 2002; Winistorfer et al., 2005; Skog 2008). Emissions from milling waste were reported as the sum of biomass burned as fuel and biomass stored in wood products with decay rates of less than one year (0.297 – 0.372 Mg C ha $^{-1}$).

Several adjustments to this estimate must be made to directly compare results presented by North et al (2009). According to the published sawmill surveys summarized by Stephens et al. (2009) approximately 6% of milling waste is stored in wood products with decay rates of less than 5 years, the cut-off used in North et al (2009). This is equivalent to an additional 0.017 – 0.022 Mg C ha $^{-1}$. Also, in the thin only treatments, emissions from the decay of masticated woody material left onsite (7.83 Mg C ha $^{-1}$) may contribute additional emissions to those study sites if assumed to decay in less than 5 years. In the thin-and-burn treatments, prescribed fire emissions included emissions from the consumption of masticated material, so adjustment to the published estimate is unnecessary. Altogether, adjusted total emissions from milling waste for thin only treatments (8.14 Mg C ha $^{-1}$) is still far less than estimates reported by North et al (2009) (18.3 Mg C ha $^{-1}$) in comparable treatments. This may reflect differences between facilities in milling efficiency or the ability to generate reconstituted wood products. It also reflects potentially substantial differences in the quantity, size distribution, and species of wood harvested between study sites. As a proxy for harvest volume in an understory thinning treatment, North et al (2009) report that approximately 27.5 Mg C ha^{-1} is stored in lumber, whereas Stephens et al (2009) report an average lumber storage of 15.2 Mg C ha^{-1} across treatments designated for thinning from below. Moreover, methods for measuring decay rates for C in wood products vary considerably (Lim et al., 1999).

In northern Arizona, Finkral and Evans (2008) report milling waste for three different wood-utilization scenarios. Wood removed during a restoration thinning treatment was primarily

used for firewood and assumed to release emissions to the atmosphere (0.02 – 0.03 Mg C ha $^{-1}$) within one year of harvest. The authors offer plausible alternative wood-utilization scenarios based on diameter requirements for processing construction materials, pallets, and paper. Harvested logs processed as paper released the most emissions associated with milling waste (2.47 – 4.94 Mg C ha $^{-1}$). Logs processed as construction materials or pallets generated approximately (0.57 – 0.85 Mg C ha $^{-1}$) emissions from milling waste. The C benefits from the substitution of home heating from firewood emissions in place of gas or oil heat is also summarized.

Ager et al (2010) used a landscape simulation model to estimate the effects of fuel treatments on C pools, including C emissions associated with milling waste. Approximately, 6 229 ha (9.1%) of a 68 458 ha watershed in southern Oregon met the criteria for simulated thinning from below. The authors used the Forest Vegetation Simulator (FVS) (Dixon 2008) to estimate wood generated from harvest activities, but do not list references for an assumption of 57% milling efficiency. This rate was calculated from reported values of merchantable material stored in long-lasting wood products (50 467 Mg C) and merchantable material removed but not stored (37 996 Mg C). Milling efficiency in empirical studies was reported at 60% (North et al., 2009), and 53% (Stephens et al., 2009).

C stored in lumber products exceeded C emissions from milling waste in all treatments in all studies (Table 2), yet in all cases, the two are linked because milling waste is directly proportional to the volume of processed material. Emissions from milling waste in understory thinning, thinning from below, and restoration thinning associated with fuel reduction treatments were shown to range from < 0.1 to 13.6% of post-treatment C stores across all studies, whereas overstory thinning units ranged from 25.5 to 34.0% (Table 1). In many cases, emissions from

milling waste exceeded emissions from prescribed fire and thus constitute a significant fraction of initial C losses associated with fuel treatments.

1.4.3 *Emissions from prescribed fire: Empirical results*

There is agreement across both observed and simulated treatments that prescribed fire constitutes a substantial magnitude of overall treatment emissions. Prescribed fire is effective at reducing surface fuels and horizontal fuel continuity (van Wagtendonk 1996; Graham et al., 2004) and is the only treatment available to successfully reduce fine surface fuels. However, prescribed fire is not a discriminating tool for reducing tree density, crown density, or fuel ladders and is commonly used in combination with thinning to achieve management goals (Gorte 2009). Prescribed fire may consume substantial surface biomass, and the smoldering consumption of the organic layer of the soil is the largest contributor to smoke production with significant impacts on soil nutrient cycling (Neary et al., 1999). Prescribed fire also generates fuels by killing understory trees (Agee 2003), and multiple treatments may be necessary to maintain reduced fire hazard over time (decades). Recent studies have shown that emissions from prescribed fire make significant contributions to the C budget of fuel treatments (Finkral and Evans, 2008; North et al., 2009; Stephens et al., 2009; Sorensen et al., 2011) (Table 3; Figure 1).

In northern Arizona, analyses of prescribed fire emissions focus on the burning of slash piles created from residual biomass left on site (sub-merchantable material) following harvest. Finkral and Evans (2008) and Sorensen et al (2011) measured the dimensions of slash piles and calculated slash biomass from volumes using equations from Hardy (1996). To estimate C mass in the piles, Finkral and Evans (2008) developed size:weight ratios by using platform scales to

weigh sample piles. Results from 5 fuel reduction treatments indicate a range of prescribed fire emissions from 1.01 to 9.58 Mg C ha^{-1} (Finkral and Evans, 2008; Sorensen et al., 2011) or 1.1 – 11.0% of the post-treatment C stores (Table 3; Figure 1).

Prescribed fires analyzed in the Sierra Nevada were implemented as broadcast burns, either backing fires or strip head-fires (Stephens et al., 2009). Stephens et al (2009) calculated emissions from prescribed fire as the difference between pre- and post-burn fuel loads measured on site according to methods described by Clinton et al (2006). Combustion efficiency was calculated as grams of CO_2 per kilogram of fuel consumed according to equations developed by Ward and Hardy (1991). North et al (2009) also calculated the difference between pre- and post-burn fuel loads to estimate prescribed fire emissions. Results from a range of prescribed fire treatments (burn only, understory thin and burn, overstory thin and burn) indicate a range of emissions from 14.8 to 34.5 Mg C ha^{-1} (North et al., 2009; Stephens et al., 2009) or 6.1 – 29.2% of the post-treatment C stores (Table 3; Figure 1).

Differences in climate, fuels, topography, and management practices cause emissions from prescribed fire to vary considerably across the US, spatially and interannually (Liu 2004). Thus, the presence of substantial variation in prescribed fire emissions across studies and burn types in Arizona and California is not unexpected. Variation may also reflect the lack of research conducted on this topic. However, the magnitude of emissions in each treatment constitutes a significant percentage of total aboveground C storage. Also, emissions from broadcast burns measured by North et al (2009) and Stephens et al (2009) are much higher than emissions generated from the burning of slash piles reported by Finkral and Evans (2008). Slash burns more efficiently when piled, with significantly less particulate matter produced per unit mass of fuel consumed, than broadcast burns of the same material (Ward et al., 1998; Hardy 1996).

1.4.4 *Emissions from prescribed fire: Modeled results*

Simulation models, such as the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS), are commonly used to track fuel dynamics and potential fire behavior over time in the context of stand development and management (Reinhardt et al. 2008). Five recent studies have used FFE-FVS to estimate emissions from simulated prescribed fire (Hurteau and North, 2009; Ager et al., 2010; Reinhardt and Holsinger, 2010; Sorensen et al., 2011). Simulated results of prescribed fire emissions are consistent with the range of results derived from empirical analyses, confirming the relative magnitude of prescribed fire contributions to total treatment emissions.

Hurteau and North (2009) modeled the effects of 8 different fuel treatments on aboveground C storage for 100 years in the southern Sierra Nevada. The authors used FFE-FVS to simulate prescribed fire at 20-year intervals to match the historic fire regime for Sierran mixed conifer forests (McKelvey and Busse, 1996; North et al., 2005). Burning conditions were calibrated at the study site to a 2001 prescribed fire considered representative of common autumn burning conditions for the study area. Results across all treatments in the simulation period suggest that individual prescribed fire events emit 4.1 – 16.3 Mg C ha^{-1}, equivalent to less than 6% of the total post-treatment C storage in all simulations. However, the sum of the 5 prescribed fire applications in the 100-year simulation period substantially exceeded simulated wildfire emissions for each treatment (Table 4).

Reinhardt and Holsinger (2010) used FFE-FVS to simulate effects of fuel treatments on 140 stands representing seven habitat types in the northern Rocky Mountains. Stand-level data from the Forest Inventory and Analysis (FIA) program (http://www.fia.fs.fed.us) were used as model inputs. In habitat types in which prescribed fire is commonly used as a management tool

(e.g., warm-dry ponderosa pine) prescribed burning was simulated the year following thinning during "moderate" burning conditions defined by Reinhardt and Holsinger (2010). An alternative burn-only treatment was also simulated for this habitat type. Results from the warm-dry ponderosa pine habitat type indicate a range of emissions (6.8 – 12.7 Mg C ha^{-1}) from prescribed fire, equivalent to 13.2 – 36.0% of the total post-treatment C stores. For context, simulated thinning treatments for this habitat type removed 10.6 Mg C ha^{-1}, suggesting prescribed fire emissions exceed the amount of C removed during harvest (Table 3; Figures 1 – 2).

Sorensen et al (2011) used FFE-FVS to simulate prescribed fire over 100 years at two treatment frequencies (10-year intervals, 20-year intervals) in 5 ponderosa pine stands in northern Arizona. Burning conditions were calibrated to a prescribed fire in a similar and adjacent stand measured in McHugh and Kolb (2003). Results indicate that each simulated prescribed fire event emits 1 – 10 Mg C ha^{-1}, equivalent to approximately 2 – 16% of the total post-treatment C stores. In all treatments, the sum of the 5 prescribed fire applications in the 100-year simulation period exceeded simulated wildfire emissions (Table 4).

Ager et al (2010) simulated prescribed fire emissions with FFE-FVS in a watershed dominated by ponderosa pine in southern Oregon. Their simulation approach included a design of 94 individual treatment polygons, each approximately 71 ha in area. Simulated fuel reduction treatments consist of a 3-year sequence of thinning from below, site removal of surface fuels, and under burning. Results indicate prescribed fire emissions equivalent to 19.5 Mg C ha^{-1}, approximately 36% of total emissions associated with the simulated treatment.

Overall, results suggest that only live C removed by thinning exceeds the contribution of prescribed fire to the total emissions of fuel treatments (Tables 1-3). Thinning-only treatments have consistently been shown to be ineffective at reducing potential wildfire behavior (Stephens

et al., 2009; Prichard et al., 2010), and thinning alone is unlikely to meet ecological objectives of restoration without prescribed fire (Weatherspoon 1996; Weatherspoon and Skinner, 2002). As a result, the combination of thinning and prescribed fire has emerged as the preferred fuels management strategy in dry temperate forests and provides the maximum protection from future high-intensity fires (Peterson et al., 2005). Consequently, the magnitude of prescribed fire emissions is a critical consideration for the C balance of fuel treatments.

1.5 Thinning and wildfire effects on C flux

Full accounting of total ecosystem C storage requires the quantification of each component of the C cycle, but complexity in atmosphere-terrestrial ecosystem interactions complicates precise measurements of C flux in time and space (Hutley et al., 2005). Traditional plot-scale biometric inventories indirectly estimate net primary productivity (NPP) at relatively coarse temporal resolutions and rely on allometric relationships to scale up measurements (Baldocchi 2003). However, eddy covariance methods measure the near-continuous exchange of CO_2 across the canopy-atmosphere interface at time scales ranging from sub-hourly to years (Wofsy et al., 1993; Baldocchi 2003). Micro-meteorological measurements associated with eddy covariance data determine environmental controls on biogeochemical fluxes at the ecosystem scale (Baldocchi 2003; Grace 2004). Despite increased resolution and precision available in eddy covariance techniques, biometry is still useful to understand components and processes of net C fluxes, especially in the differentiation of contributions to C flux between individual species or sources of respiration (Dore et al., 2010). Moreover, biometric and eddy covariance measurements work together to develop and refine process-based models that may be used to assess larger scale impacts on the C cycle (i.e., regional, continental, global) (Law et al., 2004).

Net ecosystem exchange (NEE) is a measure of the net exchange of C between an ecosystem and the atmosphere (per unit ground area) and is a primary gauge of ecosystem C sink strength (Kramer et al., 2002). Biometric methods use plot-based measurements and allometric equations to estimate NEE at relatively coarse temporal resolutions by calculating the balance between NPP and heterotrophic respiration. Eddy covariance techniques directly measure exchanges of energy, water, and CO_2 at fine resolutions and are able to uniquely monitor responses in ecosystem physiology to various environmental factors, such as thinning and disturbance (Baldocchi 2003; Law et al., 2003).

Empirical data and modeling results indicate that stand-scale CO_2 exchange is strongly dependent on stand age and time since disturbance (Thornton et al., 2002; Law et al., 2003; Pregitzer and Euskirchen, 2004). Wildfire initially reduces net ecosystem production (NEP) because decomposition of detrital pools produced from the disturbance causes respiration to exceed NPP (Dore et al., 2008). Several recent studies have used eddy covariance techniques and biometric methods to explore the effects of forest management and disturbance on ecosystem C flux in dry temperate forests (Law et al., 2004; Kaye et al., 2005; Misson et al., 2005; Irvine et al., 2007; Dore et al., 2008; Campbell et al., 2009; Dore et al., 2010). We summarize key results that help to refine the level of detail in ecosystem C exchange as it relates to fuels management and fire hazard reduction in dry temperate forests.

Law et al (2004) used a spatially nested hierarchy of observations coupled with a biogeochemical process model (Biome-BGC) to determine the relative influence of climate and disturbance on C stocks and fluxes in 4 western Oregon ecoregions. Observations range from detailed eddy covariance measurements to inventory data and remote sensing imagery. The authors developed a C budget for the study area by constructing a 5-year mean NEP for each 25-

m grid cell. Results indicate that total NEP for the study area was 13.8 Tg C yr^{-1}. The net effect of NEP, harvest removals, and wildfire emissions suggest that the study area was a net C sink (8.2 Tg C yr^{-1}). Harvest statistics from the Oregon Department of Forestry were used to estimate C removed in harvests during the sample period (5.5 Tg C yr^{-1}). Harvest removals were disproportionately from the Coast Range ecoregion, which is heavily managed in even-aged systems for timber production. A change detection analysis from remotely sensed imagery (1995-2000) was used to estimate averaged area burned by wildfire (1116 ha yr^{-1}). Contributions of wildfire emissions (1995-2000) to the regional C flux were small (0.1 Tg C yr^{-1}), yet the 2002 Biscuit Fire (200 000 ha) emitted approximately 4.1 Tg C (27.3 Mg C ha^{-1}), suggesting that large wildfires have significant effects on regional net C balance, but still do not exceed the contributions of the dominant regional disturbance (commercial forestry).

Irvine et al (2007) used a combination of biometric surveys, allometric relationships, biomass decay constants, and soil respiration assessments to measure C fluxes in the eastern Cascade Range (Oregon) after a mixed-severity wildfire to determine controls on NEP. The authors identified a trend of decreasing NPP with increasing burn severity, resulting in significantly lower NEP in burned stands than unburned stands. However, the authors found no significant change in heterotrophic respiration 2 years following wildfire regardless of substantial increases in detrital pools within burned stands. Results suggest that decay rates in detrital pools may potentially determine C recovery in stands burned by wildfire.

Dore et al (2008) used eddy covariance techniques to measure CO_2 exchange at two forest sites in northern Arizona, one burned by a high-intensity, stand-replacing wildfire in 1996 and one unburned site. The wildfire altered abiotic (e.g., soil temperature) and biotic (e.g., biomass, leaf area index) site characteristics, as well as monthly and annual C budgets. The fire

reduced annual ecosystem respiration by 30%, but significant reductions in gross primary production (60 percent) resulted in a decrease in net C exchange. Ten years after the fire, the burned site was still a moderate source of CO_2 to the atmosphere and may persist as a source of CO_2 for years due to slow vegetation recovery and poor tree establishment after the fire. In comparison, the nearby unburned site was a moderate C sink. The authors also used eddy covariance data to assess environmental controls (soil water content, vapor pressure deficit, soil temperature, air temperature) on daytime and nighttime net ecosystem exchange. Strong seasonal variability in C fluxes was captured in the C flux measurements, but trends consistently show the lasting effects of high-intensity wildfire on net ecosystem exchange.

Kaye et al (2005) used biometric surveys, clip plots, allometric equations, and radial growth analyses to measure initial biogeochemical responses to restoration thinning treatments. C, nitrogen, and phosphorous fluxes were assessed for two years following restoration thinning and composite restoration (thinning, forest floor manipulations, and prescribed fire) in comparison to untreated stands. In general, restored stands had similar plant C, nitrogen, and phosphorous cycling rates as untreated stands. Short-term restoration effects on NPP were not detected, and treatment effects on biological C fluxes were small (statistically zero) in comparison to C removed during restoration thinning (17.7 Mg C ha^{-1}) and thin-and-burn treatments (33.7 Mg C ha^{-1}). The authors suggest that the fate of thinned tree biomass and the intensity of prescribed fires remain the most important factors controlling the initial impact of restoration on C budgets.

Dore et al (2010) used a combination of biometric surveys, soil efflux measurements, and eddy covariance techniques to assess the effects of different disturbances (thinning and wildfire) on C and water exchange between the biosphere and atmosphere. Results indicate that high-

intensity wildfire has a larger influence on C balance than thinning. Wildfire was shown to reduce total ecosystem C stocks by approximately 40%, thinning reduced C stocks by 14% and increased the growth rate for residual unthinned trees, consistent with other studies in the southwestern US (Kaye et al., 2005; McDowell et al., 2006). Both disturbances reduced gross primary production (55% by wildfire, 30% by thinning) more than ecosystem respiration (33-47% by wildfire, 18% by thinning), resulting in decreases in overall C uptake. The authors detail the importance of including both high-intensity disturbance (wildfire) and partial disturbance (thinning) in estimating regional C budgets.

Several studies used biometric methods to assess the effects of fuel treatments and wildfire on C pools (Finkral and Evans, 2008; Hurteau and North, 2009; North et al., 2009; Stephens et al., 2009). Relatively few studies measured C flux (Kaye et al., 2005; Irvine et al., 2007; Campbell et al., 2009) and even fewer directly measured total ecosystem C exchange via eddy covariance techniques (Law et al., 2004; Dore et al., 2008; Dore et al., 2010). C flux studies increase the level of detail in C sequestration analyses by capturing fine-scale processes affecting ecosystem C exchange. For example, results from C flux studies indicate strong seasonal and interannual variability in C exchange (Wirth et al., 2002; Misson et al., 2005), potentially buffered effects of ecosystem respiration by strong compensatory responses of pioneer vegetation (Law et al., 2004), and lasting effects of high-intensity disturbance on NEE (Dore et al., 2010).

Disturbance is a key driver affecting the C cycle in dry temperate forests and has significant short-term effects on the C balance of a given forest (IPCC 2007). However, long-term effects of disturbance on C dynamics are more uncertain and are influenced by a wide range of factors including hydrology, soil temperature, heterotrophic respiration, erosion of soil organic

C, and rates of vegetation recovery (Dore et al., 2008). C flux measurements provide a detailed "snap shot" of the C cycle at a given location and are excellent for exploring short-term controls on C exchange, including seasonal and interannual variability (Misson et al., 2005). Plot-scale chronosequence analyses and associated space-for-time substitutions expand the temporal extent of disturbance-C dynamics, but are limited by key assumptions of constancy in biogeochemical cycles across multiple disturbances and potentially divergent successional pathways in the context of climate change (Johnson and Miyanishi, 2008; Walker et al., 2010). Attempts to reduce uncertainty in ecosystem C exchange observations include data assimilation techniques designed to combine plot-scale data and flux observations into process-based models (Kramer et al., 2002; Rayner et al., 2005; Williams et al., 2004). More recently, new methods for upscaling C and water fluxes to regional and continental scales have emerged with advances in the integration of eddy covariance measurements and remotely sensed data from satellite observations (Wylie et al., 2007; Hollifield-Collins et al., 2008; Xiao et al., 2008; Xiao et al., 2010; Xiao et al., 2011). However, despite significant advances in modeling and the spatial representation of ecosystem C exchange, uncertainty is still largest during extreme climatic events and large, severe disturbances (Xiao et al., 2010; Xiao et al., 2011). Therefore, a better understanding of contributions to C budgets from disturbance is critical because the frequency, severity, and extent of natural disturbances are expected to increase with global climate change (McKenzie et al., 2004, Westerling et al., 2006).

1.6 Fuel treatments and reduced emissions from wildfire

The potential trade-off to initial net C losses associated with fuel reduction treatments is a decreased risk of future high-severity wildfire and its associated release of C to the atmosphere

(Hurteau et al., 2008). In dry temperate forests, fuel treatments that successfully reduce surface fuels have been shown to mitigate the spread and severity of wildfire (Fulé et al., 2001; Pollett and Omi, 2002; Skinner et al., 2004; Peterson et al., 2005; Omi et al., 2006; Safford et al., 2009; Stephens et al., 2009; Prichard et al., 2010). Some recent studies use results from wildfire simulations to suggest that in addition to reducing fire severity, fuel treatments may also reduce emissions from wildfire (Finkral and Evans, 2008; Hurteau et al., 2008; Hurteau and North, 2009; Stephens et al., 2009; Reinhardt and Holsinger, 2010; Sorensen et al., 2011) (Table 5; Figure 3). However, wildfire simulations in other studies suggest that fuel treatments are unlikely to benefit C storage and may result in a reduction of overall C stocks (Mitchell et al., 2009; Ager et al., 2010; Campbell et al., 2011). Few empirical studies examine C emissions from study areas actually burned by wildfire (Campbell et al., 2007; Meigs et al., 2009; North and Hurteau, 2011), and only one reports wildfire interactions in treated and untreated stands (North and Hurteau, 2011). We synthesize findings from both simulated and observed effects of fuel treatments on wildfire emissions and compare expected C benefits from fuel treatments to previously discussed treatment costs.

Finkral and Evans (2008) use FFE-FVS to estimate the increase in crowning index (i.e., the wind speed necessary to maintain crown fire) after a restoration thinning treatment. The authors simulated a wildfire, pre- and post-treatment, under hazardous conditions (22 m s^{-1} winds, 29 °C temperature, and "low" moisture content [<10%] of surface fuels). Results suggest that the thinned stand reduces the likelihood of active crown fire, even in extreme fire conditions, and releases less C (2.41 Mg C ha^{-1}) to the atmosphere than the unthinned stand. The authors also discuss three different wood utilization scenarios for C harvested during mechanical thinning (e.g., firewood, paper, pallets/construction materials). Their findings suggest that the

utilization of long-lasting wood products may provide a greater C benefit to the system than the expected magnitude of reduced wildfire emissions.

Hurteau and North (2009) compare the response of 6 different fuel treatments to wildfire by simulating extreme wildfire conditions using FFE-FVS (17.9 m s^{-1} winds, 32.2 °C temperature, and "very low" moisture content of surface fuels). One wildfire event is simulated for each treatment in the year 2050. Wildfire emissions were the highest in the untreated stand (36.3 Mg C ha^{-1}), and fuel treatments that included prescribed fire resulted in lower wildfire emissions than thinning-only treatments (Table 4). Overall, model results suggest that fuel treatments release less C (12.7 – 26.3 Mg C ha $^{-1}$) to the atmosphere than an untreated stand.

