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Review

# Wildfire and fuel treatment effects on forest carbon dynamics in the western United States

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ABSTRACT

Sequestration of carbon (C) in forests has the potential to mitigate the effects of climate change by offsetting future emissions of greenhouse gases. However, in dry temperate forests, wildfire is a natural disturbance agent with the potential to release large fluxes of C into the atmosphere. Climate-driven increases in wildfire extent and severity are expected to increase the risks of reversal to C stores and affect the potential of dry forests to sequester C. In the western United States, 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) C dynamics across large landscapes. Results of studies on the effects of fuel treatments and wildfires on long-term C retention across large landscapes are limited and equivocal. Stand-scale studies, empirical and modeled, describe a wide range of total treatment costs (12–116 Mg C $ha^{-1}$) and 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 C effects from fuel treatments are similarly variable (−33 Mg C $ha^{-1}$ to +3 Mg C $ha^{-1}$). Studies at large spatial and temporal scales suggest that there is a low likelihood of high-severity wildfire events interacting with treated forests, negating any expected C benefit from fuels reduction. The frequency, extent, and severity of wildfire are expected to increase as a result of changing climate, and additional information on C response to management and disturbance scenarios is needed improve the accuracy and usefulness of assessments of fuel treatment and wildfire effects on C dynamics.

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## 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. Sequestration of C in forests can mitigate effects of climate change by offsetting future emissions and greenhouse gas (GHG) concentrations in the atmosphere. In the United States (US), forests annually sequester 216–313 Tg C, or the equivalent of 10–20% of fossil fuel emissions in the US (USEPA, 2010).

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 (RGGI), 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), Western Climate Initiative (WCI), and other voluntary trading programs had been established to measure, monitor, and trade GHG emissions. In the US, policy and voluntary market-based programs that encourage C sequestration have contracted from peak trading volumes in 2010. However, emergence of the California Cap and Trade Program, the first domestic compliance C market, is a notable exception to the US market trend (Peters-Stanley and Hamilton, 2012).

The effects of wildfires and management on forest C storage 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 wildfire risk, according to guidelines established in the Kyoto Protocol and CCAR (2007) (Hurteau et al., 2008). Attention to contributions from wildfire emissions and 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 policies that did not recognize the magnitude of C release in wildfire (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 dynamics in dry forest ecosystems. The synthesis is organized by individual elements of C release associated with fuel treatments (e.g., thinning, emissions from prescribed fire and wildfire) and by specific topics (i.e., thinning and wildfire effects on C flux). We compare different approaches for calculating C budgets, effects of assumptions on empirical analysis and modeling, and influence of temporal and spatial scales on inferences.

## 2. Fuel treatments: background and objectives

Forest structure, fuel characteristics, and fire regimes in dry forest ecosystems in the western US have been significantly altered in the past century (Graham et al., 2004). Legacies of fire exclusion, grazing, and timber harvest have resulted in 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 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 has contributed to increased density of overstory trees and shade-tolerant understory trees, and increased quantity and vertical and horizontal continuity of fuels (Agee, 1993).

Wildfire is substantially controlled by climate, and area burned on an annual basis is expected to increase under global warming scenarios (McKenzie et al., 2004; Littell et al., 2009). Climatic conditions favorable to large wildfire events, such as more frequent droughts and longer fire seasons, are also expected to become more widespread (Westerling et al., 2006), and climate-driven increases in extent of fire (McKenzie et al., 2004) will affect the potential of forest ecosystems to sequester C (Deal et al., 2010). Changes in disturbance regimes that affect forest demography and dynamics 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 in forest composition, fuel structure, and fire regimes. Expansion of the wildland–urban interface (WUI) has motivated policy and actions focused on reducing wildfire risks to people and homes (Youngblood, 2005). Specifically, implementation of the National Fire Plan (2001), Healthy Forests Initiative (2003) and the Healthy Forests Restoration Act (2003) has directed new resources to reduce hazardous fuels and restore fire-adapted ecosystems (Winter et al., 2004).