Reinhardt and Holsinger (2010) used FFE-FVS to simulate effects of fuel treatments on 140 stands representing seven major habitat types in the northern Rocky Mountains. The authors found that more C was released at the time of wildfire from untreated stands than treated stands in all habitat types. However, to sharpen comparisons across all studies, focus here is restricted to the authors' discussion of warm, dry ponderosa pine habitat (Fishcher and Bradley, 1987). Wildfire was simulated 5 years after the implementation of alternative fuel treatments (e.g., thin and burn, burn only). Wildfire conditions were designed to represent dry, late summer wildfires (8.9 m s^{-1} winds, 21 °C temperature, and "low" moisture content [< 10%] of surface fuels). The mean reduction in wildfire emissions was found to be significant between mechanically treated stands and untreated stands (5.04 Mg C ha $^{-1}$), but not in burn only stands (2.93 Mg C ha $^{-1}$). Results indicate that fuel treatments decrease fire severity, crown fire occurrence, and tree mortality from wildfire while reducing subsequent wildfire emissions.

Stephens et al (2009) used Fuels Management Analyst Plus (FMA) to estimate tree mortality from simulated wildfire under extreme burning conditions (97.5th percentile). In the

study area, approximately 70% of total aboveground C was contained in the live-tree C pool. In untreated stands, approximately 90% of live-tree C was determined to be at a high risk (>75% mortality) during severe wildfire. The authors report that untreated stands contain approximately 145 Mg C ha $^{-1}$ of live-tree C at high risk for wildfire mortality. Thin-only treatments are reported to have 18 Mg C ha $^{-1}$ of live-tree C at high risk for mortality, while thin-and-burn treatments reduce the value to approximately 4 Mg C ha $^{-1}$. Results suggest that fuel treatments effectively reduce tree mortality from wildfire. Specifically, treatments that combine moderate to heavy thinning from below with prescribed fire are most effective at protecting live tree C from direct mortality caused by wildfire. However, although tree mortality is an important metric for assessing stand resilience to disturbance, most C stored in forest biomass (stem wood, branches, coarse woody debris) usually remains unconsumed even by high-severity wildfires. The combustion of surface and ground fuels is the largest proportion of emissions from wildfires across the spectrum of burn severity (Campbell et al., 2007; Meigs et al., 2009). As a result, tree mortality may be an incomplete metric for estimating C emissions from wildfire.

Meigs et al (2009) report pyrogenic emissions from multiple large wildfires in the Metolius watershed in eastern Oregon. The authors used an FIA sampling design enhanced for C metrics to capture post-fire plot-scale data. They also used clip plots to assess dominant understory vegetation. Estimates of wildfire consumption were derived with the simulation program Consume 3.0 (http://www.fs.fed.us/pnw/fera/research/smoke/consume/index.shtml) augmented with local data. Ponderosa pine stands within the burned area released an average of 19.7 – 30.2 Mg C ha $^{-1}$ in emissions. Grand fir – Douglas-fir stands released an average of 16.6 – 32.3 Mg C ha $^{-1}$ (Table 6). Wildfire emissions increased monotonically with increasing burn severity. Results also suggest that the percent of biomass consumed in ponderosa pine stands was

substantially higher than in adjacent grand fir-Douglas-fir stands. Fuel treatments were not present in the study area, yet results from Meigs et al (2009) provide context, via empirical evidence across the range of burn severity, for pyrogenic emissions in ponderosa pine. Estimates are consistent with results from Campbell et al (2007), in which pyrogenic C emissions from a mixed conifer forest in the Biscuit Fire range from 12.4 to 28.6 Mg C ha^{-1}.

Mitchell et al (2009) used a forest ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on fire severity and the resulting long-term C dynamics in three Pacific Northwest ecosystems: east Cascade Range ponderosa pine forest, west Cascade Range western hemlock forest, and Coast Range western hemlock-Sitka spruce forest. STANDCARB is a forest ecosystem simulation model that integrates climate-driven growth and decomposition processes with species-specific rates of senescence, mortality, and competition that characterize forest gap dynamics. Simulation results indicate that fuel reduction treatments in each ecosystem consistently reduced fire severity. However, the authors report that reducing the fraction by which C is lost to wildfire requires the removal of a much greater amount of C, because most of the C stored in forest biomass (i.e., stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. Thus, fuel reduction treatments simulated in this experiment resulted in reduced mean stand C storage.

Sorensen et al (2011) quantified the effects of fuel treatments and wildfire on the C budget of 5 ponderosa pine stands in northern Arizona. The authors used FFE-FVS to simulate long-term effects of wildfire, thinning, and repeated prescribed fire on C storage. Multiple scenarios were considered in the 100-year simulation. High-intensity wildfire was simulated to occur once within 100 years (HF100), and once within the next 50 years in another scenario (HF50). To account for temporal randomness in ignition, wildfire simulations were conducted

once during each decade of the simulation period, and final results were averaged. In both scenarios, hazardous wildfire conditions were modeled in untreated stands with FFE-FVS (22 m s^{-1} winds, 29 °C temperature, and "very dry" surface fuels). Results indicate that wildfire emissions range from 17.0 to 24.4 Mg C ha^{-1} in the HF100 scenario and 13.9 to 21.5 Mg C ha^{-1} in the HF50 scenario. The authors also generated scenarios for two frequencies of prescribed fire applied every 10 years and 20 years for the 100-year simulation period. Cumulative prescribed fire emissions in both scenarios were higher than the one-time wildfire emissions in both HF100 and HF50 scenarios (Table 4).

North et al (2009) used FFE-FVS to calculate fire behavior (i.e., crown bulk density, torching index, crowning index), but did not estimate emissions from wildfire. The authors simulated 95th percentile burning conditions for the study area using Fire Family Plus software (Main et al., 1990). The torching index indicates the wind speed (at 6 m above surface) at which a surface fire can be expected to ignite the crown layer. All 5 treatments, relative to the control, increased torching and crowning indices and substantially decreased canopy bulk density to values generally proportional to treatment intensity.

Ager et al (2010) modeled wildfire emissions and large-scale effects of fuel treatments on fire spread and intensity using a landscape risk analysis approach. A probabilistic framework for wildfire occurrence coupled with a fire spread algorithm from FlamMap (Finney 2002) was used to simulate 30 000 burn periods at random ignition locations within a watershed in southern Oregon dominated by ponderosa pine. Fire growth calculations were generated at 90 x 90 m grids, and parameters (temperature, fuel moisture) for wildfire simulations were derived from 97th percentile August fire weather conditions described by local weather stations. Wind speed was 11.1 m s^{-1} in all simulations and wind direction was randomly simulated across three

azimuths representing dominant wind patterns in the area. Results indicate a decrease in average burn probability (treated: 0.017; untreated: 0.021) across all forested areas. Estimates of wildfire emissions were generated from FFE-FVS in treatment and control (no treatment) scenarios. Stands were selected for simulated fuel treatment based on several criteria from the Fremont-Winema National Forest and comprise 7 180 ha of 45 192 forested acres in the study watershed (~15%). In the areas designated for fuel treatment (7 180 ha), emissions from wildfire are 0.02 Mg C ha $^{-1}$ (treatment scenario) and 1.38 Mg C ha $^{-1}$ (non-treatment scenario). The relative effectiveness of fuel treatments in mitigating C emissions is substantial, but the magnitude of emissions from wildfire is only 3.3% of all emissions associated with the treatment scenario. Prescribed fire (39.9%), removal of non-merchantable material (22.4%), milling waste (14.9%), and mastication (19.5%) each constitute a much higher percentage of treatment-related emissions.

Campbell et al (2007) determined combustion factors within the Biscuit Fire in 25 different C pools, and variation was assessed as a function of remotely sensed burn severity data. To estimate pyrogenic emissions (stratified by burn severity) the authors combined the combustion factors with pre-burn fuel densities derived from FIA plots (n=180) and species-specific allometric equations. The combustion factor increases monotonically in virtually all of the 25 designated C pools as burn severity increases. Total pyrogenic emissions for the Biscuit Fire were estimated at 3.5 – 4.4 Tg C (12.4 to 28.6 Mg C ha $^{-1}$, depending on severity). Approximately 60% of all emissions result from the combustion of litter, foliage, and small downed wood. However, the authors suggest that much of the surface fuels could have been destined for biogenic emission (decay) with or without wildfire because they have a relatively short residence time (10-20 years).

North and Hurteau (2011) used empirical data to quantify wildfire emissions and short-term changes in C stocks in treated and untreated stands burned by wildfire. Twelve wildfires in California (with 19 associated fuel treatments) were selected for the study, mostly in the central and northern Sierra Nevada. To calculate C at each site, genus-specific allometric equations were applied to field data (Jenkins et al., 2004) for each of the three stand conditions (pre-treatment/pre-wildfire, untreated/burned, treated/burned). Averaged emissions from wildfire in treated stands (29.7 Mg C ha $^{-1}$) were substantially lower than average emissions from untreated stands (67.8 Mg C ha $^{-1}$). However, when live C removed during fuel treatments (50.3 Mg C ha $^{-1}$) is added to emissions from wildfire, the mean net C loss is higher in treated stands (80.2 Mg C ha $^{-1}$) than untreated stands (67.8 Mg C ha $^{-1}$). Tree mortality was significantly different in treated and untreated stands (53% and 97% respectively), as was the survivorship of large, C-rich trees (< 50 cm d.b.h.) (87% and 6%, respectively). Overall, roughly 70% of total ecosystem C in untreated stands was transferred to decomposing C pools (snags, surface fuels) with potentially significant effects on the long-term C balance of the study area.

Campbell et al (2011) used empirical data from various semi-arid conifer forests in combination with STANDCARB to assess how fuel treatments and wildfires affect aboveground C at multiple scales. Results suggest that the protection of one unit of C from wildfire combustion comes at the cost of removing three units of C in fuel treatments. The authors also used STANDCARB to explore a range of fire-return intervals (10, 50, and 250 years) for a ponderosa pine forest in eastern Oregon. Results indicate that C stores are more stable (i.e., less variation in total C) in a high-frequency, low-severity fire regime, but that a low-frequency, high-severity fire regime creates a higher long-term average of total forest C (simulation period = 500 years). The authors suggest that the fractional combustion of ecosystem C in wildfire events

is more constrained than fire frequency. Specifically, the authors report a modest increase in the long-term average fractional combustion of ecosystem C (9%) between high- and low-frequency fire regimes. Meanwhile, the mean number of simulated wildfire events varies by a factor of 25 between simulated disturbance regimes (n=50, n=2), suggesting that fractional combustion per wildfire event would have to far exceed 100% (i.e., violate the conservation of mass) in order to equal the combustion achieved in the high-frequency disturbance regime. As such, the authors conclude that fuel reduction treatments result in a net reduction in long-term C storage in all simulated disturbance regimes.

1.7 *Conclusion*

The extent to which fuel treatments benefit long-term C sequestration is a function of the cumulative difference between initial net C losses associated with fuel reduction treatments and wildfire emissions and their associated release of C to the atmosphere (Hurteau et al., 2008). This review discusses the magnitude of initial C losses to fuel treatments via whole tree removal (thinning), equipment usage, milling waste, and the consumption of surface fuels during prescribed fire (Hurteau et al., 2008; Finkral and Evans, 2008; North et al., 2009). We also discuss various methods for measuring wildfire emissions and fuel treatment costs, including biometric measurements, eddy covariance techniques, and model simulations.

Results summarized in this review assess the capacity of managed forests to sequester and retain C in dry temperate forests. There is unequivocal agreement across all studies that untreated stands release more emissions to the atmosphere during wildfire than treated stands, and that emissions increase monotonically as burn severity increases. However, most studies in this review include assumptions of future wildfire frequency and probability that skew long-term

trade-off analyses by overestimating the ability of fuel treatments to reduce wildfire emissions over long time scales. For example, fuel treatments have a finite life expectancy, and fire hazard increases over time as fuels accumulate on treated areas. Repetition and maintenance of fuel treatments are necessary in order to effectively maintain reduced fire hazard over time (Peterson et al., 2005, Johnson et al., 2007; Johnson et al., 2011) and thus must be included in analyses of long-term C storage. Also, Rhodes and Baker (2008) suggest that 2.0 – 4.2% of areas treated to reduce surface fuels are likely to encounter wildfires that would otherwise be high or moderate-high severity without treatment. Most studies in this review assume a future wildfire probability of 100%, reporting inferences that essentially detail a "best-case scenario" for wildfire emissions mitigation.

High spatial and temporal variability in the burn environment and complexity in atmosphere-terrestrial ecosystem interactions create substantial uncertainty in the estimation of emissions from wildfire. Heterogeneity in vegetation and fire intensity at multiple scales complicates the estimation of wildfire emissions using field-based methods (Wiedinmyer and Neff, 2007). At the stand scale, pre- and post-burn field measurements of C pools quantify the consumption of woody biomass (Campbell et al., 2007), but inferences about consumption in large landscapes remain uncertain, especially for litter, organic soil, and mineral soil C pools, which together comprise approximately 60% of all wildfire emissions (Campbell et al., 2007). There are also considerable challenges in estimating surface fuels and wildfire emissions from remotely-sensed data, such as the inability to detect surface fuels through forest canopies (Lachowski et al., 1995; Keane et al., 2001; Wiedinmyer and Neff, 2007). Simulation approaches are currently the most common strategy in estimating wildfire emissions, but modeling inputs are constrained by a paucity of empirical data, thus propagating uncertainty. Ultimately, improved

inferences of surface fuel consumption at large spatial scales are required to better address the capacity of dry temperate forests to sequester and retain C for climate change mitigation benefits.

The extent to which fuel treatments benefit long-term C sequestration is a function of the cumulative difference between initial net C losses associated with fuel reduction treatments and wildfire emissions and their associated release of C to the atmosphere (Hurteau et al., 2008). Our review of empirical data and model output indicate that untreated stands release more emissions to the atmosphere during wildfire than treated stands, and that tree mortality from wildfire is consistently reduced by the presence of fuel treatments. However, disagreement exists about the extent to which reduced wildfire emissions benefit long-term C sequestration. Some studies report net C benefits to fuel treatment application, whereas others suggest that any benefits to C sequestration from fuel reduction treatments are short-lived and that at long time scales, fuel reduction efforts potentially reduce C storage.

To benefit total ecosystem C storage, the removal and release of C through fuel treatments must not exceed the expected reductions in wildfire emissions. Substantial treatment costs derived from timber harvest, prescribed fire, and milling waste have been shown to exceed observed and simulated reductions in wildfire emissions. However, if thinned trees are milled into long-lasting wood products (lumber), the initial C loss associated with fuel treatments can be reduced and C benefits from fuel treatments may emerge (Finkral and Evans, 2008; Oneil and Lippke, 2010). The utilization of residual woody biomass (chips, bark, sawdust) generated from thinning for energy production provides additional C benefits by offsetting emissions associated with fossil fuel use (Perez-Garcia et al., 2006; Oneil and Lippke, 2010). However, many areas in the western US do not have biomass markets to compensate for the cost of treatments (Reinhardt and Holsinger, 2010), and many small diameter trees removed during fuel treatments are

unmerchantable and scattered onsite or burned in piles, increasing overall treatment emissions (Han et al., 2004). Though long-lasting wood products and energy utilization technologies can help to offset a portion of harvested C, a substantial contribution to total treatment costs still is derived from prescribed fire – the most successful management tool at reducing surface fuels and horizontal fuel continuity.

The capacity of fuel treatments to mitigate future fire behavior and move forest structure to a more fire-resistant condition is well described by current literature. However, C costs associated with fuel treatments have been shown to exceed the magnitude of the reduction in wildfire emissions, as a large percentage of biomass stored in forests (i.e., stem wood, branches, coarse woody debris) remains unconsumed even in high-severity fires (Campbell et al., 2007; Mitchell et al., 2009). Individual wildfires can transfer substantial amounts of forest C to the atmosphere, but much larger transfers occur between live and dead pools, where C is released gradually over decades via decomposition.

Fuel treatments may effectively reduce disturbance severity with certain known C costs, yet the expected C benefits from fuel reduction are realized only when wildfire occurs (Ager et al., 2010; Hurteau and North, 2010). Wildfire occurrence in a given area is uncertain and may never interact with treated stands with reduced fire hazard, ostensibly negating expected C benefits from fuel treatments. Ager et al., (2010) report that burn probabilities in treated stands in southern Oregon are less than 2%. Thus, there is a high probability that a treated stand does not encounter wildfire and, in fact, reduces C benefits to the system (Ager et al., 2010). Strategically located fuel treatments designed to optimize the reduction in wildfire spread per treatment area may successfully reduce the number of fuel treatments required to meet management objectives on a given landscape (Finney et al., 2007), but recent research suggests that even the most

optimal fuel treatment placements require approximately 10 hectares of treatment to protect 1 hectare of forest from wildfire (Finney et al., 2007; Campbell et al., 2011). Thus, the rarity of wildfire events creates a dilemma for those expecting to realize C sequestration benefits from fuel treatments.

The temporal horizon at which trade-offs are examined determines the degree to which fuel treatments result in a net C gain to the system. C removed or released during fuel treatments returns to the system through post-fire regeneration and rapid re-colonization of pioneer plant species (Law et al., 2004; Meigs et al., 2009). The time required to recover disturbance-related C losses (i.e., emissions, harvest) reflects disturbance intensity, and the resilience of the disturbed forest and may be shortened by treatments that successfully increase overstory survivorship (Hurteau and North, 2010). Eventually, net C assimilation from re-growth may exceed rates of decomposition and ecosystem respiration, converting disturbed landscapes from C sources to C sinks. Given the assumption that additional disturbance does not occur prior to forest maturation (i.e., recovery to initial stand age and density) the net release of C in fire-disturbed systems may eventually reach zero (Kashian et al., 2006). However, the extent to which this assumption is relevant in the context of a changing climate is unknown. Furthermore, vegetation life-form conversion (e.g., forestland to shrubland or grassland) following unsuccessful post-fire tree regeneration or resulting from disturbance to the soil microclimate are considerations for C sequestration in the context of climate change. For example, recovery of vegetation is strongly influenced by post-disturbance climatic conditions such as drought, and recent studies suggest C losses from high-severity wildfire can persist for decades as a result of protracted GPP recovery and type conversion (Savage and Mast, 2005; Deal et al., 2008; Dore et al., 2010). Fuel treatments may best benefit long-term C storage by reducing the likelihood of such events.

Literature on the topic of fuel treatment and wildfire effects on terrestrial C storage is limited, and results are equivocal. Current stand-scale studies, both empirical and modeled, show a wide range of treatment effects on C storage, complicating inferences about C benefits from fuel treatments. Studies at large spatial scales have shown that the rarity of high-severity wildfire events for any given treated stand reverses any expected C benefit, and instead results in a loss of C to the treatment system. However, few current studies satisfactorily address C response to future disturbance scenarios. There is considerable uncertainty in modeling ecosystem C exchange during extreme climatic events and large, severe disturbances – events that are expected to increase in frequency according to current projections of climate change. In order to improve the accuracy and usefulness of assessments of fuel treatment and wildfire trade-offs for C storage, future research must better quantify the entire fuel profile and consider the interaction of multiple future management and disturbance scenarios. Additional research on the potential for forested ecosystems to regenerate after large, severe disturbances, and on the utilization of woody biomass for energy, will also improve the accuracy of C tradeoff analyses.

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Table 1. Carbon emissions from equipment usage.

Study	Fuel treatment	Total equipment release (Mg C ha^{-1})	Post-treatment C storage (Mg C ha^{-1})*	Equipment release as % of post-treatment C storage
Finkral et al (2008)	Restoration thin	0.07 - 0.20 [1]	36.42	< 0.55
North et al (2009)	Understory thin	1.77	240	0.74
North et al (2009)	Understory thin and burn	1.89	190	1.00
North et al (2009)	Overstory thin	2.94	170	1.73
North et al (2009)	Overstory thin and burn	3.28	110	2.98
Stephens et al (2009)	Thin from below	< 1.0	190	< 1.0
Stephens et al (2009)	Thin from below and burn	< 1.0	118	< 1.0
Sorensen et al (2011)	Restoration thin	0.28	56.41	0.50
Sorensen et al (2011)	Restoration thin	0.15	51.59	0.29
Sorensen et al (2011)	Restoration thin	0.13	42.37	0.31
Sorensen et al (2011)	Restoration thin	0.06	57.98	0.1

* Includes all aboveground biomass

[1] Values are presented for multiple wood utilization scenarios (0.710: Firewood, 0.200: Paper, 0.116: Pallets/Construction)

Table 2. Harvested C and emissions from milling waste.

Study	Region	Fuel treatment	Harvested C (Mg C ha $^{-1}$)	C stored in wood products (Mg C ha $^{-1}$)	C emissions from milling waste (Mg C ha $^{-1}$)	Post treatment C storage (Mg C ha $^{-1}$)*	Milling waste as % of post-treatment C storage
Finkral and Evans (2008)	Northern Arizona	Restoration thin	8.24	6.11 - 6.82 [1]	0.02 - 4.94 [1]	36.42	0.05 - 13.56 [1]
North et al (2009)	Sierra Nevada, California	Understory thin and burn	54.72	32.83	21.89	190	11.52
North et al (2009)	Sierra Nevada, California	Understory thin	45.80	27.48	18.32	240	7.63
North et al (2009)	Sierra Nevada, California	Overstory thin and burn	91.87	54.40	37.47	110	34.07
North et al (2009)	Sierra Nevada, California	Overstory thin	94.48	56.20	38.28	150	25.52
Stephens et al (2009)	Sierra Nevada, California	Thin from below	23.34	15.20	8.14	118	6.90
Ager et al (2010)	Southeastern Oregon	Understory thin and burn	14.20	8.10	6.10	54.43	11.21

*Includes all aboveground biomass

[1] Values are presented for multiple wood utilization scenarios (Firewood, Paper, Pallettes/Construction)

Table 3. Summary of prescribed fire emissions.

	Study	Region	Fuel treatment	Total emissions from prescribed fire combustion (Mg C ha^{-1})	Post treatment C storage (Mg C ha^{-1}) *	Prescribed fire emission % of post-treatment C storage
Empirical studies	Finkral and Evans (2008)	Northern Arizona	Restoration thin	4.14 [1]	36.42	11.4
	North et al (2009)	Sierra Nevada, California	Burn only	14.79	240	6.2
	North et al (2009)	Sierra Nevada, California	Understory thin and burn	23.40	190	12.3
	North et al (2009)	Sierra Nevada, California	Understory thin	--	240	--
	North et al (2009)	Sierra Nevada, California	Overstory thin and burn	27.22	110	24.7
	North et al (2009)	Sierra Nevada, California	Overstory thin	--	150	--
	Stephens et al (2009)	Sierra Nevada, California	Burn only	28.12	172	16.4
	Stephens et al (2009)	Sierra Nevada, California	Thin from below and burn	34.47	118	29.2
	Sorensen et al (2011)	Northern Arizona	Restoration thin	9.58 [1]	56.41	17.0
	Sorensen et al (2011)	Northern Arizona	Restoration thin	6.39 [1]	51.59	12.4
	Sorensen et al (2011)	Northern Arizona	Restoration thin	3.80 [1]	44.88	8.5
	Sorensen et al (2011)	Northern Arizona	Restoration thin	5.82 [1]	42.37	13.7
	Sorensen et al (2011)	Northern Arizona	Restoration thin	1.01 [1]	57.98	1.7
Simulation studies	Hurteau and North (2009)	Sierra Nevada, California	Burn only	9 - 16 [2]	270 [2]	3 - 6 [2]
	Hurteau and North (2009)	Sierra Nevada, California	Understory thin and burn	8 - 12 [2]	180 [2]	4 - 6 [2]
	Hurteau and North (2009)	Sierra Nevada, California	Restoration burn	5 - 11 [2]	230 [2]	2 - 5 [2]
	Hurteau and North (2009)	Sierra Nevada, California	1865 reconstruction thin burn	4 - 10 [2]	270 [2]	1 - 4 [2]
	Ager et al (2010)	Southeastern Oregon	Understory thin and burn	19.5	54.43	35.8
	Rienhardt and Holsinger (2010)	Northern Rockies	Thin from below and burn	12.65	35.15	36.0
	Rienhardt and Holsinger (2010)	Northern Rockies	Burn only	6.79	51.56	13.2

*Includes all aboveground biomass

[1] Values represent emissions from pile burns

[2] Values estimated from (Hurteau and North 2009, Fig. 1)

Table 4. Summary of cumulative prescribed fire emissions.

Study	Region	Treatment type	Cumulative prescribed fire emissions (Mg C ha^{-1})	Simulated wildfire emissions (Mg C ha^{-1})
Hurteau and North (2009)	Sierra Nevada, California	Burn only	49.89	18.14
Hurteau and North (2009)	Sierra Nevada, California	Understory thin and burn	46.27	16.33
Hurteau and North (2009)	Sierra Nevada, California	Restoration burn	43.54	12.70
Hurteau and North (2009)	Sierra Nevada, California	1865 reconstruction thin burn	29.94	9.98
Sorensen et al (2011)	Northern Arizona	Restoration thin	34.01 / 25.38 [1]	24.38 / 21.53 [2]
Sorensen et al (2011)	Northern Arizona	Restoration thin	37.88 / 27.35 [1]	22.19 / 18.73 [2]
Sorensen et al (2011)	Northern Arizona	Restoration thin	27.62 / 20.44 [1]	16.96 / 14.00 [2]
Sorensen et al (2011)	Northern Arizona	Restoration thin	40.04 / 26.72 [1]	20.35 / 18.57 [2]
Sorensen et al (2011)	Northern Arizona	Restoration thin	32.09 / 21.78 [1]	17.17 / 13.85 [2]

[1] Rx10/Rx20 (prescribed fire simulated every 10/20 years)

[2] HF100/HF50 (no management with a high-intensity wildfire simulated within the next 100 years/50 years)

Table 5. Summary of wildfire emissions in treated and untreated stands.

Study	Region	Treatment type	Wildfire emissions in untreated stands (Mg C ha-1)	Wildfire emissions in treated stands (Mg C ha-1)	Reduction in wildfire emissions (Mg C ha-1)
Finkral and Evans (2008)	Northern Arizona	Restoration thin	8.33	5.92	2.41
Hurteau and North (2009)	Sierra Nevada, California	Understory thin	36.30	24.49	11.81
Hurteau and North (2009)	Sierra Nevada, California	Restoration thin	36.30	23.59	12.71
Hurteau and North (2009)	Sierra Nevada, California	1865 reconstruction thin	36.30	20.87	15.43
Hurteau and North (2009)	Sierra Nevada, California	Burn only	36.30	16.33	19.97
Hurteau and North (2009)	Sierra Nevada, California	Understory thin and burn	36.30	18.14	18.16
Hurteau and North (2009)	Sierra Nevada, California	Restoration burn	36.30	15.42	20.88
Hurteau and North (2009)	Sierra Nevada, California	1865 reconstruction thin burn	36.30	12.70	23.60
Rienhardt and Holsinger (2010)	Northern Rockies	Thin from below and burn	12.64	7.57	5.07
Rienhardt and Holsinger (2010)	Northern Rockies	Burn only	12.64	9.71	2.90 *
Ager et al (2010)	Southeastern Oregon	Understory thin and burn	1.38	0.02	1.36
North and Hurteau (2011)	Sierra Nevada, California	Thin from below and pile burn	67.80 [1]	29.71	40.1

* Significant difference not detected (Reinhardt and Holsinger 2010)

[1] Value represents the mean of 19 paired treated/untreated sites

Table 6. Wildfire emissions stratified by burn severity.

Study	Region	Wildfire severity	Wildfire emissions (Mg C ha^{-1})	% biomass consumed in wildfire
Campbell et al (2007)	Southwestern Oregon	Unburned/Very low	12.4	--
Campbell et al (2007)	Southwestern Oregon	Low	18.6	--
Campbell et al (2007)	Southwestern Oregon	Moderate	18.6	--
Campbell et al (2007)	Southwestern Oregon	High	28.6	--
Meigs et al (2009)	Eastern Oregon (Grand fir/Douglas-fir)	Low	16.6	13
Meigs et al (2009)	Eastern Oregon (Grand fir/Douglas-fir)	Moderate	25.3	19
Meigs et al (2009)	Eastern Oregon (Grand fir/Douglas-fir)	High	32.3	24
Meigs et al (2009)	Eastern Oregon (Ponderosa pine)	Low	19.7	23
Meigs et al (2009)	Eastern Oregon (Ponderosa pine)	Moderate	25.6	29
Meigs et al (2009)	Eastern Oregon (Ponderosa pine)	High	30.2	35

University of Washington

Abstract

SECTION 2:

Short-term fuel treatment effects on carbon storage following the 2006 Tripod Complex Fire, Okanogan-Wenatchee National Forest, Washington, USA

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Treatments that successfully reduce surface fuel in dry forests have been shown to mitigate the spread and severity of wildfire. Tree mortality and emissions from wildfire have also been shown to increase monotonically as burn severity increases. However, there is some disagreement in current literature on the extent to which reduced wildfire emissions benefit ecosystem carbon storage. With the aid of high-resolution (25-m), spatially explicit fuel maps and empirical data from the 2006 Tripod Fire in Washington (USA), we tracked carbon storage across the most recent management actions and identified treatment effects on wildfire carbon emissions. Thin-only treatments removed 38 Mg C ha^{-1} (thin-only) and increased wildfire carbon emissions (7 Mg C ha^{-1}) relative to untreated controls. Thin-and-burn treatments removed 75 Mg C ha^{-1} and

decreased wildfire carbon emissions (26 Mg C ha^{-1}). Previous work described the capacity of thin-and-burn treatments to reduce tree mortality from wildfire (Prichard et al., 2010), yet all treatments were characterized by an overall net reduction in aboveground carbon storage after consideration of initial treatment carbon costs. This study provides evidence that fuel treatments that effectively mitigate wildfire severity do not yield a net increase in carbon storage.

2.1 *Introduction*

Use of mechanical thinning and prescribed fire to reduce surface fuels in dry forest ecosystems has become increasingly common in western North America in response to increased wildfire hazard resulting from legacies of past management practices, expansion of the wildland-urban interface (WUI), and changes in climate (Covington et al., 1997; Agee and Skinner, 2005). Modifying potential fire behavior with fuel treatments is now a central management paradigm in most coniferous forests in the western United States (US) (Peterson et al., 2005; Stephens et al., 2009). Beyond wildfire risk reduction and increased resilience to disturbance, fuel treatments also increase productivity of residual trees (Thornley and Cannell, 2000), control stand structure and composition (McDowell et al., 2006), and may also provide ancillary benefits to multiple ecosystem services including, air quality, water quality, wildlife habitat, and carbon (C) sequestration (Deal et al., 2010).

Some policies and economic incentives encourage efforts to sequester C in dry forests as an approach to mitigate impacts of climate change (Fahey et al., 2010). Although a variety of silvicultural practices have been implemented to optimize C storage over time (Hoover and Stout, 2007), the long-term effects of wildfires and forest disturbances on forest C pools are poorly understood (Stephens et al., 2009). The ability of forest stands to sustain wildfire with reduced burn severity and minimal mortality of larger trees is central to the discussion of C management in dry forests. The capacity of forests to sequester and retain C is determined by the spatial distribution of stand ages and forest types, which in turn reflect the magnitude of forest fuels available for combustion, and the frequency and severity of disturbance (Goward 2008). Large quantities of C stored in forest ecosystems for centuries can be rapidly released to the atmosphere immediately following natural disturbance, such as wildfire (Körner 2003).

Consequently, fire hazard reduction must be accurately quantified in analyses of the management and sequestration of forest C.

Forest stands not subjected to thinning and fuel treatments release more C emissions to the atmosphere during wildfire than treated stands (North and Hurteau, 2011), and tree mortality from wildfire is consistently reduced by the presence of fuel treatments (Pollet and Omi, 2002; Stephens et al., 2009; Prichard et al., 2010). However, disagreement exists in the literature on the extent to which reduced wildfire emissions benefit C sequestration. Some studies report net C benefits of fuel treatment application (Finkral and Evans, 2008; North et al., 2009; Stephens et al., 2009; Reinhardt and Holsinger, 2010; Sorensen et al., 2011), while others suggest that any benefits to C sequestration from fuel reduction treatments are short-lived and that at long time scales, fuel reduction efforts potentially reduce C storage (Mitchell et al., 2010; Ager et al., 2010; Campbell et al., 2011; Hudiburg et al., 2011). Currently, analyses of C and wildfire in dry forests rely mostly on simulated wildfire events and rarely examine *in situ* effects of wildfire/treatment interactions. Empirical observations of treatment effects on C dynamics in fire-prone forests are critical to understanding the potential C benefits of fuel reduction efforts (North and Hurteau, 2011). This study explores treatment effects on C emissions from the 2006 Tripod Fire in Washington (USA) and builds on previous work by improving the resolution and accuracy at which forest fuels are considered for C budgets.

2.2 *Study Area*

The study area is located in the Okanogan-Wenatchee National Forest (OWNF), approximately 10 km north of Winthrop, Washington (Figure 1). The OWNF, in north central Washington State, extends from the crest of the Cascade Range eastward to savannah-steppe and

agricultural lands. Climate is characterized by cold winters and warm, dry summers with a prolonged summer drought. Mean annual temperature in Winthrop is 15.1° C, ranging from -11.6° C (January annual average minimum) to 30.1° C (July annual average maximum) (Western Regional Climate Center, Winthrop, Washington, http://www.wrcc.dri.edu). Mean annual precipitation is 360 mm with 70% of precipitation falling between October and March, predominantly as snow. Topography is extremely rugged with numerous subdrainages and deep, steep-sided valleys (Barksdale 1975; Raymond et al., 2006). Soils are generally coarse-textured Andisols with high gravel content (Natural Resources Conservation Service 2008).

In 2006, the Tripod Complex Fire burned through a variety of fuel treatments in the OWNF. The wildfires initiated as two lightning strikes and spread over 70 000 hectares of dry forest, burning with a mixture of crown and variable-intensity surface fire (Prichard et al., 2010). Fuel treatments located within the fire perimeter are widespread and of various ages, including thin only, thin and burn, and shelterwood treatments. Extensive mountain pine beetle (*Dendroctonus ponderosae* Hopkins) disturbance was present at higher elevations (> 1800 m) within and beyond the fire perimeter at the time of the Tripod Fire.

The area of focus for this study consists of low-elevation (< 1800m) dry forests and encompasses the locations where fuels reduction management activity is documented and expected to continue. This corresponds to areas historically characterized by high frequency, low severity wildfire. In the OWNF, a 6–12 year fire return interval (i.e., mean fire-free period) has been estimated from fire scar records at low elevations (Ohlson 1996, Everett et al., 2000, McKenzie et al., 2006). Prior to the Tripod Complex Fire, however, wildfires had been largely excluded from much of the study area for over 80 years (Prichard et al., 2010). Forests within the study area of the Tripod Fire are dominated by conifers and include multi-aged stands of

Douglas-fir (*Pseudotsuga menziesii* (Mirb.)), ponderosa pine (*Pinus ponderosa* P. & C. Lawson), western larch (*Larix occidentalis* Nutt.), and grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.).

The study area constitutes approximately 30% or 21 000 hectares of the area burned by the Tripod Complex Fire (Figure 2). The remainder of the fire area is mid- to high-elevation (1800-2500 m) forest dominated by Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), lodgepole pine (*Pinus contorta* Dougl. ex Loud. var. *latifolia* Engelm.), subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.), subalpine shrub and grassland, and bare rock. The research area was restricted to areas subject to fuels management in order to capture the trade-off between initial net C losses (e.g., thinning, prescribed fire) and potentially reduced C emissions associated with mitigated wildfire severity.

2.3 *Methods*

The concurrence of spatially explicit fuel maps (Raymond et al., 2006; McKenzie et al., 2007) and empirical data on fuel treatments in a large wildfire (Prichard et al., 2010) provides an opportunity to quantify a range of wildfire effects on C pools at a high resolution across a large spatial scale. In this study, a method is presented for the examination of treatment and wildfire effects on C dynamics in the OWNF. Specific attention is given to the characterization of the entire fuel profile as it interacts with treatment and disturbance.

Raymond et al. (2006) and McKenzie et al. (2007) demonstrate use of the Fuel Characteristic Classification System (FCCS) for fuel mapping at two scales and resolutions: the conterminous USA (CONUS) at 1-km resolution and at 25-m resolution across the OWNF. The FCCS fuelbed is a systematic catalog of physical properties of any wildland setting (Sandberg et al., 2001). Fuelbeds are assessed with peer-reviewed software system designed to characterize

fuel properties and potential fire behavior based on the inherent combustion of the components of individual fuelbeds. Each FCCS fuelbed represents a distinct fire environment and accounts for all aboveground combustible material at a given resolution (Ottmar et al., 2007). Fuelbeds specific to the OWNF were created from field data, fuels photo series, plant association descriptions, expert knowledge, and validated by local fire and fuel management personnel (Raymond et al., 2006). Each fuelbed is organized into vertical strata consisting of canopy fuels, shrub vegetation, herbaceous vegetation, woody fuels, litter, moss, and duff (Figure 3). Allometric methods are used to estimate biomass and C content for each fuelbed component and stratum. The FCCS software also calculates surface fire behavior, crown fire potential, and available fuel potential (Sandberg et al., 2001).

One of 40 fuelbeds was assigned to each 25-m cell in the OWNF (27,353,425 cells) based on forest composition, major vegetation forms, and species groups. Fuelbeds are further subdivided into structural or age classes (Raymond et al., 2006; McKenzie et al., 2007). Successional pathways for fuelbeds mapped in the OWNF are temporally linked such that fuel characteristics change logically through time with “natural” succession (i.e., time or wildfire) or silvicultural practices (Wright 2010). FCCS is used in this study to calculate and track C stores temporally across fuel strata, forest type, burn severity, disturbance regimes, and management history.

To establish a baseline for pre-treatment C storage, spatial data from FCCS (http://www.fs.fed.us/pnw/fera/fccs/maps.shtml) were summarized to determine the area represented by each fuelbed in the study domain and in each fuel treatment mapped in Prichard et al., (2010). Five fuelbeds comprise approximately 80% of the study area, and 10 fuelbeds comprise 95% of the study area. The entire study area consists of 17 fuelbeds (Table 1), with the

majority of fuelbeds representing a mix of ages in dry forest dominated by Douglas-fir, ponderosa pine, and grand fir. FCCS was used to calculate fuel loadings and C storage for each of the 17 fuelbeds, and results were expanded to the treatment area and study domain based on the calculated constancy of each fuelbed.

To quantify the effects of fuel treatments on C storage, the eight most abundant fuelbeds in the treatment area (Table 1) were customized for each treatment unit with the aid of empirical data from Prichard et al., (2010) and detailed harvest and burn records from the Methow Valley Ranger District (MVRD). Customizations of treated fuelbeds targeted only fuelbeds in which thinning is logical (i.e., shrubland and grassland excluded). Species and size specific harvest records were used to control changes in species composition. Empirical data (Prichard et al., 2010) were used to customize variables within the "canopy" stratum (i.e., tree height, height to live crown, density, and D.B.H.).

Customizations to the "woody fuel" stratum primarily address harvest residue (i.e., branches, bark, and foliage) that is left on site to decay, assembled into piles and burned, or burned during a broadcast prescribed fire at a later date. For the study area, harvest residue varies by tree species and tree age, but is equivalent to approximately 30% of aboveground live-tree biomass (Smith and DeBell, 1973; Harmon et al., 1996). Harvest residue in treatment units was added to the FCCS fuelbed "woody fuel" stratum and subsequently included in prescribed fire and wildfire emissions calculations. Specifically, harvest residue was considered "sound" and increased the depths of litter and woody fuel, proportional to the amount of C removed during harvest. No attempt was made to customize "shrub" or "nonwoody fuels" strata. Ultimately, customized fuelbeds reflect the structure and composition of each uniquely treated stand. Fuel loadings and C storage were then calculated in FCCS. Fuelbeds in untreated (control) units were

not changed.

2.4 *Sampling Design*

The sampling design for Prichard et al. (2010) includes eight thin-only units and eight thin-and-burn units (Figure 1). Eight control units with no record of harvest or burning were randomly selected at least 0.4 km from the Tripod perimeter and within 0.8 km of road access. Treatment units ranged in size from 3.3 ha to 41.9 ha, and each control unit was 8.2 ha.

Mechanical thinning in the study area was completed 7-16 years prior to the Tripod Fire and included both thin-from-below and shelterwood harvests. Thinning operations were primarily tractor-logged (12 of 16 treatment units). Operations for the 4 remaining treatments were conducted with helicopter operations. Prescribed burns were conducted on thin-and-burn units between 0 and 6 years prior to the Tripod Complex Fire. Burn records exist for each unit, and all burns used in the study were recorded by the OWNF as “successful” and accomplished fuel reduction objectives by reducing fine surface fuels (< 7.6 cm diameter) by 90–100% and large surface fuels (> 7.6 cm diameter) by up to 70% (Prichard et al., 2010).

Nested, variable-radius (3.58 to 25.37 m) circular plots oriented in a systematic grid were established in each treatment unit. At each plot, the following measurements were collected for each tree: DBH (cm), crown base height (m), height to live crown (m), tree height (m), maximum height of crown scorch (m), minimum and maximum bole char (m), percentage of the crown volume scorched, and tree severity index (US Department of the Interior National Park Service 2003). Tree burn severity classes were defined as follows: 1 = unburned, 2 = scorched foliage, 3 = lightly burned (some foliage and small twigs burned), 4 = moderately burned (foliage and small stems consumed), and 5 = severely burned (only charred stems remain).

2.5 *Carbon loss attributed to fuel treatment implementation*

Mechanical thinning and surface fuel treatments (i.e., prescribed fire) are commonly implemented to reduce fuel loadings and reduce the severity of wildfire and the risk of crown fire (Agee and Skinner, 2005). However, fuel reduction treatments coincide with an initial net C loss to the treatment area. Thinning treatments reduce C storage through the removal of trees and the combustion of fossil fuel in logging machinery and log transport. C is also lost through slash burning, milling waste, and the decay of logging slash and wood products. Prescribed fire releases C directly to the atmosphere via the combustion of surface fuels. We determined the magnitude of initial C loss in treatment areas relative to observed wildfire emissions in treated and untreated areas within the Tripod Fire perimeter.

To estimate C losses associated with prescribed fire in the study area, CONSUME 4.1 (http://www.fs.fed.us/pnw/fera/research/smoke/consume/index.shtml), a program designed to import fuels data directly from FCCS, was used to generate reports of fuel consumption and emissions by combustion phase. MVRD treatment records indicate that each prescribed fire in the study area was successfully conducted during the spring burning season (late April – early May). Environmental variables for prescribed fires were derived from spring prescribed burning conditions commonly used for fuels management in the OWNF (Harrod et al., 2009). To capture the range of potential C emissions from prescribed fire, two fires simulated under “dry” and “moist” spring burning conditions were conducted for each FCCS fuelbed (Table 2). CONSUME 4.1 also calculates a variety of pollutant emissions and heat release, but the focus of this study extends only to the amount of C released during prescribed fire.

MVRD treatment records were used to estimate C losses associated with tree removal (thinning) in the study area. Volume summaries for each treatment unit include data by species

and diameter class for the net volume and number of stems of timber removed (Table 3). Diameter classes are divided into “saw-timber” (> 17.5 cm DBH), and “non-saw timber” (12.7 to 17.5 cm DBH). Species-specific factors (i.e. specific gravity, wood density, and percent C content) were used to convert merchantable wood yield to C yield (Birdsey 1996).

During the milling process, approximately 40% of the merchantable yield of harvested trees (i.e., “saw-timber”) is converted to milling waste (Skog and Nicholson, 2000; Ingerson 2009; North et al., 2009). The remainder is stored for decades to centuries as a variety of long-lasting wood products. For this analysis, C stored in long-lasting wood products is subtracted from the total treatment cost for a given unit. All “non-saw timber” is assumed to be collected and processed as cogeneration without any contribution to milling waste. There are additional C benefits to the harvest system if energy produced by the burning of milling waste substitutes for fossil fuel consumption (Chen et al., 2000; Perez-Garcia et al., 2005), but such benefits are not within the scope of this study.

Forest operations produce C emissions by burning fossil fuels to power vehicles and machinery. The overall amount of fossil fuel consumption for a given site depends on the intensity and frequency of silvicultural treatments (Markewitz 2006). Equipment usage associated with mechanical thinning is described in several recent studies (Markewitz 2006; Finkral and Evans, 2009, North et al., 2009; Stephens et al., 2009; Sorensen et al., 2011). The magnitude of the release of C via equipment usage is a function of the volume per hour efficiency of a machine (i.e., liters/hour), number of hours per unit of area for a particular activity, and C emitted per volume of fuel consumed (Markewitz 2006). Factors such as haul length and site characteristics (i.e., slope) are also key considerations, but are reflected in the calculation of time per unit area for each activity. To estimate the C equivalent of equipment

usage, data sources from previous studies include logging contractor records, electronic activity recorders on logging equipment, and average consumption rates from manufacturer sources (e.g., Stihl [Virginia Beach, Virginia, USA], Caterpillar [Peoria, Illinois, USA]). Fuel consumption rates were converted to CO_2 emissions using the mean C contents of gasoline and diesel (EPA 2005) as well as the molecular mass ratio of CO_2 to C.

In this study, the machinery used to carry out mechanical thinning (i.e., chainsaws, tractor-skidders, log-loaders) is consistent with equipment used in previous studies. Transport distance is also consistent with previous studies (< 235 km). All harvested material in the study area was directed 90 – 120 km to a processing facility in Omak, WA. In the preceding literature review, emissions released from equipment used to harvest, load, and transport logs during fuel reduction treatments were found to be within a range of 0.05–3.28 Mg C ha^{-1}, approximately 0.5–3.0% of total post-treatment C storage (Table 4). Due to the absence of original records for equipment use for this study, published estimates from North et al. (2009) are used to estimate the C equivalent of equipment usage. Results presented by Finkral and Evans (2009) and Sorensen et al. (2011) are excluded because the harvest volume and overall working conditions (e.g., flat ground and openness of the stand) differed.

In the study area, a Columbia 107-II aircraft was used to yard four treatment units within the study area totaling approximately 100 hectares. In helicopter-logged units, the C equivalent of equipment usage was calculated using methods consistent with published estimates for other harvest machinery (see above). Volume per hour efficiency (n = 642 liters/hour) and other specifications are available for the aircraft (http://www.colheli.com). However, records of helicopter hours and average yarding distance are not available. To compensate for this data gap, previously published estimates of helicopter thinning efficiencies in the Pacific Northwest

(Flatten, 1991; Born, 1995; Wang et al., 2005) were used to approximate the number of helicopter hours required to yard the 4 treatment units (i.e., 270 hours). Once logs are helicopter-yarded, the same principles of haul length and log transport are considered to account for delivery of harvested material to the processing facility. Fuel consumption rates were converted to CO_2 emissions using the mean C contents of jet fuel (EPA 2005) as well as the molecular mass ratio of CO_2 to C.