Public and private land managers treat hazardous fuels using mechanical thinning, prescribed fire, and other techniques such as mastication. The primary objective of fuel treatments is to reduce fuel loads (quantity) and change the fuel profile (spatial arrangement) to minimize risk of high-intensity wildfires, with emphasis on reducing surface fuels, ladder fuels, and crown density (Agee and Skinner, 2005). Altering the fuel load enhances sustainable management of vegetation, wildlife habitat, and watersheds; increases the safety of wildland firefighters and people living in the WUI; and reduces suppression costs associated with high-intensity fires (Busby, 2002). Altered forest structure and composition have, in some areas, reduced ecological integrity and resilience to disturbance (Harrod et al., 2007). Fuel treatments can restore structural and functional components in dry forests while simultaneously reducing wildfire risk.

## 3. Carbon and wildfire

Wildfire poses a significant risk of reversal to C stores in dry forest ecosystems, undermining the permanence of C sequestration strategies by releasing C to the atmosphere through combustion and through 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). Wildfires are spatially and temporally variable, and estimates of surface fuel consumption at coarse scales are often uncertain. Fuel treatments can reduce both tree mortality and emissions from wildfire, but heterogeneous burn environments, 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. Because the capacity of forests to sequester C is critical for climate change mitigation, reversal risks to C stores from wildfire must be assessed for accurate C accounting.

The magnitude of the effect of wildfire on net biome productivity depends on 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, on average, 4–6% of annual anthropogenic emissions. However, over several decades, delayed mortality after wildfire and decomposition from fire-killed trees may release up to three times the amount of C to the atmosphere as the fire itself (AuClair and Carter,

1993). Re-growth of vegetation following wildfire may eventually offset C released during decomposition (Fig. 1), but long-term C recovery is a function of the capacity of forests to regenerate in response to the rate, severity, and extent of disturbance (Frolking et al., 2009; McKinley et al., 2011). Decomposition of coarse fuels not directly consumed by fire depends on climate, soil microflora, disturbance severity, and substrate quality (Agee, 1993), and emission of C by decomposition may exceed assimilation of C by vegetation for decades following wildfire. Vegetation response to wildfire is highly variable, but rapid re-colonization by post-fire, non-tree vegetation affects net primary productivity (NPP) after wildfire and harvest (Fig. 2) (Law et al., 2004; Meigs et al., 2009). Vegetation life-form conversion (e.g., forestland to shrubland) following wildfire has 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).

Since 1980, wildfire area burned (WFAB) in the western US has been higher than in the previous 30 years (NIFC, 2010), providing a context for reducing fire hazard through effective fuel treatment programs. Although fuel reduction treatments in dry forests clearly mitigate the spread and severity of wildfire (Agee and Skinner, 2005; Omi et al., 2006; Prichard et al., 2010), 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).

## 4. Fuel treatments and initial carbon loss

Fuel reduction treatments coincide with an initial net C loss to the treatment area (Fig. 2). Thinning treatments reduce standing C stocks via whole-tree removal and also release C through combustion of fuel in logging machinery, transportation of stems and logging residue, slash burning, 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 0.5–3% of total aboveground C stock (Finkral and Evans, 2008; Stephens et al., 2009; North et al., 2009; Hurteau and Brooks, 2011; Winford and Gaither, 2012). Milling waste and emissions from prescribed fire are the largest source of C released in fuel treatments, and though highly variable, can each exceed 20% of post-treatment aboveground C stock (North et al., 2009). Additional C costs vary considerably depending on the fate of 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).

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 20 years or more, treated areas can be overwhelmed by intense fires burning in adjacent areas (Agee and Skinner, 2005). Thus, multiple prescribed fires or mechanical thinnings necessary to meet long-term fuels management objectives (Peterson et al., 2005; Johnson et al., 2007) must be included in analyses of C sequestration.

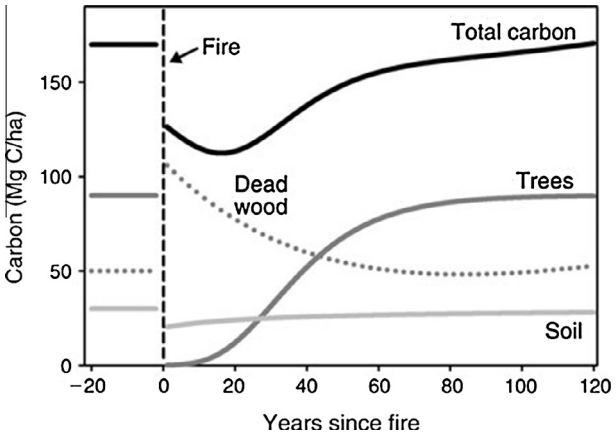



**Fig. 1.** Forest regeneration and C recovery following wildfire. Over time, provided there is regeneration, a forest will recover the C lost from fire and from the decomposition of trees killed by the fire. This concept is illustrated here by showing C stored in forests as live trees, dead wood, and soil and how these C pools change after fire. Model output is from an analysis published in Kashian et al. (2006). Reprinted from McKinley et al. (2011).