2.6 *Carbon losses attributed to wildfire*

The potential trade-off to initial net C losses associated with fuel reduction treatments is a reduced risk of high-severity wildfire and its associated release of C to the atmosphere (Hurteau et al., 2008). To determine the magnitude of C loss attributed to the Tripod Fire, spatial data from the Monitoring Trends in Burn Severity (MTBS) database (http://www.mtbs.gov) were summarized by burn severity classification (e.g., very low, low, moderate, high) in the study domain and in each fuel treatment (Figure 4). In this case, burn severity is a general term that refers to the proportion of consumption of organic material or vegetation mortality that occurs as the direct consequence of fire (Lentile et al., 2006). FCCS fuelbeds were also summarized in the study domain and in each fuel treatment (Figure 4). Environmental variables (i.e., duff fuel moisture, 1000 hr fuel moisture, 10 hr fuel moisture) were developed for each burn severity classification in accordance with values reported in the East Cascades variant of the Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Reinhardt et al., 2008) and expert opinion (Ottmar, personal communication) (Table 2). CONSUME 4.1 was then used to generate combustion factors (e.g., the fraction of biomass consumed by fire) and emission reports for each FCCS fuelbed, fuelbed stratum, and burn severity in the study domain (Table 5).

2.7 *Data Analysis*

C storage, prescribed fire emissions, and wildfire emissions were each summarized by treatment unit (Table 6). C storage was summarized by treatment unit at three discrete time-steps (e.g., pre-treatment, post-treatment, post-wildfire). Although fuels are dynamic and highly variable over space and time, these distinct snapshots of fuel composition, structure, and quantity allow for a quantitative evaluation of C storage across the most recent management and disturbance events. Emission calculations were summarized by treatment unit, with comparisons measured between treatment type (i.e., thin and burn, thin only) and between time steps.

To calculate C storage, area-weighted C densities were calculated for each treatment unit based on the coverage of FCCS fuelbeds according to the formula:

$$\bar{X}_w = \frac{\sum W_i X_i}{\sum W_i}$$

where X_w is the weighted mean treatment C storage, W_i is the weighted FCCS coverage variable, and X_i is the observed FCCS C calculation. Pre-treatment C storage is calculated as an area-weighted mean of C density for 25-m resolution FCCS fuelbeds mapped by Raymond et al. (2006). Mechanical thinning treatments unique to each unit in the study design were then used to customize post-treatment fuelbeds with the aid of data from Prichard et al. (2010) and detailed silvicultural records from the MVRD. Emissions from prescribed fire for each treatment unit were also calculated as an area-weighted mean, with fuel consumption per fuelbed weighted by its coverage within the treatment unit. Emissions from wildfire are calculated for each treatment as an area-weighted mean from both fuelbed coverage and burn severity (MTBS).

To test for differences in emissions from wildfire between thin-only, thin-and-burn, and untreated units, a one-factor analysis of variance (ANOVA) was conducted on the area-weighted

means for each treatment unit with a pairwise comparison of means among all treatment combinations (Table 6). A normal quantile-quantile (QQ) plot was used to visually test that the data were normally distributed and satisfied the assumptions of ANOVA. The plotted data points follow a potentially nonlinear pattern, suggesting that the data may not be normally distributed. However, the null hypothesis that wildfire emissions are normally distributed could not be rejected at $\alpha = 0.05$ using the Shapiro-Wilk test for normality ($W = 0.9228$, $p = 0.067$). Due to the small sample size and small magnitude by which the Shapiro-Wilk test could not reject the null hypothesis, a non-parametric Kruskal-Wallis test was also conducted to test for differences in the mean ranks of wildfire emissions between treatments.

2.8 *Results: Fuel treatment costs*

Thinning treatments substantially decrease tree density, canopy bulk density, and total aboveground C storage to values generally proportional to treatment intensity (North et al., 2009; Sorensen et al., 2011). In this study, MVRD harvest records indicate that 13 to 48 Mg C ha^{-1}, or 14–31% of total aboveground C storage, was removed during thinning treatments (Tables 3 – 3b). Post-treatment tree density ranged from 172 to 475 trees per hectare (Prichard et al., 2010).

Fossil fuel consumption required to implement fuel reduction treatments is also generally proportional to treatment intensity and frequency (Markewitz 2006). Insufficient machine operation records for the study area required the use of published estimates to derive the C equivalent of equipment usage for fuel treatment applications. North et al. (2009) is the most analogous study site with respect to thinning intensity, working conditions (slope), and pre-treatment C storage. Emissions released from harvest machinery were found to be within a range of 1.7 to 3.3 Mg C ha^{-1}, or less than 3.0% of total post-treatment C storage for that study (Table

4). Due to remoteness and steep terrain, four treatment units in this study area were helicopter logged. Results indicate that although helicopter logging in the study area required the consumption of substantial jet fuel (> 1800 liters ha $^{-1}$), C emissions to the atmosphere were small relative to total treatment C costs (0.43 Mg C ha $^{-1}$). C emissions from helicopter use in these four treatment units are considered in addition to the emissions released from equipment used to harvest, load, and transport logs to the processing facility. Overall, equipment usage accounts for 1.9 to 3.4 Mg C ha^{-1}, or 1–2 % of total post-treatment C. The quantity of milling waste created from the processing of logs is also proportional to the quantity of logs harvested from each treatment unit (i.e., thinning intensity). In the study area, milling waste was estimated to range from 5 to 20 Mg C ha^{-1}, approximately 3–12 % of total post-treatment C (Table 6).

Contributions from prescribed fire to the C costs of fuel treatment application are substantial and approach the magnitude of C removed during whole-tree harvest (Table 6). However, prescribed fire effects can be highly variable, depending on fuels, stand structure, and fire weather (Harrod et al., 2009). Due to operational and liability constraints, prescribed fires for the study area were conducted in spring (late April – early May). Fuel moisture varies considerably by season, and as a result, spring prescribed fire is generally characterized by higher fuel moistures and lower proportional fuel consumption than fall prescribed fire (Kauffman and Martin, 1989, Knapp et al., 2009). To capture the range of potential emissions from prescribed fire, two simulated fires representing “dry” and “moist” fuel conditions were conducted for each FCCS fuelbed. Reports of fuel consumption and emissions by combustion phase, generated by CONSUME 4.1, indicate that 23–34 Mg C ha^{-1}, or 13–21% of the total post-treatment C, is released from the treatment units during prescribed fire (Table 7). Relative to “moist” fuel conditions, “dry” conditions resulted in a 6–10% increase in C emissions.

The sum of the net C loss for thin-and-burn units is variable and ranges from 53 to 90 Mg C ha^{-1}, or 20–29% of pre-treatment C storage. The net C loss for thin only units ranges from 18 to 41 Mg C ha^{-1}, or 9–14% of pre-treatment C storage. Variability primarily reflects differences in post-treatment residual tree density (172–475 trees ha^{-1}), which in turn, affects quantities of harvest residue, operational C release, and milling waste.

2.9 *Results: Wildfire emissions*

Emissions calculations indicate a significant difference between thin-and-burn and thin-only treatments in the study area (Figure 5). Thin and burn wildfire emissions were also significantly different than control treatments ($\alpha = 0.05$, $p < 0.001$) (Figure 5). Thin-and-burn treatments successfully reduced wildfire C emissions relative to control treatments, whereas thin-only units increased emissions, though not significantly ($p < 0.275$) (Figure 5). The increase in emissions from thin-only units is not unexpected because dispersed logging slash combined with extreme fire weather likely contributed to intense surface fire behavior and high tree mortality in thin only units (Prichard et al., 2010). Fire severity measures (i.e., maximum bole char, percent crown scorch, burn severity index) were consistently higher in thin-only units than control units, though not significantly different (Prichard et al., 2010). In the same study, fire severity measures in thin-and-burn treatments were nearly all significantly lower than those in both thin-only and control treatments (Prichard et al., 2010).

Fuel treatments have been shown to reduce fire severity and fire behavior (Agee and Skinner, 2005; Omi et al., 2006; Safford et al., 2009; Prichard et al., 2010), although few studies have connected efforts to mitigate fire severity with ecosystem C storage (Finkral and Evans, 2008; Hurteau et al., 2008, Stephens et al., 2009). Results from these studies suggest that the

potential benefit to C storage from fuel treatment application is a reduction of future wildfire emissions. Results from this study indicate that thin-and-burn treatments do, in fact, reduce wildfire emissions. The magnitude of emissions reduction in thin-and-burn treatments ranged from 12 to 16 Mg C ha^{-1}. In thin-only units, wildfire emissions increased from 1 to 6 Mg C ha^{-1}. However, to determine the extent to which fuel treatments benefit C storage, wildfire emissions must be compared to the quantity of C removed during treatment application.

Initial treatment C costs were derived from both measured values (i.e., prescribed fire, harvest removals) and previously published estimates (i.e., equipment usage, milling waste). Not surprisingly, thin-and-burn treatment costs were significantly higher than thin-only costs ($p < 0.001$). Prescribed fire emissions were shown to be a substantial treatment cost (23 to 34 Mg C ha^{-1}), exceeded only by C removed during harvest (19 to 69 Mg C ha^{-1}). Total treatment costs for thin-and-burn units ranged from 53 to 90 Mg C ha^{-1}, whereas total treatment costs for thin-only units ranged from 18 to 41 Mg C ha^{-1} (Table 6).

C costs associated with each treatment unit in the study exceeded observed reductions in wildfire emissions (Table 6). In thin-and-burn units, the amount of C removed during fuels treatment was 3–6 times larger than observed reductions in wildfire emissions. In thin-only units, observed increases in wildfire emissions following treatment exceed any expected benefits to ecosystem C storage. Moreover, only some fraction of areas treated to reduce surface fuels are likely to encounter wildfires that would otherwise be high or moderate-high severity without treatment (Rhodes and Baker 2008). A GIS summarization of MVRD treatment records in the study area indicates that approximately 425 ha of the 880 ha treated areas (thin only and thin and burn) conducted prior to the Tripod Fire (1990–2004) were not within the Tripod Fire perimeter and have yet to experience wildfire of any severity. This proportion of the treated landscape that

experienced wildfire far exceeds an estimate of 2.0–4.2% reported by Rhodes and Baker (2008), but still strengthens the conclusion of overall net C reduction by fuel treatments.

2.10 *Discussion*

This study provides evidence that fuel treatments located in the 2006 Tripod Fire result in a net reduction of ecosystem C storage. Results of studies on the effects of fuel treatments and wildfires on C storage across any scale are limited and equivocal. Studies that report net C benefits from fuel treatments in fire-prone forests are typically based on incomplete metrics of C accounting (i.e., tree-based C, fire-killed tree mortality) (Hurteau et al., 2008; Stephens et al., 2009); fail to address wildfire burn probability, resulting in an overestimation of treatment efficacy in wildfire simulations (Finkral and Evans, 2008; Hurteau et al., 2008, North et al., 2009; Stephens et al., 2009; Reinhardt and Holsinger, 2010; Sorensen et al., 2011); or are set in geographic regions where fuel conditions are such that treatment costs and fire emissions are relatively low (Finkral and Evans, 2008; Sorensen et al., 2011). Studies that account for these points report net reductions in C storage as a result of fuel treatment application (Mitchell et al., 2009; Ager et al., 2010; Campbell et al., 2011), but rely almost exclusively on simulation modeling with few empirical observations.

Comparable studies with the most analogous initial stand conditions (North et al., 2009; Stephens et al., 2009; North and Hurteau, 2011) report similar magnitudes of C removed during harvest, prescribed fire emissions, and wildfire emissions (Table 7). In these studies, the quantity of C removed during harvest (23 to 95 Mg C ha^{-1}) is the largest source of C in total fuel treatment costs and is generally proportional to thinning intensity (Table 7). Similarly, results presented in this study attribute 13 to 45 Mg C ha^{-1} to the amount of C removed during harvest.

Prescribed fire emissions rank as the next largest source of C in total treatment costs, and results reported in this study are also within the range of values published by comparable studies (14 to 35 Mg C ha^{-1}). The magnitude of wildfire emissions from untreated stands reported in this study (29 to 44 Mg C ha^{-1}) is within the range of published results from comparable studies (33 to 68 Mg C ha^{-1}), and values are also consistent with other reports of high-severity wildfire emissions in the Pacific Northwest (28 to 32 Mg C ha^{-1}) (Campbell et al., 2007; Meigs et al., 2009).

Results from northern Arizona and the northern Rockies (Finkral and Evans, 2008; Reinhardt and Holsinger, 2010; Sorensen et al., 2011) indicate much less C removed during harvest (6 to 32 Mg C ha^{-1}) and less emissions from prescribed fire (1 to 12 Mg C ha^{-1}). In both studies from northern Arizona (Finkral and Evans, 2008; Sorensen et al., 2011), prescribed fires are conducted as the burning of slash piles. Slash piles burn more efficiently, with significantly less smoke produced per unit mass of fuel consumed than broadcast burns of the same material (Hardy 1996; Ward et al., 1998). Reinhardt and Holsinger (2010) report fewer emissions in both prescribed and wildland fire in the northern Rockies (6 to 12 Mg C ha^{-1}) which is most likely a reflection of low pre-treatment C storage (< 60 Mg C ha^{-1}), especially in surface fuel loadings.

This study is unique in its attention to the quantification of surface fuels across a large spatial scale. With the aid of high-resolution, spatially explicit fuel maps (Raymond et al., 2006; McKenzie et al., 2007) and empirical data from a large wildfire (Prichard et al., 2010), we accurately tracked C storage across the most recent management actions and disturbances in the study region. General principles from this study apply to all dry temperate forests in the western US, but specific inferences may be limited forests of the inland Pacific Northwest. Future research must continue to refine estimates of surface fuel consumption in both prescribed fire and wildfire, because fuel consumption is potentially the most variable physical quantity in the C

budget of dry forests. Additional research is also needed to identify patterns of wildfire effects on C budgets at regional and sub-continental scales. Sensitivity of C stores to potential changes in the disturbance regime over long time scales will help to characterize how C budgets may respond to a changing climate.

Policy makers are in the process of determining how to regulate carbon dioxide and other greenhouse gases. There is ongoing activity to create tradable C credits using a market-based system (Lippke and Perez-Garcia, 2008). Although the future of C emission taxes and cap and trade systems is unclear and faces significant political resistance in the US, a global trend of increased regulation of C is likely. C accounting for forests in the US is currently voluntary, yet a variety of compliance markets for C trading and voluntary state or regional policy mechanisms now exist (Fahey et al., 2010). Accurately quantified effects of forest management and disturbance on C budgets are critical to the effectiveness of emissions regulation and climate change mitigation, especially in dry temperate forests characterized by vulnerability to C release from wildfire.

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Table 1. Fractional coverage of FCCS fuelbeds in treatments that include prescribed fire. List of FCCS fuelbeds (n=17) occurring within treatment areas and their associated fractional coverage of the study area. 5 individual fuelbeds comprise 80 percent of the study area.

				FCCS Fuelbed #	Fuelbed name
0.47	0.47	0.47	0.47	522	Douglas-fir -- ponderosa pine -- grand fir forest, 40-80 yr
0.10	0.10	0.10	0.10	517	Douglas-fir -- ponderosa pine -- grand fir forest, 25-40 yr
0.09	0.09	0.09	0.09	527	Douglas-fir -- ponderosa pine -- grand fir forest, 80-150 yr
0.09	0.09	0.09	0.09	359	Subalpine fir -- Engelmann spruce -- Douglas-fir -- lodgepole pine forest
0.05	0.05	0.05	0.05	664	Douglas-fir / ninebark forest, 40-80 yr
	0.03	0.03	0.03	231	Sagebrush shrubland
	0.03	0.03	0.03	678	Ponderosa pine -- western larch -- Douglas-fir forest, 40-80 yr
	0.03	0.03	0.03	230	Huckleberry - false huckleberry shrubland
		0.03	0.03	636	Ponderosa pine forest, 40-80 yr
		0.03	0.03	507	Douglas-fir -- ponderosa pine -- grand fir forest, 15-25 yr
			0.02	233	Cheatgrass grassland
			0.01	598	Lodgepole pine forest, 20-40 yr
			0.01	501	Douglas-fir -- ponderosa pine forest, 0-15 yrs shrubland
			tr	628	Ponderosa pine forest, 25-40 yr
			tr	574	Lodgepole pine -- subalpine fir forest, 10-40 yr
			tr	680	Ponderosa pine -- western larch -- Douglas-fir forest, 80-100 yr
			tr	661	Douglas-fir / ninebark forest, 15-40 yr
0.80	0.90	0.95	0.99		TOTAL

Table 2. Environmental variables used in the modeling of prescribed fire and wildfire. Environmental variables were modified to represent "dry" and "moist" fuel moisture within a range of established conditions for spring prescribed fire (OWNF). Environmental variables were developed for each burn severity classification in accordance with values reported in (FFE-FVS) (Reinhardt et al., 2008) and expert opinion (Ottmar, personal communication).

	Prescribed fire		Wildfire severity			
	Moist	Dry	High	Moderate	Low	Very Low
Duff fuel moisture %	90	60	15	30	60	90
1000 hr fuel moisture %	12	7	9	10	12	14
10 hr fuel moisture %	13	9	3.5	4.5	6	8
Canopy loading consumed %	15	30	90	75	50	25

Table 3. Summary of Methow Valley Ranger District treatment records and species-specific carbon factors (example). PICO, *Pinus contorta*; PSME, *Psuedostuga menziesii*. (2): Signifies "non-saw timber" (12.7–17.6 cm DBH). All other values represent "saw-timber" (> 17.6 cm DBH).

									m^3	m^3	Removed during harvest	
Unit(s)	Area (ha)	Species	Density (kg m^{-3})	% C	Multiplier	Total wood volume (m^3)	kg C m^{-3}	# of Trees	Vol (Gross)	Vol (Net)	Mg C	Mg C ha^{-1}
Soaker 9/10	19.1	PICO	419.82	0.51	2.25	723.66	214.97	1105	321.06	297.55	155.56	8.14
		PICO (2)	419.82	0.51	2.25	199.97	214.97	846	88.72	88.72	42.98	2.25
		PSME	449.84	0.51	1.68	1631.47	230.35	2891	974.01	918.51	375.72	19.67
		PSME (2)	449.84	0.51	1.68	115.16	230.35	649	68.75	68.75	26.52	1.39
		Total						5491	1452.54	1373.53	600.78	31.45

Table 3b. Carbon removed during harvest for thin-and-burn and thin-only treatment units. PICO, *Pinus contorta*; PSME, *Psuedostuga menziesii*; ABLA, *Abies lasiocarpa*; PIPO, *Pinus ponderosa*. (2): Signifies "non-saw timber" (12.7–17.6 cm DBH). All other values represent saw-timber (> 17.6 cm DBH).

Thin-and-burn

Unit(s)	Area (ha)	Species	Mg C ha^{-1}
Soaker 13	4.5	PICO	4.94
		PICO (2)	0.68
		PIPO	4.67
		PSME	6.31
		PSME (2)	0.85
		Total	17.46
Soaker 9/10	19.1	PICO	8.14
		PICO (2)	2.25
		PSME	19.67
		PSME (2)	1.39
		Total	31.45
Soaker 5	8.1	PICO	4.63
		PICO (2)	1.09
		PIPO	5.58
		PSME	34.13
		PSME (2)	2.27
		Total	47.70
Bear 12	22.4	PICO	8.50
		PICO (2)	2.14
		PIPO	2.13
		PSME	18.88
		PSME (2)	1.50
		Total	33.15
Bear 2	5.7	PICO	5.20
		PICO (2)	0.47
		PIPO	2.80
		PSME	8.95
		PSME (2)	1.43
		Total	18.85
Bear 49	7.3	PICO	10.81
		PICO (2)	3.36
		PSME	25.22
		PSME (2)	6.30
		Total	45.69
Bear 50	6.5	PICO	2.50
		PICO (2)	0.86
		PSME	19.87
		PSME (2)	8.00
		Total	31.23
Solar 91	5.3	PICO	4.28
		PSME	9.03
		PSME (2)	17.92
		Total	31.23

Thin-only

Unit(s)	Area (ha)	Species	Mg C ha^{-1}
Soaker 49	3.3	PICO	13.83
		PICO (2)	9.62
		PSME	1.53
		PSME (2)	0.76
		Total	25.75
Soaker 8	4.9	PICO	8.91
		PICO (2)	1.25
		PIPO	0.20
		PIPO (2)	0.06
		PSME	16.45
		PSME (2)	0.81
		Total	27.68
Soaker 47	11.4	PICO	4.78
		PICO (2)	4.32
		PSME	3.15
		PSME (2)	1.03
		Total	13.28
Solar 87	12.2	PICO	6.56
		PICO (2)	3.15
		PIPO (2)	0.33
		PSME	14.66
		Total	24.70
Solar II 84	41.9	ABLA	0.05
		PICO	4.11
		PICO (2)	1.64
		PSME	21.00
		PSME (2)	2.34
		Total	29.15
Solar II 82	14.7	ABLA	0.05
		PICO	4.09
		PICO (2)	1.64
		PSME	20.93
		PSME (2)	2.33
		Total	29.04
Solar II 16	11	PICO	5.23
		PICO (2)	1.76
		PSME	21.36
		PSME (2)	1.99
		Total	30.34
Solar II 12	28.5	PICO	5.29
		PICO (2)	1.78
		PSME	21.60
		PSME (2)	2.01
		Total	30.68

Table 4. Emissions from equipment usage.

Study	Fuel treatment	Total equipment release (Mg C ha^{-1})	Post treatment C storage (Mg C ha^{-1}) *	Equipment release as % of post-treatment C storage
Finkral and Evans (2008)	Restoration thin	0.07 - 0.20 [1]	36.42	< 0.55
North et al (2009)	Understory thin	1.77	240	0.74
North et al (2009)	Understory thin and burn	1.90	190	1.00
North et al (2009)	Overstory thin	2.94	170	1.73
North et al (2009)	Overstory thin and burn	3.28	110	2.98
Stephens et al (2009)	Thin from below	< 1.0	190	< 1.0
Stephens et al (2009)	Thin from below and burn	< 1.0	118	< 1.0
Sorensen et al (2011)	Restoration thin	0.28	56.41	0.50
Sorensen et al (2011)	Restoration thin	0.15	51.59	0.29
Sorensen et al (2011)	Restoration thin	0.05	44.88	0.11
Sorensen et al (2011)	Restoration thin	0.13	42.37	0.31
Sorensen et al (2011)	Restoration thin	0.06	57.98	0.10

* Includes all aboveground biomass

[1] Values are presented for multiple wood utilization scenarios (Firewood, Paper, Pallettes/Construction)

Table 5. Pre-treatment aboveground biomass (Mg C ha^{-1}) and combustion factors (fraction of biomass consumed) by burn severity.

	FCCS Fuelbed #	522		517		527		359		664		731		678		730	
Pre-Treatment	Canopy	348.82		96.14		184.32		657.91		85.67		17.46		314.54		0.48	
	Ground fuels	7.43		14.27		9.42		49.09		8.50		0.00		15.58		0.18	
	Litter-lichen-moss	3.36		1.82		2.66		1.66		0.92		1.71		4.06		0.84	
	Nonwoody	0.11		0.06		0.11		0.34		0.67		0.06		0.22		0.00	
	Shrub	1.18		3.95		2.54		3.37		0.92		10.18		0.04		7.62	
	Woody fuels	22.78		29.10		14.09		27.51		11.93		0.34		21.74		0.22	
	TOTAL	383.68		145.32		213.14		739.87		108.60		19.66		356.19		9.35	
Burn Severity			CF		CF		CF		CF		CF		CF		CF		CF
	Canopy	13.51	0.04	10.74	0.11	13.57	0.07	12.46	0.02	6.24	0.07	1.37	0.08	10.18	0.03	0.35	0.72
HIGH	Ground fuels	0.84	0.11	7.04	0.49	5.92	0.63	47.16	0.96	1.78	0.21	0.00	--	11.15	0.72	0.00	0.00
	Litter-lichen-moss	3.36	1.00	1.69	0.93	1.24	0.47	1.01	0.61	0.61	0.66	0.10	0.06	4.00	0.98	0.44	0.53
	Nonwoody	0.10	0.93	0.05	0.93	0.10	0.93	0.31	0.93	0.62	0.93	0.05	0.93	0.21	0.93	0.00	--
	Shrub	0.53	0.45	1.08	0.27	0.38	0.15	1.68	0.50	0.41	0.45	6.51	0.64	0.02	0.43	4.48	0.59
	Woody fuels	19.18	0.84	24.13	0.83	12.32	0.87	23.68	0.86	9.22	0.77	0.31	0.93	17.63	0.81	0.21	0.93
	C EMISSIONS	37.52	0.10	44.74	0.31	33.54	0.16	86.30	0.12	18.88	0.17	8.35	0.42	43.17	0.12	5.48	0.59
MOD	Canopy	11.26	0.03	8.95	0.09	11.31	0.06	10.39	0.02	5.20	0.06	1.14	0.07	8.48	0.03	0.29	0.60
	Ground fuels	0.78	0.11	6.97	0.49	5.88	0.63	47.11	0.96	1.74	0.21	0.00	--	11.07	0.71	0.00	0.00
	Litter-lichen-moss	3.36	1.00	1.69	0.93	1.24	0.47	1.01	0.61	0.61	0.66	0.10	0.06	4.00	0.98	0.44	0.52
	Nonwoody	0.10	0.93	0.05	0.93	0.10	0.93	0.31	0.93	0.62	0.93	0.05	0.93	0.21	0.93	0.00	--
	Shrub	0.53	0.45	1.08	0.27	0.38	0.15	1.68	0.50	0.41	0.45	6.51	0.64	0.02	0.43	4.48	0.59
	Woody fuels	19.11	0.84	24.03	0.83	12.28	0.87	23.58	0.86	9.17	0.77	0.31	0.93	17.54	0.81	0.21	0.93
	C EMISSIONS	35.15	0.09	42.78	0.29	31.19	0.15	84.08	0.11	17.75	0.16	8.12	0.41	41.31	0.12	5.42	0.58
LOW	Canopy	7.51	0.02	5.97	0.06	7.54	0.04	6.92	0.01	3.47	0.04	0.76	0.04	5.65	0.02	0.19	0.40
	Ground fuels	0.70	0.09	6.88	0.48	5.83	0.62	47.05	0.96	1.69	0.20	0.00	--	10.94	0.70	0.00	0.00
	Litter-lichen-moss	3.36	1.00	1.69	0.93	1.24	0.47	1.01	0.61	0.61	0.66	0.10	0.06	4.00	0.98	0.43	0.51
	Nonwoody	0.10	0.93	0.05	0.93	0.10	0.93	0.31	0.93	0.62	0.93	0.05	0.93	0.21	0.93	0.00	--
	Shrub	0.53	0.45	1.08	0.27	0.38	0.15	1.68	0.50	0.41	0.45	6.51	0.64	0.02	0.43	4.48	0.59
	Woody fuels	18.96	0.83	23.82	0.82	12.18	0.86	23.38	0.85	9.05	0.76	0.31	0.92	17.35	0.80	0.21	0.92
	C EMISSIONS	31.17	0.08	39.48	0.27	27.27	0.13	80.36	0.11	15.84	0.15	7.74	0.39	38.17	0.11	5.31	0.57
VERY LOW	Canopy	3.75	0.01	2.98	0.03	3.77	0.02	3.46	0.01	1.73	0.02	0.38	0.02	2.83	0.01	0.10	0.20
	Ground fuels	0.58	0.08	6.75	0.47	5.75	0.61	46.96	0.96	1.61	0.19	0.00	--	10.77	0.69	0.00	0.00
	Litter-lichen-moss	3.36	1.00	1.69	0.93	1.24	0.47	1.01	0.61	0.61	0.66	0.10	0.06	4.00	0.98	0.42	0.50
	Nonwoody	0.10	0.93	0.05	0.93	0.10	0.93	0.31	0.93	0.62	0.93	0.05	0.93	0.21	0.93	0.00	--
	Shrub	0.53	0.45	1.08	0.27	0.38	0.15	1.68	0.50	0.41	0.45	6.51	0.64	0.02	0.43	4.48	0.59
	Woody fuels	18.81	0.83	23.59	0.81	12.08	0.86	23.17	0.84	8.93	0.75	0.31	0.91	17.16	0.79	0.20	0.91
	C EMISSIONS	27.14	0.07	36.14	0.25	23.32	0.11	76.60	0.10	13.91	0.13	7.35	0.37	34.98	0.10	5.20	0.56

Refer to Table 1 for descriptions of each FCCS fuelbed.

Table 6. ANOVA tables for treatment C costs and wildfire C emissions.

ANOVA table for treatment C costs.

Source of variation	DF	SS	MS	F value	Pr (> F)
Treatment	1	5542.4	5542.4	57.71	2.49 e -6
Residuals	14	1344.5	96		
Total	15	6886.9			

ANOVA table for wildfire C emissions.

Source of variation	DF	SS	MS	F value	Pr (> F)
Treatment	2	4267.5	2113.8	40.88	5.74 e -8
Residuals	21	1096	52.2		
Total	23	5363.5			

Table 7. Summary of treatment C costs and wildfire emissions.

		Area (ha)	Pre-treatment C (Mg C ha^{-1})	Harvested C (Mg C ha^{-1})	Merchantable C removed in harvest (Mg C ha^{-1})	Lumber (Mg C ha^{-1})	Harvest residue (Mg C ha^{-1})	Rx emissions HIGH (Mg C ha^{-1})	Rx emissions LOW (Mg C ha^{-1})	Milling waste (Mg C ha^{-1})	Timber operations (Mg C ha^{-1})	Total treatment costs HIGH (Mg C ha^{-1})	Total treatment costs LOW (Mg C ha^{-1})	Wildfire emissions (Mg C ha^{-1})
Thin-and-burn	Soaker 13	4.5	230.55	25.09	17.56	10.54	7.53	32.72	30.06	7.02	1.90	56.19	53.53	13.82
	Soaker 9/10	19.1	348.33	44.94	31.45	18.87	13.48	32.93	30.93	12.58	3.28	74.86	72.85	18.75
	Soaker 5	8.1	310.60	68.14	47.70	28.62	20.44	27.71	25.18	19.08	3.28	89.59	87.06	15.57
	Bear 50	5.3	287.26	44.61	31.23	18.74	13.38	33.89	31.37	7.54	1.90	69.20	66.68	20.37
	Bear 49	7.3	292.76	65.27	45.69	27.41	19.58	24.53	23.01	13.26	3.28	78.93	77.40	15.52
	Bear 12	5.7	343.94	47.36	33.15	19.89	14.21	33.30	30.71	18.28	3.28	82.32	79.74	17.75
	Bear 2	22.4	275.67	26.93	18.85	11.31	8.08	26.93	24.69	12.49	1.90	56.94	54.70	12.63
	Solar 91	6.5	330.92	44.62	31.23	18.74	13.39	31.05	28.49	12.49	3.28	72.70	70.14	13.60
Thin-only	Soaker 49	3.3	278.07	36.78	25.75	15.45	11.03	--	--	10.30	2.94	34.57	34.57	44.78
	Soaker 8	4.9	389.87	39.55	27.68	16.61	11.86	--	--	11.07	2.94	36.95	36.95	49.68
	Soaker 47	11.4	177.54	18.97	13.28	7.97	5.69	--	--	5.31	1.90	18.21	18.21	38.03
	Solar 87	12.2	286.73	35.29	24.70	14.82	10.59	--	--	9.88	2.94	33.29	33.29	43.84
	Solar II 84	41.9	346.81	41.64	29.15	17.49	12.49	--	--	11.66	3.37	39.18	39.18	44.13
	Solar II 82	14.7	272.08	41.48	29.04	17.42	12.44	--	--	11.62	3.37	39.04	39.04	44.94
	Solar II 16	11	392.80	43.34	30.34	18.20	13.00	--	--	12.13	3.37	40.64	40.64	51.46
	Solar II 12	28.5	443.59	43.83	30.68	18.41	13.15	--	--	12.27	3.37	41.06	41.06	71.00
Control	C1	8.2	317.10											34.13
	C2	8.2	342.95											39.75
	C3	8.2	382.74											41.40
	C4	8.2	200.55											41.57
	C5	8.2	283.25											31.91
	C6	8.2	229.24											28.46
	C7	8.2	146.79											22.89
	C8	8.2	270.76											41.68

Table 8. Harvested carbon, post-treatment carbon storage, emissions from prescribed fire and wildfire.

Study	Region	Fuel treatment	Harvested C (Mg C ha $^{-1}$)	Post-treatment C storage (Mg C ha^{-1}) *	Prescribed fire emissions (Mg C ha^{-1})	Wildfire emissions (Mg C ha^{-1})
Finkral and Evans, (2008)	Northern AZ	Restoration thin	8.24	36.42	4.14 [1]	5.92
Hurteau and North, (2009)	Sierra Nevada, CA	Burn only	--	270 [2]	9 - 16 [2]	16.33
Hurteau and North, (2009)	Sierra Nevada, CA	Understory thin and burn	47.8	180 [2]	8 - 12 [2]	18.14
Hurteau and North, (2009)	Sierra Nevada, CA	Understory thin	47.8	185 [2]	--	24.49
Hurteau and North, (2009)	Sierra Nevada, CA	Restoration burn	--	230 [2]	5 - 11 [2]	15.42
Hurteau and North, (2009)	Sierra Nevada, CA	Restoration thin	--	245 [2]	--	23.59
Hurteau and North, (2009)	Sierra Nevada, CA	1865 reconstruction thin burn	65	270 [2]	4 - 10 [2]	12.70
Hurteau and North, (2009)	Sierra Nevada, CA	1865 reconstruction thin	65	275 [2]	--	20.87
North et al., (2009)	Sierra Nevada, CA	Burn only	--	240	14.79	--
North et al., (2009)	Sierra Nevada, CA	Understory thin and burn	54.72	190	23.40	--
North et al., (2009)	Sierra Nevada, CA	Understory thin	45.80	240	--	--
North et al., (2009)	Sierra Nevada, CA	Overstory thin and burn	91.87	110	27.22	--
North et al., (2009)	Sierra Nevada, CA	Overstory thin	94.48	150	--	--
Stephens et al., (2009)	Sierra Nevada, CA	Thin from below	23.34	118	34.47	--
Stephens et al., (2009)	Sierra Nevada, CA	Burn only	--	172	28.12	--
Ager et al., (2010)	Southeastern OR	Understory thin and burn	14.20	54.43	19.5	0.02
Rienhardt and Holsinger, (2010)	Northern Rockies, ID/MT	Thin from below and burn	10.64	35.15	12.65	7.57
Rienhardt and Holsinger, (2010)	Northern Rockies, ID/MT	Burn only	10.64	51.56	6.79	9.71
North and Hurteau, (2011)	Sierra Nevada, CA	Thin from below and pile burn	50.3 [3]	153 [3]	--	29.7 [3]
Sorensen et al., (2011)	Northern AZ	Restoration thin	6.11	57.98	1.01 [1]	20.35 / 18.57 [4]
Sorensen et al., (2011)	Northern AZ	Restoration thin	15.47	51.59	6.39 [1]	22.19 / 18.73 [4]
Sorensen et al., (2011)	Northern AZ	Restoration thin	20.00	42.37	5.82 [1]	16.96 / 14.00 [4]
Sorensen et al., (2011)	Northern AZ	Restoration thin	32.47	56.41	9.58 [1]	24.38 / 21.53 [4]

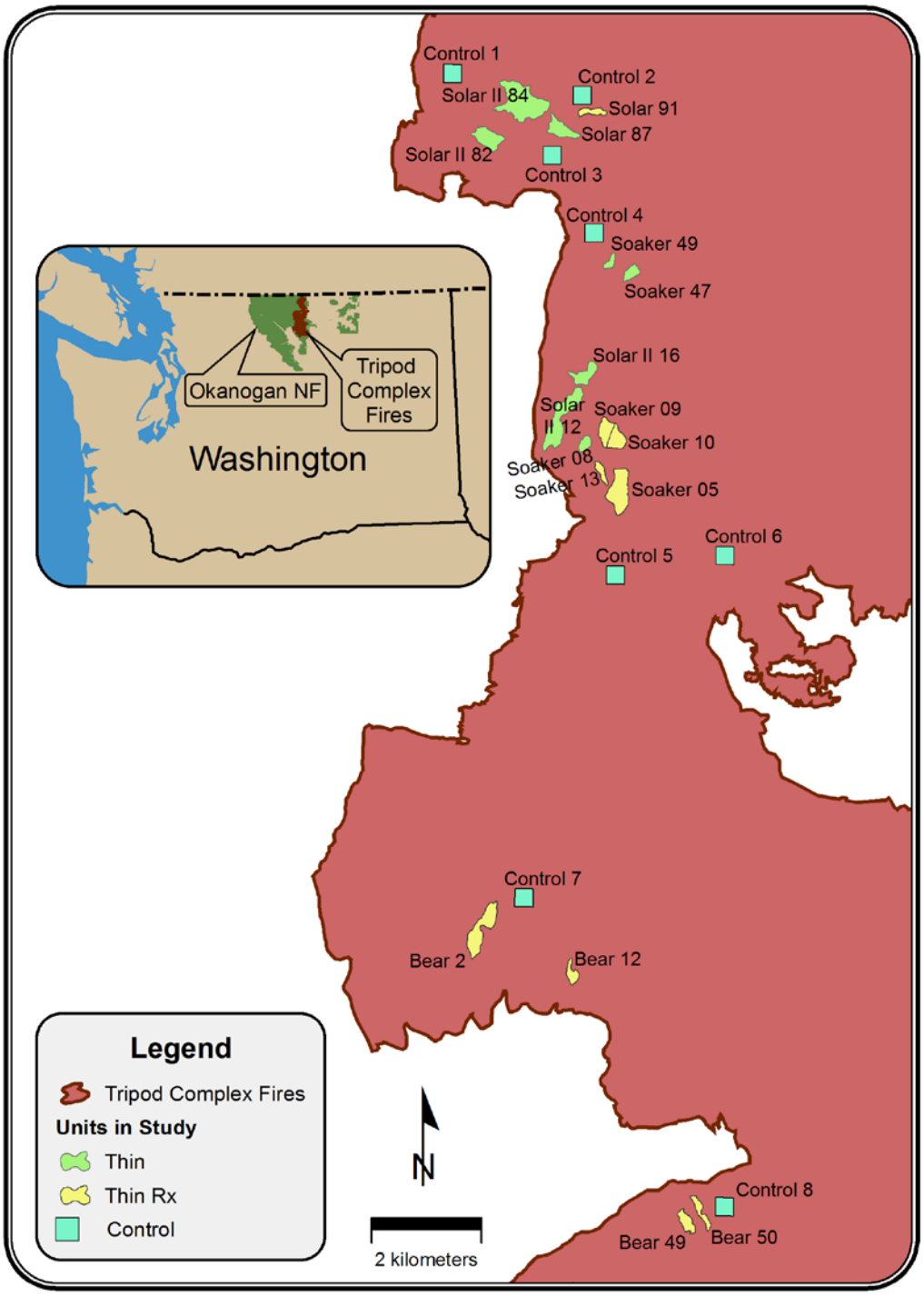
*Includes all aboveground biomass

[1] Values represent emissions from pile burns

[2] Values from Figure 1 (Hurteau and North, 2009)

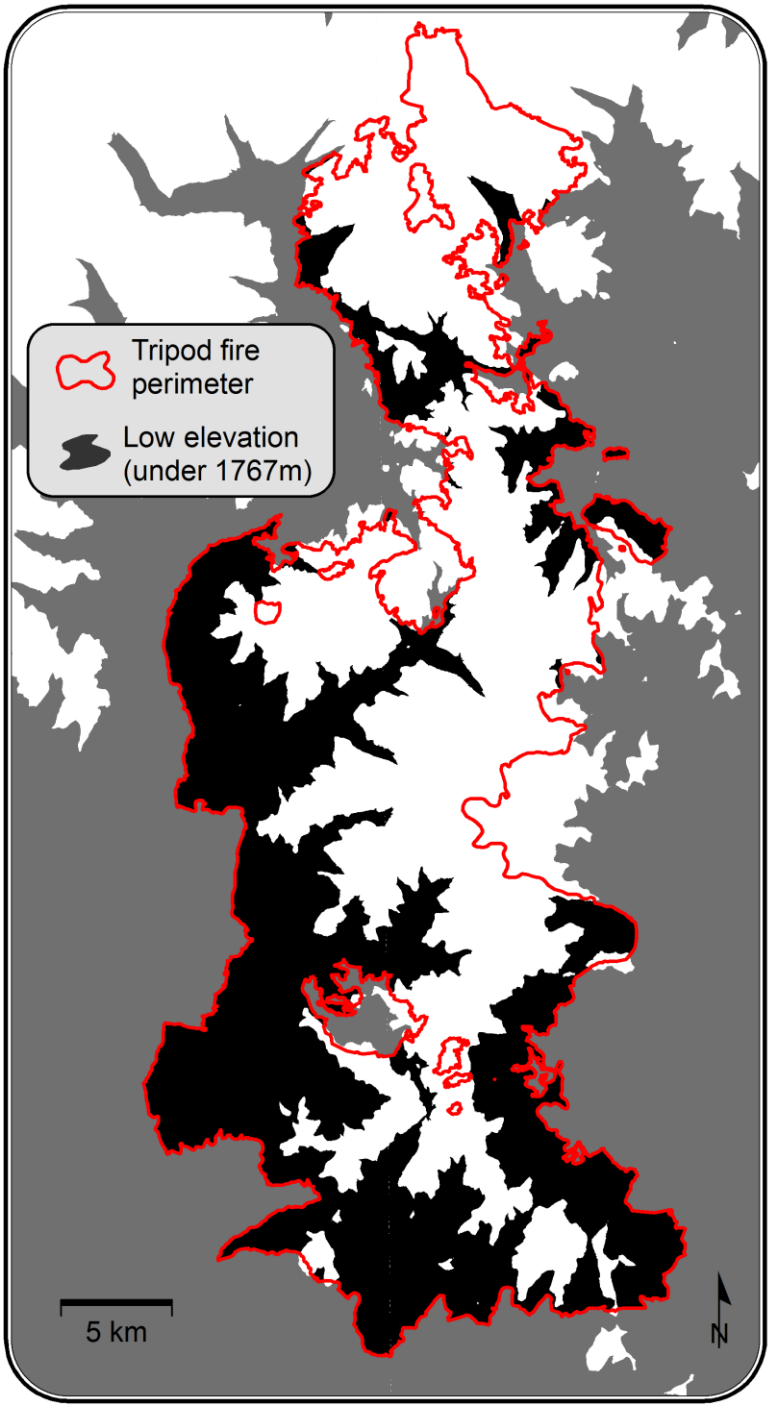
[3] Value represents the mean of 19 paired treated/untreated sites

[4] HF100/HF50 (High-intensity wildfire simulated within the next 100 years / 50 years)



Map credit: R. Norheim

Figure 1. Study area map and treatment locations within the Tripod Fire.



Map credit: R. Norheim

Figure 2. Potential fuels management area in ponderosa pine/Douglas-Fir forests within the Tripod Fire perimeter.

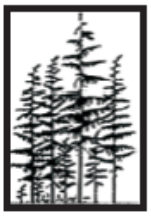
Stratum		Category
CANOPY		Trees, snags, ladder fuels
SHRUBS		Primary and secondary layers
NONWOODY VEGETATION		Primary and secondary layers
WOODY FUELS		All wood, sound wood, rotten wood, stumps, and woody fuel accumulations
LITTER-LICHEN-MOSS		Litter, lichen, and moss layers
GROUND FUELS		Duff, basal accumulations, and squirrel middens

Figure 3. FCCS fuelbeds by strata and categories.

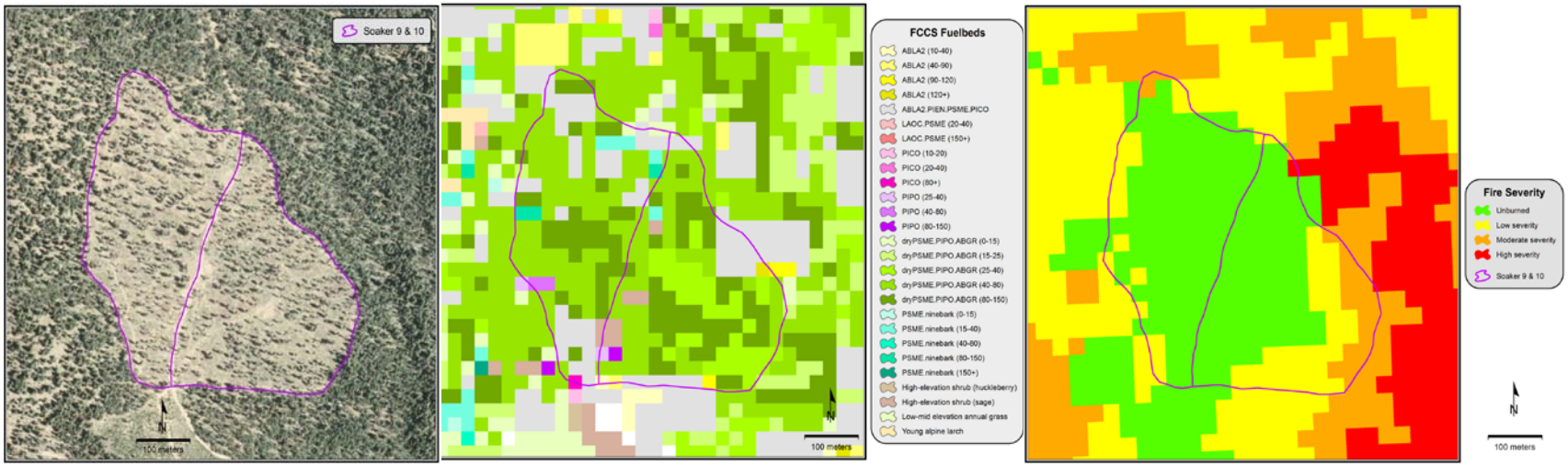


Figure 4. (*left*) Example fuel treatment within the Tripod Fire: National Agricultural Imagery Program (NAIP), 1-m resolution aerial imagery. (*center*) Pre-treatment FCCS fuelbeds, 25-m resolution. (*right*) MTBS burn severity, 30-m resolution.