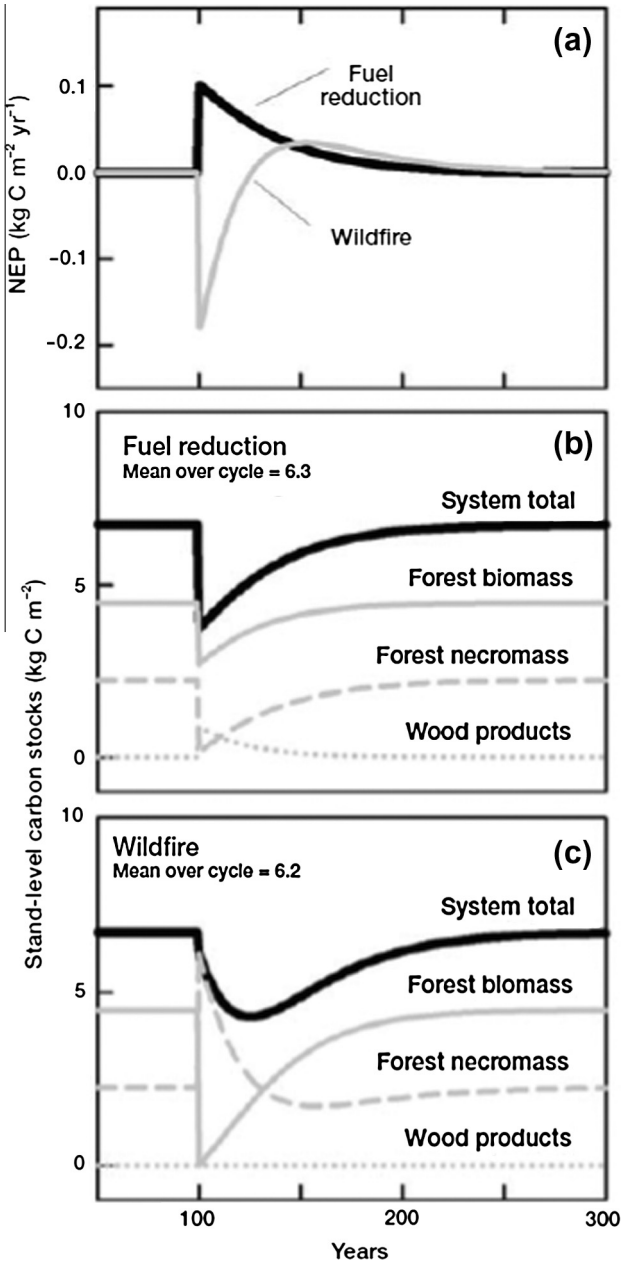



**Fig. 2.** (a) Simulated net ecosystem production (NEP) in a ponderosa pine forest representative of the eastern Cascades and (b and c) C stocks throughout an entire disturbance interval, initiated by either wildfire or fuel-reduction treatment. Unlike the stand subject to fuel reduction via thinning, the combination of low biomass and high necromass after wildfire functions to drive NEP below zero. Nevertheless, although initial losses associated with wildfire were much lower than those in the fuel-reduction treatment, the two scenarios achieved parity in C stocks over the entire disturbance interval. Reprinted from Campbell et al. (2011).

### 4.1. Emissions from equipment usage

Forest operations produce C emissions by burning fossil fuels used to power vehicles and machinery, and fossil fuel consumption depends on the intensity and frequency of silvicultural treatments (Markewitz, 2006). Estimation of C emissions from equipment usage requires knowledge of efficiency of machines (liters of fuel per hour), number of hours per unit of area for a particular activity, and C emitted per volume fuel consumed (Markewitz, 2006). Data sources for equipment usage include logging contractor records, electronic activity recorders on logging equipment, and 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 mean C contents of gasoline and diesel (EPA, 2005) and the molecular mass ratio of $CO_2$ to C (Stephens et al., 2009). Summaries for dry temperate forests (Finkral and Evans, 2008; North et al., 2009; Stephens et al., 2009; Sorensen et al., 2011) indicate that emissions from equipment used to harvest, load, and transport logs during fuel treatments are 0.05–1.20 Mg C $ha^{-1}$, or 0.5–3.0% of post-treatment C storage (Table 1).