Species codes for key FCCS fuelbeds (stand age in parentheses):

ABLA: *Abies lasiocarpa*
PIEN: *Picea engelmannii*
PSME: *Psuedotsuga menziesii*
PICO: *Pinus contorta*
LAOC: *Laryx occidentalis*
PIPO: *Pinus ponderosa*
Ninebark: *Physocarpos capitatus* (shrub)

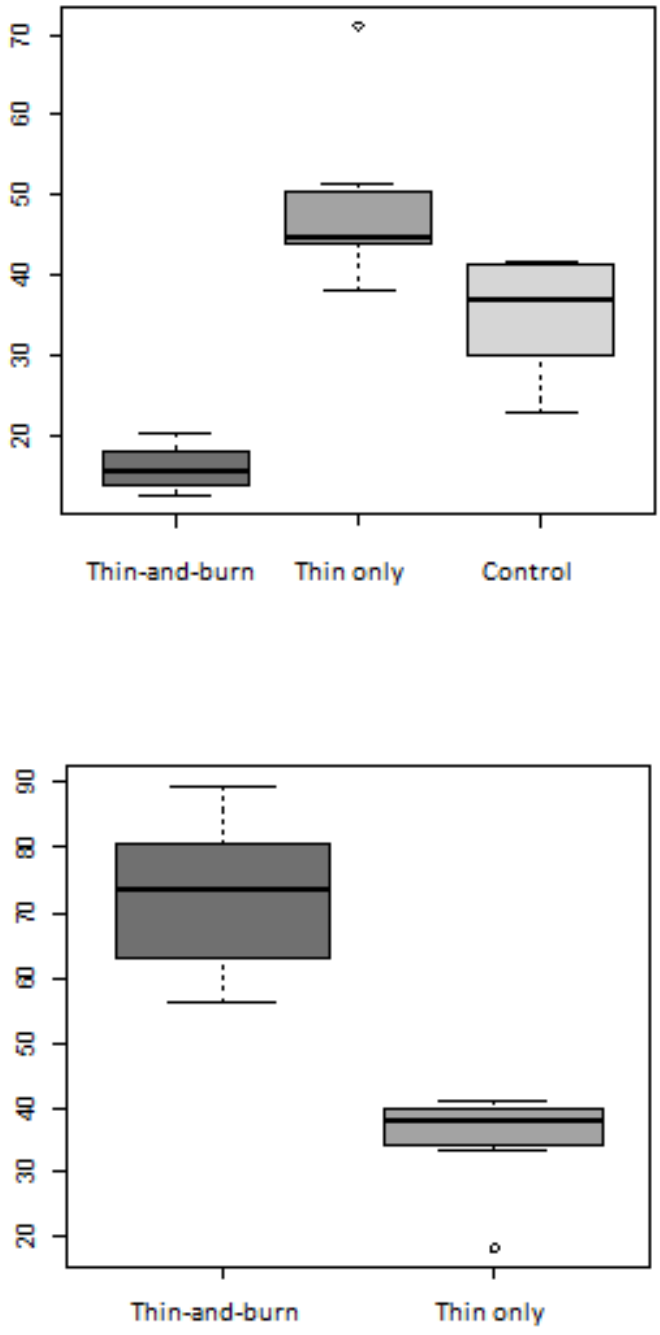


Figure 5. Treatment effects on wildfire emissions and initial treatment C costs (Mg C ha^{-1}). Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper.

University of Washington

Abstract

SECTION 3

Effects of fire and fuel treatments on carbon dynamics in dry temperate forests, eastside Cascade Range, Washington, USA

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Sequestration of carbon in forests has the potential to mitigate effects of global climate change by offsetting future emissions and greenhouse gas (GHG) concentrations in the atmosphere. In dry temperate forests, however, wildfire is a natural disturbance agent with the potential to release large fluxes of carbon into the atmosphere. Climate-driven increases in wildfire frequency, extent, and severity are expected to increase the risks of reversal to carbon stores and affect the potential of dry forests to sequester carbon. Fuel treatments that successfully reduce surface fuels in dry forests can mitigate the spread and severity of wildfire by reducing both tree mortality and emissions from wildfire. However, uncertainty in future wildfire frequency and extent complicate assessments of long-term (decades to centuries) carbon dynamics. We quantify

trade-offs between fuel treatments and carbon storage by simultaneously simulating a broad range of treatment intensities and wildfire frequencies over a 100-year period. We assessed the sensitivity of carbon stores to different levels of treatment intensity and potential changes in the disturbance regime to characterize how carbon stocks might respond to a changing climate in dry temperate forests in the eastside Cascade Range of Washington. The simulation results suggest that we should expect carbon stores to decrease, regardless of treatment level, as wildfire frequency increases. The simulations also show that carbon stability increases proportionally to treatment intensity. That no level of treatment recouped treatment-related carbon costs via reductions in wildfire emissions suggests that the primary long-term benefit to carbon storage from fuel treatments may be to stabilize carbon stores and reduce the likelihood of extreme carbon losses, such as vegetation life-form conversion.

3.1 *Introduction*

Forest structure, fuel characteristics, and fire regimes in millions of hectares of dry forest ecosystems in the western United States (US) have been significantly altered in the past century (Graham et al., 2004). Legacies of fire suppression, fire exclusion, grazing, and timber harvest have resulted in widespread accumulation of surface and canopy fuels and, in turn, have increased the probability of severe and extensive wildfires when compared with pre-settlement forests under natural disturbance regimes (Stephens 1998). Alteration of fire regimes is greatest in arid and semiarid forests, primarily forests dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) or both, which formerly had more frequent and lower severity wildfires than today (Agee 1993). Fire exclusion and fire suppression have contributed to increases in overstory stem density and growth of shade-tolerant, fire-intolerant vegetation; and the quantity, vertical arrangement, and horizontal continuity of forest fuels (Agee 1993).

In addition to contributions to fuel accumulation from past management practices, wildfire occurrence and extent are substantially controlled by climate and are expected to continue to increase under global warming scenarios (McKenzie et al., 2004; Littell et al., 2009). Climatic conditions favorable to large wildfire events, such as seasonal droughts and longer fire seasons, are also expected to become more widespread (Westerling et al., 2006). Climate-driven increases in wildfire frequency, extent, and severity (McKenzie et al., 2004, Westerling et al., 2006) are, in turn, expected to affect the potential of forest ecosystems to sequester carbon (C) (Deal et al., 2010). Changes in disturbance regimes affect forest age-class structure as well as forest dynamics, and are expected to affect C budgets (Kurz et al., 1995). Moreover, wildfires emit carbon dioxide (CO_2) and other greenhouse gases (GHG) into the atmosphere and act as a

positive feedback that may exacerbate effects of climate change (IPCC 2007).

Use of mechanical thinning and prescribed fire to reduce surface fuels has become increasingly common in dry forest ecosystems in western North America (Covington et al., 1997; Agee and Skinner, 2005). The modification of potential fire behavior with fuel treatments is now a central management focus in most coniferous forests in the western US (Stephens et al., 2009). Beyond wildfire risk reduction and increased resilience to disturbance, fuel treatments can in some cases increase wood production in residual trees (Thornley and Cannell, 2000) and control stand structure and composition (McDowell et al., 2006), and may also provide ancillary benefits to ecosystem services such as air quality, water quality, wildlife habitat, and C sequestration (Deal et al., 2010).

This study reports simulated treatment and wildfire effects on C dynamics at the forest stand scale in the eastside Cascade Range, USA. To complement previous work, simulations were conducted in 5 stands in the Okanogan-Wenatchee National Forest (OWNF) selected to be comparable to pre-treatment stand conditions within the treatment units located within 2006 Tripod Fire (Prichard et al., 2010). The Fire and Fuels Extension to the Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Crookston, 2003; Dixon 2008, Reinhardt et al., 2008) was used to simulate treatment and wildfire effects on forest C pools. We simultaneously explored a broad range of treatment intensities, and stochastically modeled wildfire at multiple frequencies over a 100-year simulation period.

3.2 Methods

We used the East Cascades variant of FFE-FVS (Reinhardt and Crookston, 2003; Dixon 2008, Reinhardt et al., 2008) to model C dynamics across a range of management and

disturbance scenarios. FVS is the most widely used growth and yield model in the US, and is accepted by the Chicago Climate Exchange and California Climate Action Registry for growth and yield modeling pertaining to forest C sequestration (Reinhardt and Holsinger, 2010). FFE-FVS simulates fire and fuel dynamics over time by integrating the FVS stand development model (Wykoff et al., 2008), Rothermel's (1972) fire behavior model, and the First Order Fire Effects Model (FOFEM) (Reinhardt et al., 1997) to predict tree mortality, fire behavior, fuel consumption, and C emissions for both prescribed and wildland fire (Reinhardt and Holsinger, 2010).

FFE-FVS modeling inputs were selected from the Forest Inventory and Analysis (FIA) database (http://www.fia.fs.fed.us) to best represent pre-treatment stand conditions observed within the 2006 Tripod Fire (Prichard et al., 2010) (Table 1). Five stands from OWNF were selected based on overstory composition, basal area, tree density, and proximity (elevation, distance) to the Prichard et al. (2010) study area. As a result of the restriction of stand selection, specific inferences target the Tripod Fire region, but general principles of the simulation design may apply to other dry temperate forests with a structural legacy of fire suppression in the western US.

To capture a range of potential future wildfire frequency and management intensity in the study domain, 16 scenarios were detailed in FVS: four levels of wildfire frequency (none, low, moderate, high) and four levels of management intensity (none, low, moderate, high) (Figure 1). Management scenarios were determined based on residual basal area targets for thinning regimes (low = 23 $m^2 ha^{-1}$, moderate = 18 $m^2 ha^{-1}$, high = 13 $m^2 ha^{-1}$) and prescribed fire intervals (low = no prescribed fire, moderate = 20 years, high = 10 years). Thinning from below to the desired residual basal area occurred in the first year of the simulation period. Additional thinning events

were simulated at 40-year intervals to maintain residual basal area targets for each management scenario. If the basal area of a given stand failed to exceed the basal area target for the scheduled treatment, no thinning was simulated. Similarly, if prescribed fire events were scheduled to occur within 20 years (moderate) or 10 years (high) of a stochastic wildfire event, that scheduled prescribed fire event was removed from the simulation design.

Dendrochronologically derived fire histories from the study region were used to determine simulated wildfire frequencies (Ohlson 1996, Everett et al., 2000, McKenzie et al., 2006). “Low” wildfire frequency corresponds to a mean fire-free interval (MFFI) equivalent to the fire suppression period measured in the OWNF by Everett et al. (2000) (MFFI = 38.3 years). “Moderate” wildfire frequency (MFFI = 19.1) approximates a doubling in wildfire events relative to the fire suppression period. “High” wildfire frequency (MFFI = 9.0 years) reflects an estimate of pre-settlement MFFI estimated from fire scar records at low elevations in the OWNF. “No treatment” and “no wildfire” scenarios were also included in the scenario matrix.

To stochastically simulate wildfire events in FVS, a Poisson distribution was used to translate future fire frequencies (MFFI) into discrete wildfire events on the simulation timeline. The Poisson probability model adequately approximates random processes, such as wildfire ignition (Mandallaz and Ye, 1997), and is characterized by the counting of individual events over space and time (Sundararajan 1991; Wiitala 1999). Given a fixed simulation length (100 years) and predetermined wildfire frequencies (MFFI = 38.3, 19.1, 9.0 years), we invoked a Poisson random sample without replacement to generate lists of years in which wildfire is simulated for each scenario. For each wildfire, estimates of fire behavior, tree mortality, fuel consumption, and C emissions are generated. Ten replicate Poisson-derived wildfire scenarios were generated in order to capture a range of future wildfire activity within each frequency level (Table 2).

Live tree (stems, branches, and foliage), dead tree (stems, branches, and foliage), forest dead and down wood (all woody surface fuel regardless of size), forest floor (litter and duff), herb, and shrub C stocks are calculated by FVS for each year of the simulation. Carbon is transferred among pools as trees and other vegetation grow and die, or are removed through a prescribed thinning, and as dead material decomposes. Fires consume portions of each C pool and promote the transfer of C among pools. We used the C calculations from FFE-FVS to evaluate trade-offs and interactions between management and wildfire.

3.3 Analysis

The simulation experiment fits a randomized block design with two treatment blocks (management intensity and wildfire frequency) of four levels each (none, low, moderate, and high). Permutational multivariate analysis of variance (PERMANOVA) (Anderson 2001) was used to test for differences in C dynamics between the four levels of wildfire frequency and the four levels of treatment intensity. PERMANOVA was used because assumptions required for traditional analysis of variance (ANOVA) (i.e., normality of distribution, homogeneity of variance) were not satisfied for many response variables. Treatment effects were measured relative to a control, or no treatment scenario, only within each level of wildfire frequency. Comparisons across frequency levels were not considered. Post-hoc pairwise comparisons were made at $\alpha = 0.05$. All tests (permutations = 999) were conducted using the "Euclidean" distance measure in the R programming environment (R Development Core Team, 2012). Because treatment effects are simulated (not observed), statistical significance is to some extent an artifact of the performance of FFE-FVS. However, the identification of patterns and relative differences in C dynamics between multiple future scenarios creates an opportunity to expand and refine the

temporal context in which assessments of managing for C storage are considered.

3.4 *Assumptions*

To capture the range of variability in the long-term C response to wildfire frequency, some key ecological variables were modeled as constants in FFE-FVS. Seedling regeneration was modeled following wildfire using the default partial establishment model for the East Cascades variant of FVS and augmented with seedling densities published in the full establishment model for the Northern Idaho/Inland Empire variant of FVS (Keyser, 2008). Numerous factors (i.e., burn severity, proximity of seed source, seed predation, and climate) affect the post-fire regeneration response of conifers in the study area with diverse effects on seedling establishment (Savage and Mast, 2005). The regeneration model did not attempt to characterize variability in natural regeneration, nor did it consider the persistence of long-term effects of fire on the seedling environment. The frequent occurrence of wildfire in our experimental design most likely tempers any effect that variable regeneration rates would have on model outputs. However, potentially significant effects from regeneration rates on long-term C dynamics would be expected during extreme climate events, such as prolonged drought. Future research is needed to quantify the sensitivity of C to post-fire regeneration rates.

Climate and weather variables were also assumed remain constant throughout the simulation period. At fine spatial and temporal scales, climate directly affects photosynthesis and respiration rates, nutrient dynamics, and productivity through impacts on decomposition rates, soil moisture, and soil temperature (Littell et al., 2008; Lo et al., 2011). At broader scales, climate patterns, such as those that are driven by ocean-atmosphere interactions (i.e., El Niño Southern Oscillation (ENSO), Pacific Decadal Oscillation (PDO)) have been identified as key

drivers of productivity (Fagre et al., 2003) and regional disturbance regimes (Heyerdahl et al., 2008). Moreover, climate change is expected to have pronounced ecological consequences in forested ecosystems with affects to tree growth, mortality, and regeneration (Crookston et al., 2010). In this study, fire frequencies were designed to provide a proxy for potential climate effects on disturbance, yet no attempt was made to model changing climate directly.

Environmental variables (i.e., temperature, wind speed, fuel moisture) for both prescribed and wildfire events were held constant for the duration of the simulation period (Table 3). Environmental variables for simulated prescribed fires were derived from spring prescribed burning conditions commonly used for fuels management in the OWNF (Harrod et al., 2009). A narrow range of environmental conditions must be satisfied prior to the application of prescribed fire to ensure safety and to meet ecological objectives. As a result, the assignment of constant, regionally specific, environmental variables for simulated prescribed fire should adequately approximate expected burn conditions. For any given wildfire, however, environmental variables are characterized by substantial interannual, seasonal, and diurnal variability, which correspond to varied levels of burn severity. To standardize inferences of the C response to wildfire frequency, our simulation design holds environmental variables for wildfire constant. This effectively allows simulated fire behavior and resultant C dynamics to be a result of stand structure (i.e., treatment factor), wildfire frequency, or an interaction of the two factors. Environmental variables for wildfires (Table 3) were derived from values measured during the Tripod Fire (Evers et al., 2006) and are consistent with values reported for “very dry” conditions in the East Cascades variant of FFE-FVS (Keyser and Dixon, 2008). Changes in environmental variables has the potential to significantly affect the severity at which wildfire occurs. Additional research is necessary to identify the sensitivity of C to specific environmental variables.

The interaction of multiple disturbances (i.e., insect outbreaks) was not considered in this study. For example, western spruce budworm (*Choristoneura occidentalis* Freeman) and mountain pine beetle (*Dendroctonus ponderosae* Hopkins) are prominent disturbance agents in the OWNF, including within the domain of inference for this study. The sensitivity of C to insect outbreaks and the interaction of insect outbreaks with wildfire is potentially another critical component of long-term patterns of C dynamics in dry forest ecosystems.

3.5 *Results*

Total C release associated with treatments was calculated as the sum of C removed during harvests and C emissions from prescribed fires. Mean quantities of C removed during harvests increased proportional to treatment intensity. "Low" treatments removed an average of 53.4 Mg C ha^{-1}, "moderate" treatments removed 57.6 Mg C ha^{-1}, and "high" treatments removed 66.7 Mg C ha^{-1} (Table 4). "Moderate" frequency prescribed fire (f = 20 years) totaled an average of 57.4 Mg C ha^{-1} in C emissions, while "high" frequency prescribed fire (f = 10 years) totaled an average of 74.3 Mg C ha^{-1} in C emissions (Table 5). Accounting for C stored in long-lasting wood products and energy requirements for thinning operations, total C release associated with the application of fuel treatments (harvest + prescribed fire) summed to 32.0 Mg C ha^{-1} ("low"), 87.7 Mg C ha^{-1} ("moderate"), and 109.9 Mg C ha^{-1} ("high") (Table 6).

C emissions from wildfire decreased monotonically with treatment intensity in each level of wildfire frequency (Figure 2). Mean wildfire C emissions ranged from 17.5 to 84.2 Mg C ha^{-1} ("low"), 41.9 to 133.2 Mg C ha^{-1} ("moderate"), and 56.0 to 166.5 Mg C ha^{-1} ("high"). Treatment level effects on wildfire C emissions were also measured relative to a control, or no treatment scenario, in each level of wildfire frequency. Wildfire C emissions increased significantly in

thin-only or "low" treatment levels ($p < 0.05$) and were reduced in treatment levels that included prescribed fire (i.e., “moderate”, “high”) (Figure 3). The magnitude of reduction in wildfire C emissions increased with treatment intensity, and treatment effects were more pronounced as wildfire frequency increased, but total C release associated with the application of treatments exceeded reductions in wildfire C emissions in all scenarios (Table 7).

Changes (t_{100} - t_1) in C stored in live trees, dead trees, and surface fuels are also reported by FFE-FVS. Standing dead C increased significantly ($p < 0.05$) in “low” and “no” treatment scenarios at all wildfire frequency levels. Treatment levels that included prescribed fire (i.e., “moderate”, “high”) showed the largest reduction in C stored in standing dead trees (Figure 4). Mean changes in the magnitude of C stored in standing dead trees ranged from -16.5 to 16.2 Mg C ha^{-1} (-172% to +169%) (“no”), -6.2 to 19.1 Mg C ha^{-1} (-65% to +199%) (“low”), -4.3 to 12.3 Mg C ha^{-1} (-44% to +128%) (“moderate”), and -8.3 to 17.8 Mg C ha^{-1} (-86% to +185%) (“high”) depending on wildfire frequency level. Significant treatment effects were identified at each wildfire frequency level ($p < 0.05$). Pairwise comparisons revealed no significant differences between “moderate” and “high” treatments at any wildfire frequency level. “Low” treatment was significantly different from “no” treatment ($p < 0.05$) at “no” and “low” wildfire frequency levels, but not at higher wildfire frequency (Figure 4).

Changes (t_{100} - t_1) in mean of surface fuel C storage ranged from -2.4 to 60.9 Mg C ha^{-1} (-9% to +230%) (“no”), -1.6 to 29.3 Mg C ha^{-1} (-6% to +111%) (“low”), -4.0 to 5.8 Mg C ha^{-1} (-15% to +22%) (“moderate”), and -7.0 to -3.9 Mg C ha^{-1} (-27% to -15%) (“high”). Treatment levels that included prescribed fire (“moderate”, “high”) were the most effective at stabilizing C stored in surface fuels over time, but the effect was not significant at “moderate” and “high” wildfire frequencies (Figure 5). Pairwise comparisons identified treatment effects at “no” and

"low" wildfire frequency levels ($p < 0.05$), suggesting that the magnitude of treatment effects on C stored in surface fuels increases as wildfire frequency decreases (Figure 5).

Changes ($t_{100} - t_1$) in mean live tree C storage ranged from -43.1 to 91.6 Mg C ha^{-1} (-26% to +55%) ("no"), -15.5 to 28.6 Mg C ha^{-1} (-9% to +17%) ("low"), -34.8 to -7.3 Mg C ha^{-1} (-4% to +21%) ("moderate"), and -61.4 to -42.2 Mg C ha^{-1} (-37% to -25%) ("high"). The magnitude of C stored in live trees decreased significantly ($p < 0.05$) in all scenarios with the exception of the scenario characterized by no wildfire and no treatment (Figure 6). However, relative to a control (no treatment) scenario, all levels of treatment at each wildfire frequency resulted in a significant increase of C stored in live trees ($p < 0.05$).

Total aboveground C is calculated as the sum of C stored in live trees, dead trees, surface fuels, shrubs, and herbs. Net C storage is calculated as the sum of all treatment costs and fire emissions (prescribed fire and wildfire) subtracted from the change in total aboveground C($t_{100} - t_1$). Total aboveground C storage and net C storage decrease with both increased treatment intensity and increased wildfire frequency (Figures 7, 8). Relative to a control, "low" intensity treatment scenarios resulted in the smallest relative decrease in C storage in all wildfire frequency levels, but are also characterized by the most standing dead C and C stored in surface fuels. Changes ($t_{100} - t_1$) in mean total aboveground C storage ranged from -61.3 to 171.7 Mg C ha^{-1} (-31% to +86%) ("no"), -19.3 to 77.0 Mg C ha^{-1} (-10% to +39%) ("low"), -42.1 to 4.8 Mg C ha^{-1} (-21% to +2%) ("moderate"), and -35.3 to -75.6 Mg C ha^{-1} (-38% to -18%) ("high"). Frequency level changes ($t_{100} - t_1$) in net C storage ranged from -157.0 to 171.7 Mg C ha^{-1} ("no"), -7.6 to -150.0 Mg C ha^{-1} ("low"), -125.0 to -186.3 Mg C ha^{-1} ("moderate"), and -236.2 to -177.1 Mg C ha^{-1} ("high"). Pairwise comparisons indicate that treatments of any level have positive effects only on total aboveground C storage or net C storage at moderate and high wildfire

frequencies ($p < 0.05$) (Figures 7, 8).

Results from our simulation experiment also show that C emissions and tree mortality per wildfire event decrease significantly ($p < 0.05$) with treatment intensity and wildfire frequency (Figures 9, 10). Frequency level changes in mean C emissions per wildfire event ranged from 5.6 to 26.4 Mg C ha^{-1} ("low"), 5.8 to 15.0 Mg C ha^{-1} ("moderate"), and 5.1 to 13.1 Mg C ha^{-1} ("high"). Frequency level changes in tree mortality (basal area killed) per wildfire event ranged from 1.7 to 7.1 m^2 ha^{-1} ("low"), 1.3 to 4.6 m^2 ha^{-1} ("moderate"), 1.0 to 2.9 m^2 ha^{-1} ("high"). For context, recall that basal area is controlled for each treatment level (13 to 23 m^2 ha^{-1})

For all C variables, the variance observed in replicated simulations each scenario decreased with increasing in treatment intensity. "No" treatment scenarios, regardless of wildfire frequency level, are characterized by variance 8–12 times larger than "high" treatment scenarios. The variance of "low" treatment scenarios (i.e., thin-only) is consistently larger than treatment levels that include prescribed fire. A permutation based test of multivariate homogeneity of group dispersions revealed no significant difference in variance between "moderate" and "high" treatments for most variables of interest ($p < 0.05$). The strong evidence for unequal variance in the simulation design reflects a potential for increased uncertainty concerning future C dynamics in scenarios that do not include prescribed fire.

3.6 *Discussion*

Wildfire poses a significant and increasing risk of reversal to C stores in dry forests. Higher temperatures projected in global warming scenarios are expected to alter the frequency, severity, and extent of natural disturbances, including wildfire (McKenzie et al., 2004, Westerling et al., 2006). Climate-driven increases in wildfire frequency, extent, and severity are,

in turn, expected to affect the potential of forest ecosystems to sequester C, potentially undermining the permanence of C sequestration strategies (Deal et al., 2010). We simulated a broad range of future management and disturbance scenarios to identify the sensitivity of long-term C storage to changes in the dominant disturbance agent for the study area, wildfire. The simulated management intensities may also help to identify the capacity to which we can expect to manage for C storage, given expected changes to the fire regime.

Wildfire affects C storage directly through the combustion of biomass and indirectly through its effects on mechanisms for C uptake (i.e., tree morality and stand structure). As the frequency of wildfire increases, the magnitude of C emissions from each wildfire event decreases (Figure 9). Treatments that include prescribed fire (i.e., “moderate”, “high”) also work to effectively mitigate C emissions from wildfire events. Fuels have less time to accumulate and regeneration has less time to establish vertical continuity in the fuel profile (Peterson et al., 2007). However, results from our simulation experiment show that the factor by which wildfire frequency increases exceeds management-related decreases in C emissions per wildfire event (Figure 9). This produces a net increase in C emissions by wildfire over time as frequency increases.

Similarly, as the frequency of wildfire increases, fires tended to be less severe and stands experience less tree mortality per wildfire event. However, as with the trade-off between C emissions and wildfire frequency, the factor by which wildfire frequency increases exceeds the factor by which tree mortality per wildfire event decreases (Figure 10). As such, the cumulative effects of increased wildfire frequency result in reduced C storage in live trees and an overall reduction in net C storage.

All treatment levels were shown to be effective at increasing the magnitude of C stored

in live trees, relative to a control, or no treatment scenario. Also, at increased wildfire frequencies (i.e., “moderate” and “high”), all treatment levels resulted in the retention of more aboveground C relative to no treatment scenarios, but not at “low” or “no” wildfire frequencies. Treatments that included prescribed fire were also shown to be effective at reducing standing dead C and C emissions from wildfire, and stabilizing surface fuel loads. Moreover, the magnitude of treatment effects on C dynamics increased proportionally to increases in wildfire frequency. However, the reduction in wildfire C emissions did not exceed the amount of C removed during treatment in any scenario, even after accounting for C stored in long-lasting wood products. Thus, no treatment level was observed to have a direct net positive effect on C retained in the system (Table 6). However, treatments indirectly provide other benefits, such as the reduction of risks to wildfire and stabilization of C stores.

Testing for relative differences in the distributions of simulated C response to altered wildfire frequency allows for the identification of treatment effects on C dynamics over time. However, from a management perspective, consideration of the range of simulated results (i.e., the variance) across scenarios is in itself meaningful. Changes in variance have a greater effect on the frequency of extreme outcomes than changes in the mean (Mearns et al., 1997). Variance increases in this simulation experiment reflect increased uncertainty concerning future C dynamics in a given scenario. “No” treatment levels are consistently characterized by variance 10–12 times greater than comparable “high” treatment levels, depending on the variable of interest. Variance in “low” treatment levels exceeds that of “high” treatment levels by 2–8 times. Some amount of variance may be attributable to the small sample size included in the experimental design, but the pattern of substantial variance decrease with increasing treatment intensity is present for all C variables and for all stands. This indicates that untreated stands and

stands with low intensity treatment are more susceptible to extreme C outcomes. Some extreme C outcomes may include widespread overstory mortality, regeneration failure, and vegetation life-form conversion. Other extreme C outcomes may include a persistent increase in C storage, regardless of wildfire frequency or severity, at the expense of increased risks to wildfire beyond the simulation horizon. This suggests that trade-offs exist between the capacity of treatments to reduce risks to wildfire and the stability and magnitude of C stores (Hurteau and Brooks, 2011).

Our results suggest that C stores will decrease, regardless of treatment level, if wildfire frequency increases. Should fire exclusion persist throughout the next century, the C response and the associated risks to wildfire would be characterized by the "no" wildfire scenario. The complete suppression of wildfire in the future may become increasingly difficult, however, with a combination of a warmer climate and the continued accumulation of fuels related to fire exclusion.

3.7 *Conclusion*

Recent studies have shown that untreated stands release more emissions to the atmosphere during wildfire than treated stands (Finkral and Evans, 2008; Hurteau et al., 2008; Stephens et al., 2009; Reinhardt and Holsinger, 2010), and that wildfire emissions increase monotonically as burn severity increases (Campbell et al., 2007; Meigs et al., 2009). However, most studies overestimate the capacity of fuel treatments to mitigate C release from wildfire by neglecting to consider uncertainty in future wildfire frequency. Our experimental design quantifies trade-offs between fuel treatments and C storage by simultaneously modeling a broad range of treatment intensities and wildfire frequencies over a 100-year simulation period. We identified patterns of wildfire emissions and tree mortality, while characterizing the C response

to potential shifts in the fire regime. The results suggest we should expect C stores to decrease, regardless of treatment level, as wildfire frequency increases. Treatment regimes that include prescribed fire, however, may be able to preserve greater amounts of C relative to no treatment scenarios as wildfire frequency increases. We also show that C stability increases proportionally to treatment intensity, and although there is an overall net reduction in C storage, treatments that include prescribed fire effectively decrease the risks to of C release to wildfire and reduce the likelihood of extreme C outcomes, such as vegetation life-form conversion.

The modeling scenarios explored in this study were designed to bracket potential ranges of future forest structure, management activity, and fire occurrence in dry temperate forests of the OWNF. Modeling plausible and diverse scenarios identifies the capacity for managing C storage, given expected changes to the fire regime. Future research is necessary to improve the level of detail in the modeling procedures and expand the population of inference. Post-fire tree regeneration is exceptionally variable and strongly influences forest C recovery. Empirical research on the long-term effects of fire on the regeneration environment would help parameterize models such as FVS. Also, additional emphasis on the spatial scale of treatment and wildfire effects on C storage would improve inferences about long-term C modeling. We quantified stand-scale C responses to wildfire frequency, but characterizing the C response across landscapes may highlight additional patterns related to C dynamics over time.

As a result of the projected decrease in C stores for all future (non-zero) wildfire frequency levels and the failure of any treatment level to recoup treatment C costs with reductions in wildfire emissions, the primary long-term benefit to ecosystem C from fuel treatments in this system may be to stabilize C stores, reduce the likelihood of extreme negative C outcomes, and minimize long-term C losses. Ultimately, multiple objectives are associated

with the implementation of fuel treatments in forest ecosystems. Fuel treatments are management tools that affect air quality, water quantity, and wildlife habitat (Deal et al., 2010). The true effectiveness of fuel treatment applications is a function of their benefits to an array of land management objectives, not simply C sequestration (Hurteau and Brooks, 2011).

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Table 1. Initial stand conditions for FIA stands modeled in FFE-FVS. Standard deviations reported in parentheses.

Stand ID	176	347	777	74	847	Average
Aboveground live	144.0	172.7	175.9	213.1	125.8	166.3 (33.3)
Standing dead	6.1	0.0	4.5	13.3	4.6	9.6 (4.8)
Forest down dead wood	13.1	16.5	17.4	19.5	16.4	16.6 (2.3)
Forest floor	7.4	11.0	10.8	9.7	10.1	9.8 (1.4)
Shrub	1.0	1.1	1.1	1.1	1.1	1.1 (0.1)
Total Aboveground C	171.5	201.3	209.7	256.7	158.0	199.4 (38.4)

Table 2. Schedule of wildfire events for each frequency level. Wildfire events reflect the outcome of a Poisson random sample without replacement.

	Replicate	Wildfire Year																	
Low	1	2040	2056																
	2	2091																	
	3	2032	2062	2073	2112														
	4	2021	2045	2064	2074	2087													
	5	2026	2050	2072	2112														
	6	2080																	
	7	2012	2031																
	8	2030	2035	2081															
	9	2028	2060	2084															
	10	2055	2081																
Moderate	1	2015	2021	2023	2025	2029	2036	2071	2088	2109									
	2	2049	2055	2084															
	3	2041	2070	2089															
	4	2012	2014	2024	2032	2041	2055	2077	2112										
	5	2016	2018	2037	2055														
	6	2017	2035	2049	2061	2103	2110												
	7	2051	2074	2084	2094	2103	2105												
	8	2037	2041	2054	2062	2074	2075	2078											
	9	2035	2041	2054	2062	2074	2075	2078											
	10	2017	2029	2031	2043	2065	2100												
High	1	2023	2024	2049	2073	2082	2095	2097	2110	2112									
	2	2023	2025	2034	2039	2041	2058	2062	2089	2108									
	3	2018	2022	2024	2026	2033	2048	2054	2056	2063	2071	2076	2078	2080	2087	2096	2102	2104	2112
	4	2037	2090	2093	2098	2099	2100	2106											
	5	2048	2052	2068	2079	2082													
	6	2013	2020	2036	2049	2060	2078	2097	2098	2105	2111								
	7	2028	2035	2038	2039	2048	2052	2053	2065	2091	2092								
	8	2018	2021	2072	2104	2110													
	9	2020	2021	2040	2052	2079	2080	2084	2089	2093									
	10	2018	2019	2022	2043	2049	2051	2054	2058	2064	2066	2067	2076	2105	2106	2111			

Table 3. Environmental variables for all simulated wildfire and prescribed fire events.

	Units	Wildfire	Prescribed fire
Duff fuel moisture	%	15	50
1 hr fuel moisture	%	4	8
10 hr fuel moisture	%	4	8
100 hr fuel moisture	%	5	10
1000 hr fuel moisture	%	10	15
Live fuel moisture	%	70	110
Wind speed (6 m above vegetation)	$m\ s^{-1}$	8.9	3.6
Temperature	˚C	21.1	15.6
Stand area burned	%	100	70

Table 4. Mean quantity of C removed during harvest (Mg C ha^{-1}) for all stands and replicates over the 100-year simulation. Standard deviations are reported in parentheses.

Treatment intensity	Wildfire frequency: High	Mod	Low	No
High	67.4 (23.0)	71.5 (23.4)	75.6 (25.3)	52.3 (18.0)
Mod	49.6 (23.0)	56.6 (26.9)	64.3 (30.6)	43.2 (19.5)
Low	38.3 (28.1)	49.7 (36.9)	66.2 (43.3)	76.4 (66.9)
No	--	--	--	--

Table 5. Mean quantity of C emissions from prescribed fire (Mg C ha^{-1}) for all stands and replicates over the 100-year simulation. Standard deviations are reported in parentheses.

Treatment intensity	Wildfire frequency: High	Mod	Low	No
High	74.7 (14.9)	70.6 (15.0)	80.0 (6.4)	71.8 (22.8)
Mod	54.7 (10.0)	57.7 (7.6)	63.1 (6.6)	54.0 (18.3)
Low	--	--	--	--
No	--	--	--	--

Table 6. Mean quantity of total C emissions (Mg C ha^{-1}) from treatment application (harvested C + C emissions from prescribed fire) for all stands and replicates over the 100-year simulation. Total C emissions are corrected for C stored in long-lasting wood products and energy requirements for thinning operations. Standard deviations are reported in parentheses.

Treatment intensity	Wildfire frequency: High	Mod	Low	No
High	110.6 (16.6)	108.2 (19.8)	119.2 (11.6)	102.5 (29.4)
Mod	83.5 (12.7)	89.3 (14.3)	97.8 (15.2)	81.1 (24.9)
Low	24.3 (11.2)	28.9 (14.8)	35.5 (17.3)	40.3 (27.0)
No	--	--	--	--

Table 7. Mean total C emissions from treatment application relative to reductions in C emissions from wildfire (Mg C ha^{-1}). Negative values indicate more C was removed during treatment than gained via reductions in C emissions from wildfire.

Treatment intensity	Wildfire frequency: High	Mod	Low	No
High	-0.2	-16.9	-52.6	-84.5
Mod	-83.7	-37.3	-34.1	-65.4
Low	-98.9	-50.5	-28.9	-18.7
No	--	--	--	--

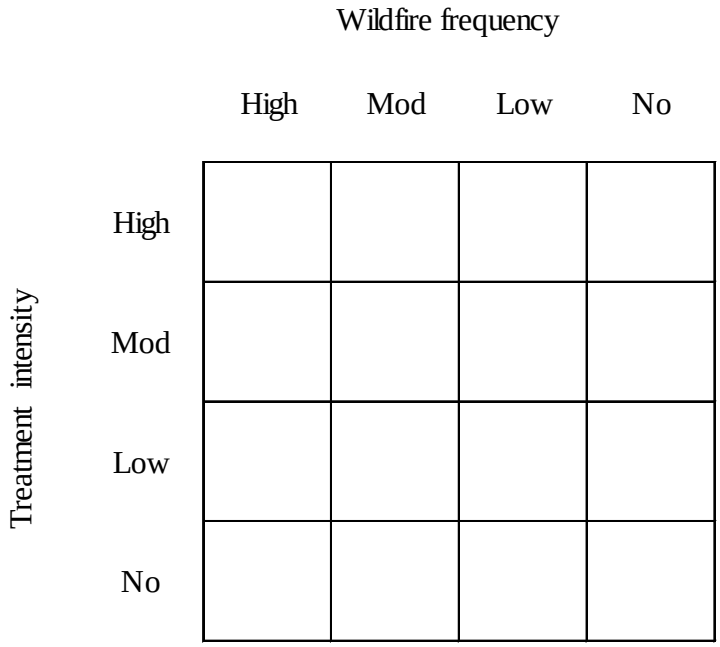

Figure 1. Simulation design (16 scenarios). Wildfire frequency levels reflect a gradient of MFFI (high = 9 years, moderate = 18.2, low = 38.3, no = n/a). Treatment levels reflect (1) residual basal area following thinning from below (high = 13 m^2 ha^{-1}, moderate = 18 m^2 ha^{-1}, low = 23 m^2 ha^{-1}, no = n/a), and (2) prescribed fire frequency (high = 10 years, moderate = 20, low = n/a, no = n/a).

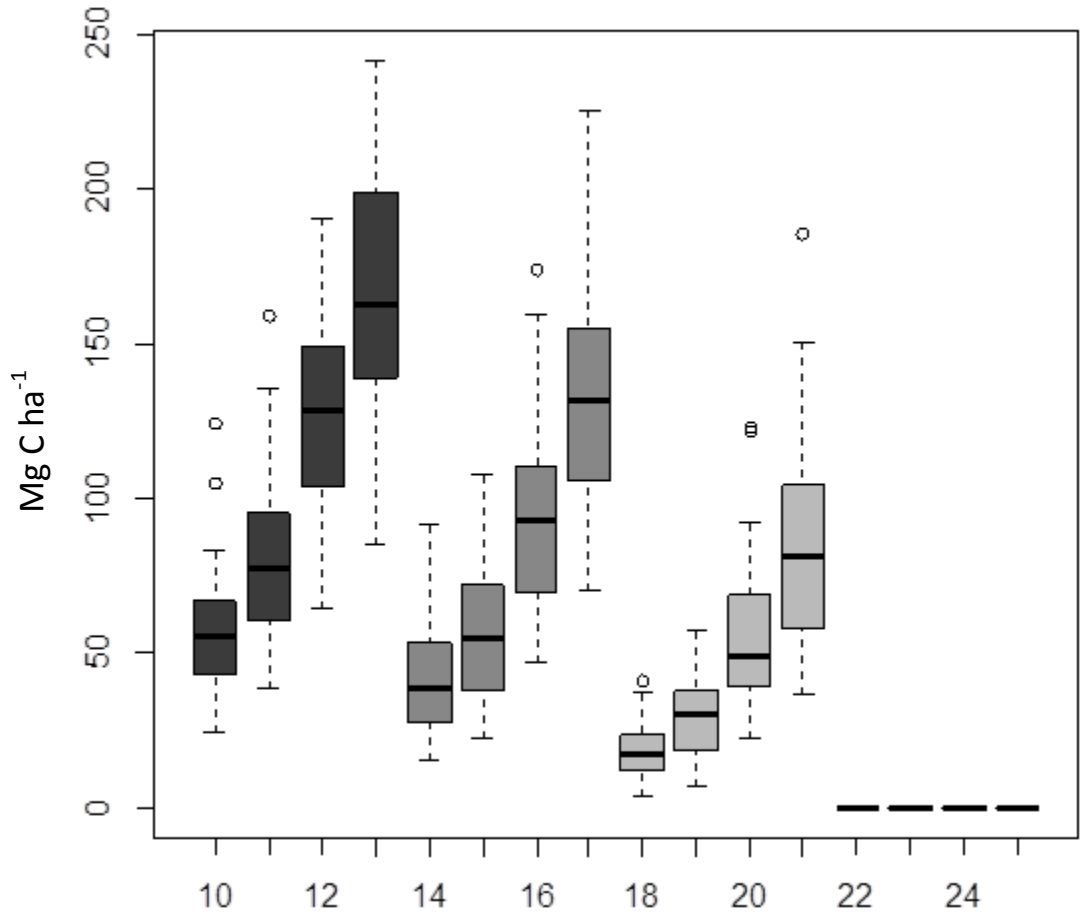

Figure 2. C emissions from wildfire summed over the 100-year simulation period (Mg C ha^{-1}). Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.

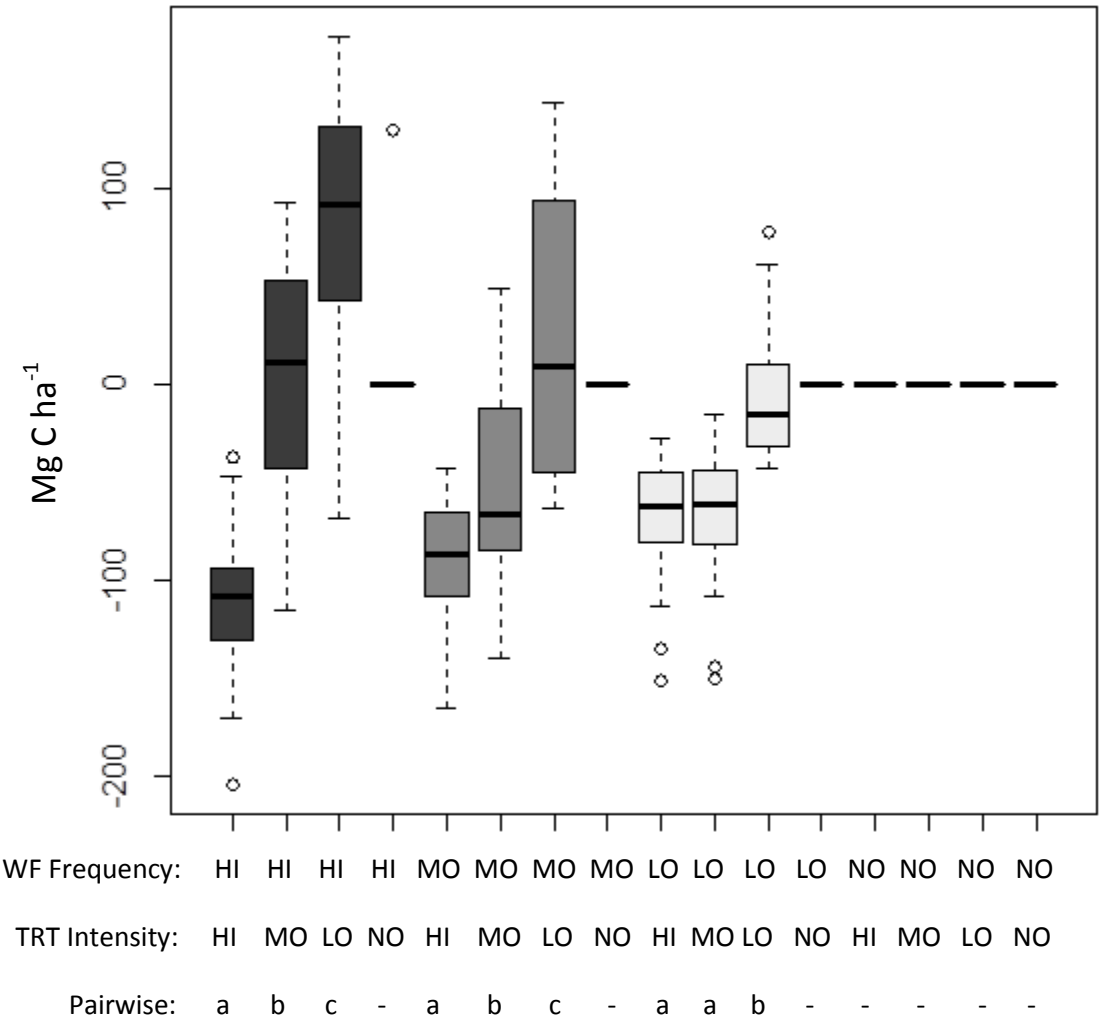

Figure 3. Treatment effects on wildfire C emissions (Mg C ha^{-1}). Values reflect the difference in wildfire C emissions for individual replicates relative to the control replicates for each scenario. Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.