Stephens et al. (2009) examined 12 treatment units (14–29 ha) in mixed-conifer forest in the Sierra Nevada, including mechanical, burn only, and mechanical and burn treatments. Mechanical thinning was a moderate to heavy thin from below. Following harvest, 90% of the remaining understory conifers and hardwoods were masticated (Stephens et al., 2009) resulting in equipment release of 0.91 Mg C $ha^{-1}$, less than 1% of post-treatment C storage. The mastication treatment was not conducted in other studies, potentially explaining why equipment emissions surpassed estimates in comparable understory thin units in North et al. (2009).

Six fuel treatments (burn only, understory thin, understory thin and burn, overstory thin, overstory thin and burn, control) were applied in a full-factorial 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, reporting the longest haul distance (235 km) for log transport to the nearest sawmill. 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 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 of 0.07–0.20 Mg C $ha^{-1}$, less than 1% of post-treatment C storage. The authors suggest that flat ground, openness of the stand, and easy working conditions explain the low estimate of C emission.

Sorensen et al. (2011) examined five treatment units in forests in northern Arizona. Silvicultural prescriptions included low thinning and crown thinning of varying intensities. The authors worked with logging contractors to estimate diesel and gasoline consumption of logging machinery used during the harvest (chainsaws, skidders, feller bunchers, processors, forwarders, and tracked fuel consumption associated with employee and equipment transport. C equivalent emissions were 0.05–0.28 Mg C $ha^{-1}$, less than 1% of post-treatment C storage and the smallest component of the C budget in the thinning operation.

Emissions from equipment usage during fuel treatments across all four studies amount to a small percentage of the total aboveground C stock. There is far greater variability and magnitude in treatment-related C emissions from prescribed fire, harvested C, waste associated with milling operations, and decay of wood products.

### 4.2. Emissions from milling waste and decay of wood products

Emissions from milling waste are one of the largest sources of C emissions related to fuel treatments (North et al., 2009). From tree harvest to disposal of wood products from harvested trees, C is lost at each step of the processing chain due to physical breakdown of wood (Ingerson, 2009). Estimates from a California sawmill suggest that 60% of logs is converted to lumber, and the remaining 40% is milling waste (North et al., 2009), which is consistent with other studies (Skog and Nicholson, 2000; Ingerson, 2009; Oneil and Lippke, 2010; data summarized in Stephens et al. (2009)). Large fractions of milling waste in California are converted to energy at cogeneration plants (14–47%) or reconstituted as wood products (e.g., particleboard) with longer life spans for C storage (Skog, 2008; Stephens et al., 2009). Many reconstituted wood products sequester C for decades and comprise 40% of milling waste in California (from Stephens et al. (2009)). Manufacturing of reconstituted wood products converts potential milling waste to longer lasting products at the cost of increased energy inputs and resins (Ingerson, 2009). Milling waste may also recycled (9%) or composted into mulch (8%) (Skog, 2008). Most milling waste (67%) is sent to landfills, although anaerobic conditions coupled with high lignin concentrations in wood and paper make this material resistant to decay (Gower, 2003; Skog, 2008). Alternatively, milling

**Table 1**
Emissions from equipment usage.

| Study | Fuel treatment | Total equipment release (Mg C $ha^{-1}$) | Post treatment C storage (Mg C $ha^{-1}$)[a] | Equipment release as % of post-treatment C storage |
|---|---|---|---|---|
| Finkral and Evans (2008) | Restoration thin | 0.07–0.20[b] | 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 | 0.99 |
| North et al. (2009) | Overs tory thin | 2.94 | 170 | 1.73 |
| North et al. (2009) | Over story 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 |

[a] Includes all aboveground biomass.
[b] Values are presented for multiple wood utilization scenarios (firewood, paper, pallettes/construction).

waste burned as fuel results in immediate release of emissions. Utilization of woody biomass as energy can potentially reduce GHG emissions relative to fossil fuel alternatives (Chen et al., 2000; Perez-Garcia et al., 2006; Oneil and Lippke, 2010). However, the extent to which bioenergy systems in temperate forests mitigate GHG emissions is primarily a function of the displaced energy source (coal, natural gas, and ethanol). C benefits derived from biomass utilization may be overstated if life cycle assessments assume "carbon neutrality" or do not include biomass-based C emissions (McKechnie et al., 2011; Cherubini et al., 2012).

Forest harvesting generally releases 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). Consideration of the end use of wood products is critical to C budgets in managed forests, because decay rates and life spans of forest products vary substantially. Lumber in home construction stores C for 50–100 years, whereas wood used to construct pallets has a median lifespan of 6 years (Skog and Nicholson, 1998). Wood composted as mulch or used as landscaping material has a median lifespan of 5 years (Stephens et al., 2009) (Table 1). Paper and shipping materials decay rapidly in comparison to solid wood (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, and using wood in long-lasting products can produce a net gain in C storage even after considering emissions from future wildfire (Finkral and Evans, 2008). In the inland Northwest, C storage in long-lived (>80 years) wood products exceeds simulated C losses from wildfire, even when accounting for a doubling of historic fire rate as a result of climate change (Oneil and Lippke, 2010).

The highest reported magnitude of milling waste associated with thinning is 18.3–38.2 Mg C $ha^{-1}$ (North et al., 2009). In this study, material harvested during fuel treatment was processed at a sawmill 235 km from the study site. Approximately 40% of logs delivered to the mill resulted in waste, most of which was burned for electricity and sold as landscaping material (North et al., 2009). Bark and mulch by-products generally decompose within 5 years (Stephens et al., 2009), and burned fuel is released immediately to the atmosphere. Overstory thinning treatments are unique to North et al. (2009), and estimates far exceed waste generated from other fuel treatments, a function of the volume of logs processed.

Stephens et al. (2009) report milling waste from fuel treatments in the central Sierra Nevada. 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 treatments. A conversion factor (3.67:1) was used to convert $CO_2$ equivalent to C. Fractions of log mass converted to wood products were determined from sawmill surveys (Morgan et al., 2004; Milota et al., 2005; Stephens et al., 2009), and decay rates for wood products were estimated from historic use data (Winjum et al., 1998; McKeever, 2002; Winistorfer et al., 2005; Skog, 2008). Emissions from milling waste were the sum of biomass burned as fuel and biomass stored in wood products with decay rates of less than 1 year (0.30–0.37 Mg C $ha^{-1}$). Masticated woody material left onsite (7.83 Mg C $ha^{-1}$) may contribute to additional C release if assumed to decay in less than 5 years. Yet, total emissions from milling waste after consideration of masticated material in 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. Moreover, methods for measuring decay rates for C in wood products vary considerably (Lim et al., 1999).

Wood removed during a restoration thinning in northern Arizona was primarily used for firewood and assumed to release emissions of 0.02–0.03 Mg C $ha^{-1}$ within 1 year of harvest (Finkral and Evans, 2008), although the authors offer plausible alternative wood utilization scenarios. Harvested logs processed as paper released the most emissions associated with milling waste (2.47–4.94 Mg C $ha^{-1}$), whereas logs processed as construction materials or pallets generated 0.57–0.85 Mg C $ha^{-1}$ emissions.

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. 6229 ha (9.1%) of a 68,458 ha watershed in southern Oregon met the criteria for thinning from below. The authors used the Forest Vegetation Simulator (FVS) (Dixon, 2008) to estimate wood generated from harvest activities, assuming 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,

**Table 2**
Harvested C and emissions from milling waste

| Study | Region | Fuel treatment | Harvested C | C stored in wood products | C emissions from milling waste | Post treatment C storage[a] | Milling waste as % of post-treatment C storage |
|---|---|---|---|---|---|---|---|
| | | | Mg C $ha^{-1}$ | | | | |
| Finkral and Evans (2008) | Northern Arizona | Restoration thin | 8.24 | 6.11–6.82[b] | 0.02–4.94[b] | 36.42 | 0.05–13.56[b] |
| 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 |

[a] Includes all aboveground biomass.
[b] Values are presented for multiple wood utilization scenarios (firewood, paper, pallettes/construction).

the two are linked because milling waste is directly proportional to the volume of processed material. Emissions from milling waste in thinning treatments associated with fuel reduction ranged from <0.1% to 13.6% of post-treatment C stores across all studies, whereas emissions from overstory thinning 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.