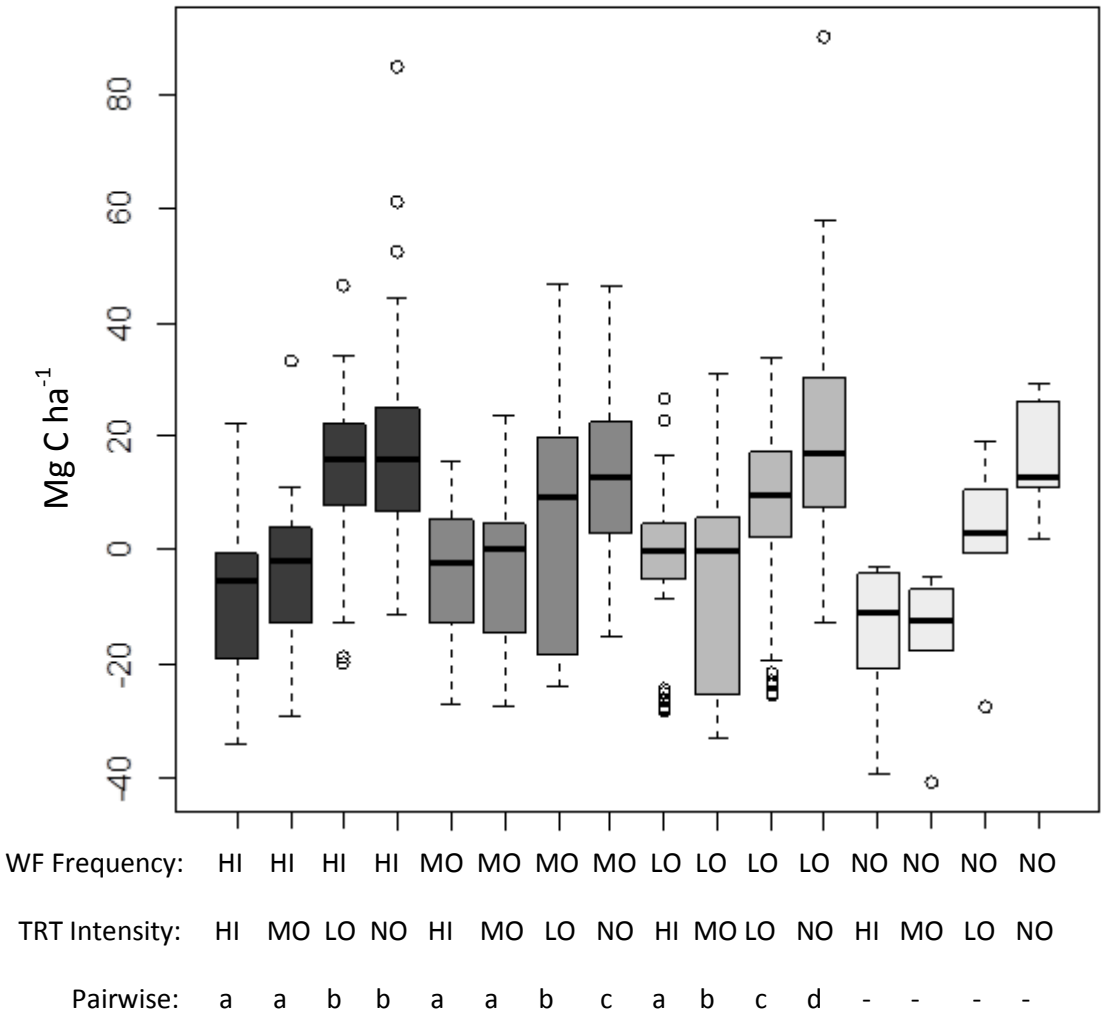

Figure 4. Standing dead C ($t_{100} - t_1$) (Mg C ha^{-1}). Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.

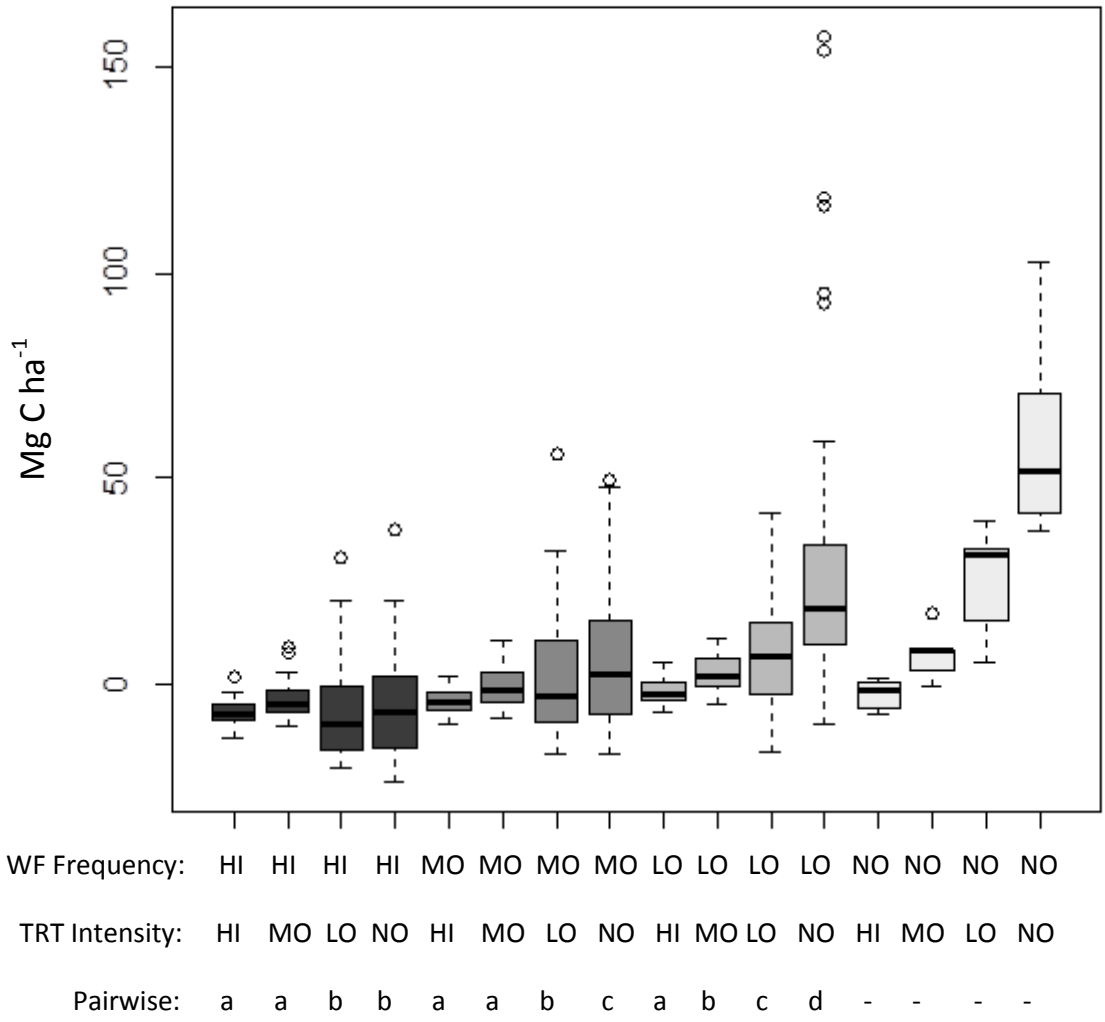

Figure 5. C stored in surface fuels ($t_{100} - t_1$) (Mg C ha^{-1}). Surface fuels reflect the sum of all woody fuels (1 hr – 1000 hr), litter, and duff. Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.

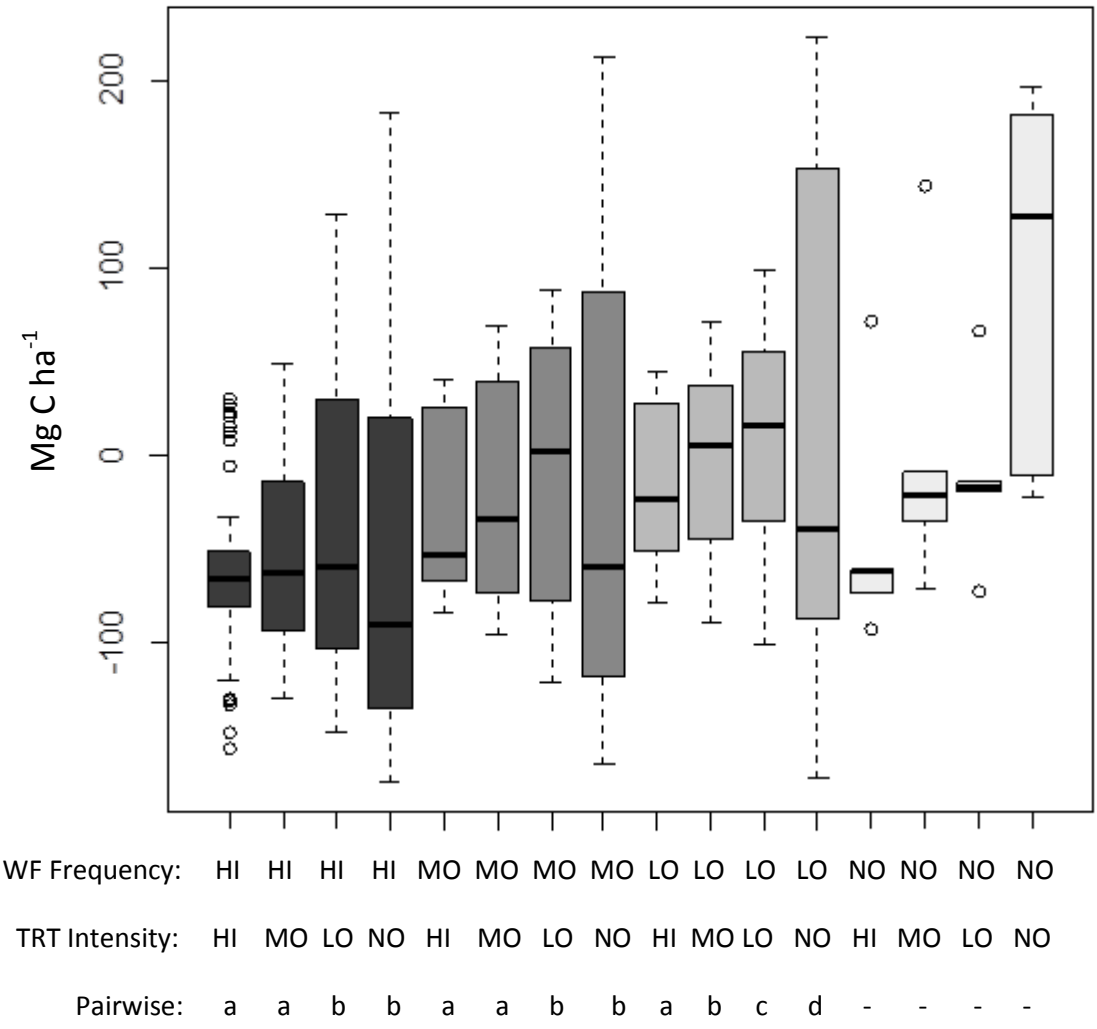

Figure 6. C stored in live trees ($t_{100} - t_1$) (Mg C ha^{-1}). Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.

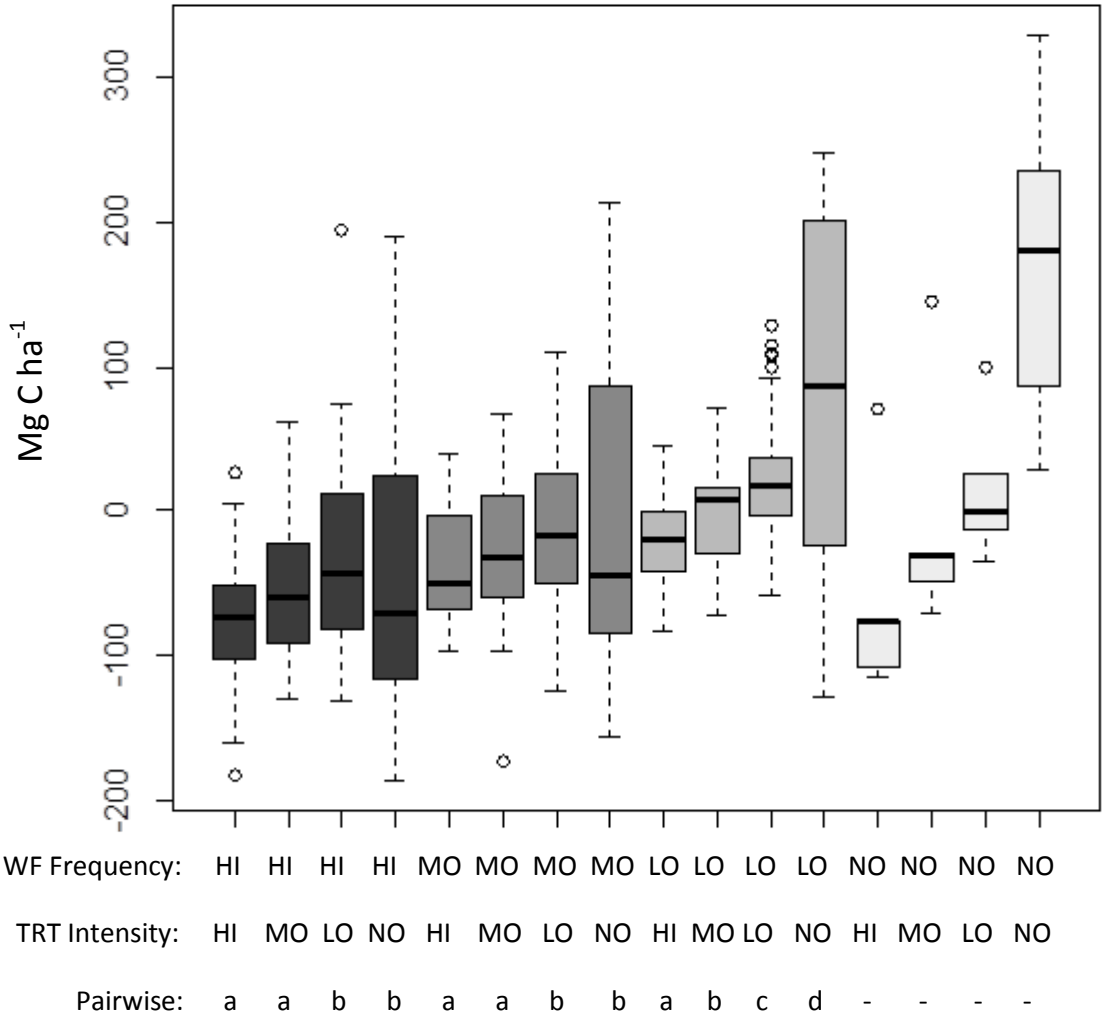

Figure 7. Total aboveground C ($t_{100} - t_1$) (Mg C ha^{-1}). Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.

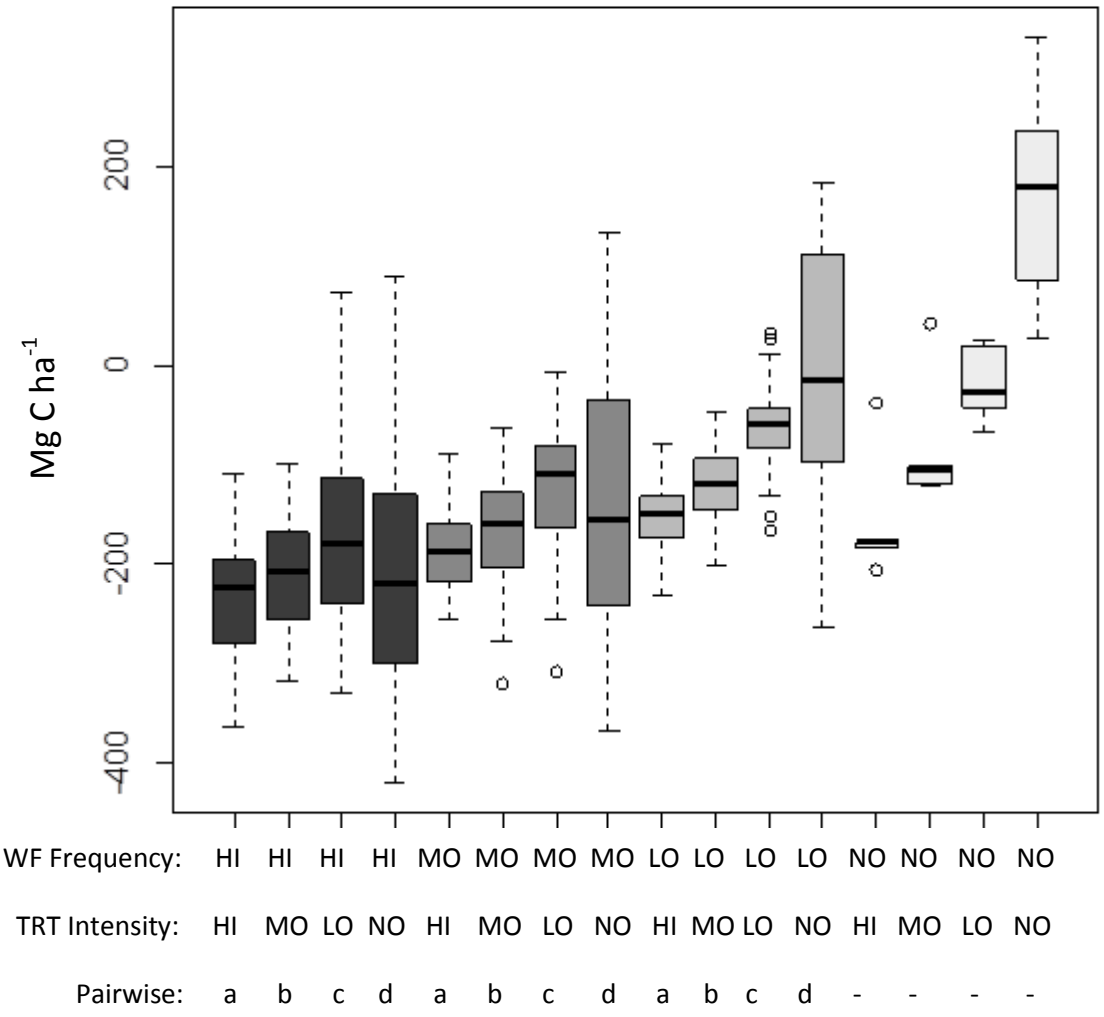

Figure 8. Net C storage (Mg C ha^{-1}) reflects the sum of all treatment costs and fire emissions (prescribed fire and wildfire) subtracted from the change in total aboveground C (t_{100} - t_1). Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.

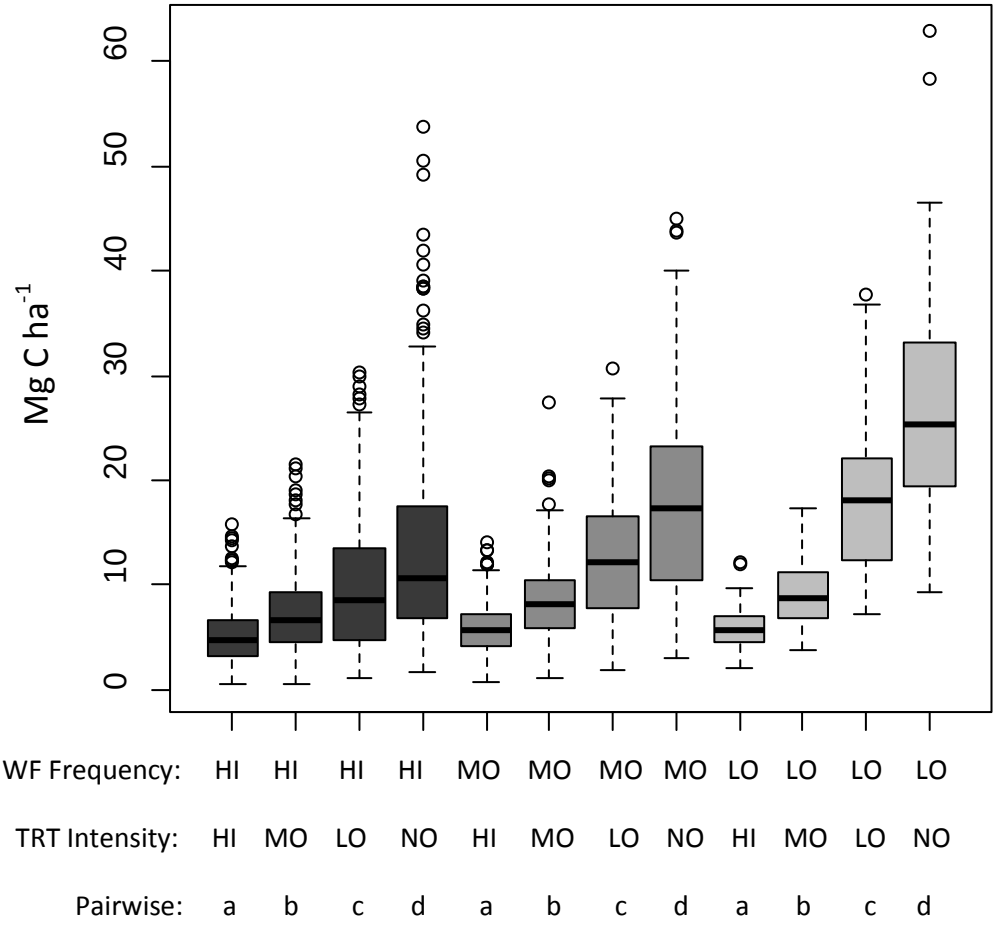

Figure 9. Wildfire C emissions (Mg C ha^{-1}) per event. Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels

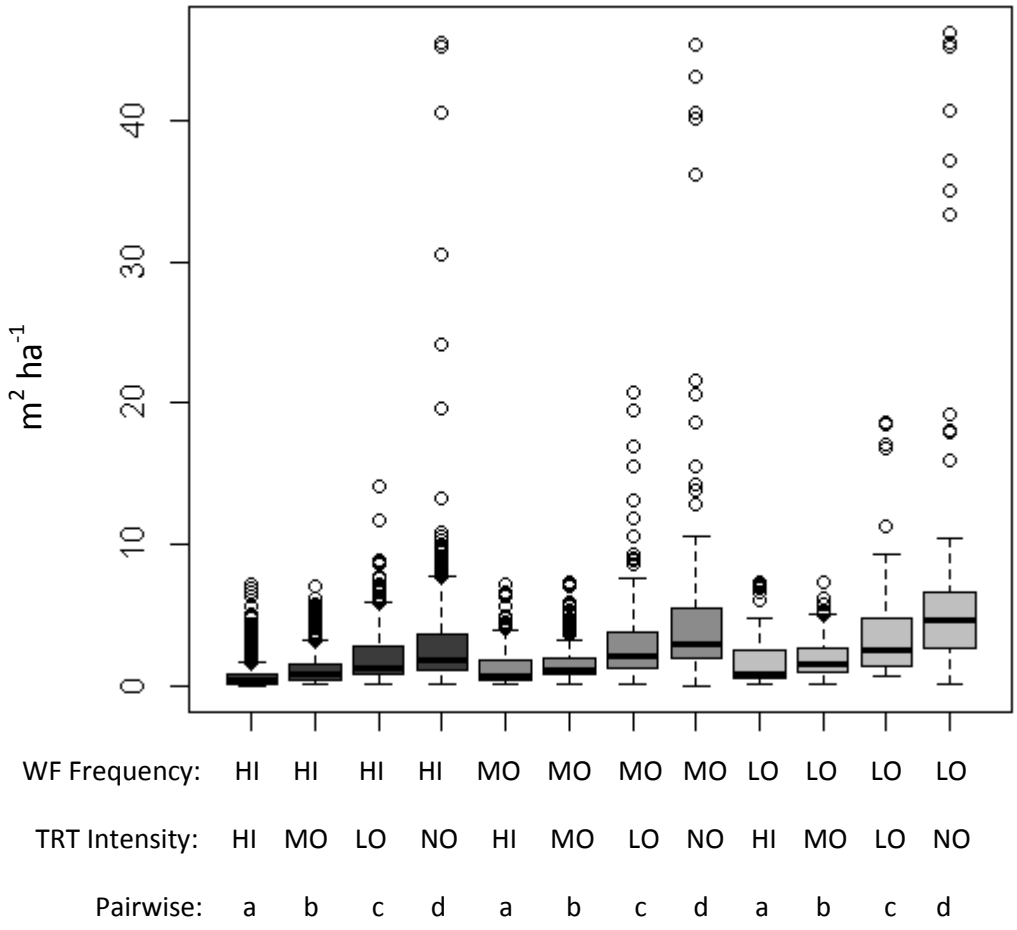

Figure 10. Basal area killed (m^2 ha^{-1}) per wildfire event. Box plots represent minimum, 25% quantile, median, 75% quantile, and maximum values from lower to upper. Pairwise tests indicate significant ($p < 0.05$) distributional differences detected between scenarios within wildfire frequency levels.