### 4.3. Emissions from prescribed fire: empirical results

Agreement exists across observed and simulated treatments that prescribed fire constitutes a substantial proportion of treatment emissions (Finkral and Evans, 2008; North et al., 2009; Stephens et al., 2009; Sorensen et al., 2011) (Table 3). Prescribed fire is effective at reducing fine surface fuels and horizontal fuel continuity (van Wagtendonk et al., 1996; Graham et al., 2004), but is not reliable for reducing tree density, crown density, or fuel ladders, often used in combination with thinning to achieve management

**Table 3**
Summary of prescribed fire emissions.

| Study | Region | Treatment type | Total emissions from prescribed fire combustion (Mg C ha$^{-1}$) | Post treatment C storage (Mg C ha$^{-1}$)[a] | Prescribed fire emission % of post-treatment C storage | Harvested C (Mg C ha$^{-1}$) |
|---|---|---|---|---|---|---|
| *Empirical studies* | | | | | | |
| Finkral and Evans (2008) | Northern Arizona | Restoration thin | 4.14[b] | 36.42 | 11.4 | 8.24 |
| 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 | 54.72 |
| North et al. (2009) | Sierra Nevada, California | Understory thin | – | 240 | – | 45.81 |
| North et al. (2009) | Sierra Nevada, California | Overstory thin and burn | 27.22 | 110 | 24.7 | 93.68 |
| North et al. (2009) | Sierra Nevada, California | Overstory thin | – | 150 | – | 95.71 |
| 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 | 23.34 |
| Sorensen et al. (2011) | Northern Arizona | Restoration thin | 9.58[b] | 56.41 | 17.0 | 32.47 |
| Sorensen et al. (2011) | Northern Arizona | Restoration thin | 6.39[b] | 51.59 | 12.4 | 15.47 |
| Sorensen et al. (2011) | Northern Arizona | Restoration thin | 3.80[b] | 44.88 | 8.5 | 20.00 |
| Sorensen et al. (2011) | Northern Arizona | Restoration thin | 5.82[b] | 42.37 | 13.7 | 6.11 |
| Sorensen et al. (2011) | Northern Arizona | Restoration thin | 1.01[b] | 57.98 | 1.7 | 8.24 |
| *Simulation studies* | | | | | | |
| Hurteau and North (2009) | Sierra Nevada, California | Burn only | 9–16[c] | 270–360[c] | 3–6[c] | – |
| Hurteau and North (2009) | Sierra Nevada, California | Understory thin and burn | 8–12[c] | 180–200[c] | 4–6[c] | 47.80 |
| Hurteau and North (2009) | Sierra Nevada, California | Restoration burn | 5–11[2] | 230–270[2] | 2–5[2] | 65.00 |
| Hurteau and North (2009) | Sierra Nevada, California | 1865 Reconstruction thin bur | 4–10[c] | 270–330[c] | 1–4[c] | – |
| Ager et al. (2010) | Southeastern Oregon | Understory thin and burn | 19.5 | 54.43 | 35.8 | 14.20 |
| Reinhardt and Holsinger (2010) | Northern Rockies | Thin from below and burn | 12.65 | 35.15 | 36.0 | 10.64 |
| Reinhardt and Holsinger (2010) | Northern Rockies | Burn only | 6.79 | 51.56 | 13.2 | – |

[a] Includes all aboveground biomass.
[b] Values represent emissions from pile burns.
[c] Values from Fig. 1 (Hurteau and North, 2009).

goals (Gorte, 2009). Prescribed fire may consume substantial surface biomass, with smoldering consumption of the organic layer contributing to smoke and affecting soil nutrient cycling (Neary et al., 1999). Prescribed fire can generate fuels by killing understory trees (Agee, 2003), and multiple treatments may be necessary to maintain reduced fire hazard over time.

Finkral and Evans (2008) and Sorensen et al. (2011) measured the dimensions of slash piles created from residual unmerchantable material following harvest, and calculated slash biomass using equations from Hardy (1996). To estimate C mass in the piles, Finkral and Evans (2008) developed size: weight ratios using platform scales to weigh piles. Five fuel reduction treatments produced 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 post-treatment C stores (Table 3).

Stephens et al. (2009) calculated emissions from broadcast burns in the Sierra Nevada as the difference between pre- and post-burn fuel loads measured on site, using the methods of Clinton et al. (2006). Combustion efficiency was calculated as g $CO_2$ - $kg^{-1}$ of fuel consumed, according to equations in Ward and Hardy (1991). North et al. (2009) also calculated the difference between pre- and post-burn fuel loads to estimate prescribed fire emissions. A range of prescribed fire treatments (burn only, understory thin and burn, overstory thin and burn) produced 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 post-treatment C stores (Table 3).

Differences in climate, fuels, topography, and management practices cause emissions from prescribed fire to vary differ (Liu, 2004), so substantial differences in prescribed fire emissions across studies and burn types is not unexpected. Emissions from broadcast burns (North et al., 2009; Stephens et al., 2009) are much higher than emissions generated from burning slash piles (Finkral and Evans, 2008). Slash burns more efficiently when piled, with less particulate matter produced per unit mass of fuel consumed than broadcast burns of the same material (Hardy, 1996).

### 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 the effects of stand development and management on fuel dynamics and potential fire behavior over time (Reinhardt et al., 2008). Simulated results of prescribed fire emissions using FFE–FVS are consistent with the range of results derived from empirical analyses, confirming the relative magnitude of prescribed fire contributions to treatment emissions (Hurteau and North, 2009; Ager et al., 2010; Reinhardt and Holsinger, 2010; Sorensen et al., 2011).

Hurteau and North (2009) modeled the effects of eight different fuel treatments on aboveground C storage for 100 years in the southern Sierra Nevada, using FFE–FVS to simulate prescribed fire at 20-year intervals (McKelvey and Busse, 1996; North et al., 2005). Burning conditions were calibrated at the study site to a 2001 prescribed fire considered representative of autumn burning conditions. Results across all treatments in the simulation period suggest that individual prescribed fires emit 4.1–16.3 Mg C $ha^{-1}$, less than 6% of the post-treatment C storage in all simulations. The sum of the five prescribed fires in the 100-year 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 in the northern Rocky Mountains. Stand-scale data from the Forest Inventory and Analysis (FIA) program (http://www.fia.fs.fed.us) were used as model inputs, and prescribed burning was simulated the year following thinning during "moderate" burning conditions. A burn-only treatment was also simulated. 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, or 13.2–36.0% of the total post-treatment C stores. Simulated thinning treatments remove 10.6 Mg C $ha^{-1}$, suggesting prescribed fire emissions exceed the amount of C removed during harvest (Table 3).

Sorensen et al. (2011) used FFE–FVS to simulate prescribed fire over 100 years at 10- and 20-year intervals in five ponderosa pine stands in northern Arizona, and burning conditions were calibrated to a prescribed fire in an adjacent stand (McHugh and Kolb, 2003). Each simulated prescribed fire emits 1–10 Mg C $ha^{-1}$, equivalent to 2–16% of the post-treatment C stores. In all treatments, the sum of prescribed fire applications over 100 years exceeds 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 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 underburning. Prescribed fire emissions are 19.5 Mg C $ha^{-1}$, or 36% of total emissions associated with the simulated treatment.

Overall, only live C removed by thinning exceeds the contribution of prescribed fire to total emissions of fuel treatments (Tables 1–3). Thinning-only treatments are ineffective at reducing potential wildfire behavior (Stephens et al., 2009; Prichard et al., 2010), and are unlikely to meet ecological objectives of restoration

**Table 4**
Summary of cumulative prescribed fire emissions.

| Study | Region | Treatment type | Cumulative Rx emissions (Mg C $ha^{-1}$) | Simulated wildfire emissions (Mg C $ha^{-1}$) |
|---|---|---|---|---|
| Sorensen et al. (2011)[a,b] | Northern Arizona | Restoration thin | 34.01/25.38 | 24.38/21.53 |
| Sorensen et al. (2011)[a,b] | Northern Arizona | Restoration thin | 37.88/27.35 | 22.19/18.73 |
| Sorensen et al. (2011)[a,b] | Northern Arizona | Restoration thin | 27.62/20.44 | 16.96/14.00 |
| Sorensen et al. (2011)[a,b] | Northern Arizona | Restoration thin | 40.04/26.72 | 20.35/18.57 |
| Sorensen et al. (2011)[a,b] | Northern Arizona | Restoration thin | 32.09/21.78 | 17.17/13.85 |
| 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 bur | 29.94 | 9.98 |

[a] Rxl0/Rx20 (prescribed fire simulated every 10/20 years).
[b] HF100/HF50 (no management with a high-intensity wildfire simulated within the next 100 years/50 years).

without removal of surface fuels (Weatherspoon, 1996; Weatherspoon and Skinner, 2002). As a result, a combination of thinning and prescribed fire is a preferred fuels management strategy in dry forests (Peterson et al., 2005), and the magnitude of prescribed fire emissions is therefore a critical consideration for C balance of fuel treatments.

## 5. Thinning and wildfire effects on C flux

Complexity in atmosphere–terrestrial ecosystem interactions complicates precise measurements of C flux in time and space (Hutley et al., 2005). Plot-scale biometric inventories indirectly estimate NPP at coarse temporal resolutions and rely on allometric relationships to scale up measurements (Baldocchi, 2003). Eddy covariance methods measure the exchange of $CO_2$ across the canopy–atmosphere interface at time scales 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 informs components and processes of net C fluxes, especially in the differentiation of contributions to C flux between individual species or respiration sources (Dore et al., 2010). A combination of biometric and eddy covariance measurements can refine process-based models used to assess larger scale effects on the C cycle (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 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 monitor responses in ecosystem physiology to environmental factors such as thinning and disturbance (Baldocchi, 2003; Law et al., 2003).

Empirical data and modeling indicate that stand-scale $CO_2$ exchange depends 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 disturbance causes respiration to exceed NPP (Dore et al., 2008). Recent studies have used eddy covariance techniques and biometric methods to explore the effects of 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, 2010; Campbell et al., 2009).

Law et al. (2004) used a spatially nested hierarchy of observations coupled with a process model (Biome-BGC) to determine the relative influence of climate and disturbance on C stocks and fluxes in western Oregon. Observations include eddy covariance measurements, inventory data, and remote sensing imagery. The authors developed a C budget by constructing a 5-year mean NEP for each 25-m grid cell. 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 is a net C sink (8.2 Tg C $yr^{-1}$). Harvest statistics were used to estimate C removed during the sample period (5.5 Tg C $yr^{-1}$). Harvest removals were disproportionately from the Coast Range ecoregion, which is managed in even-aged systems. Change detection analysis (1995–2000) was used to estimate average 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 although large wildfires affect regional net C balance, they do not exceed the contributions of commercial forestry.

Irvine et al. (2007) used 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, but found no significant change in heterotrophic respiration 2 years following wildfire regardless of increases in detrital pools in burned stands. This suggests that decay rates in detrital pools may determine C recovery in burned stands.

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 and leaf area index) conditions, as well as monthly and annual C budgets. The fire reduced annual ecosystem respiration by 30%, but significant reductions in gross primary production (GPP) (60%) resulted in a decrease in net C exchange. Ten years after the fire, the burned site was still a moderate source of $CO_2$ and may persist as a source for years due to slow vegetation recovery and tree establishment. The unburned site was a moderate C sink. Strong seasonal variability in C fluxes was captured in the C flux measurements, but the effects of high-intensity wildfire on NEE persisted.

Kaye et al. (2005) used biometric surveys, clip plots, allometric equations, and radial growth analyses to measure biogeochemical responses to restoration thinning treatments in ponderosa pine forest. 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 in comparison to C removed during thinning (17.7 Mg C $ha^{-1}$) and thin-and-burn treatments (33.7 Mg C $ha^{-1}$). The fate of thinned tree biomass and intensity of prescribed fires remain the most important factors controlling initial effects 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 in ponderosa pine forest. High-intensity wildfire has a larger influence on C balance than thinning: wildfire reduces total ecosystem C stocks by 40%, thinning reduces C stocks by 14%, and growth rate of residual unthinned trees increases (Kaye et al., 2005; McDowell et al., 2006). Both disturbances reduced GPP (55% by wildfire, 30% by thinning) more than ecosystem respiration (33–47% by wildfire, 18% by thinning), resulting in lower C uptake.

Several studies used biometric methods to assess the effects of fuel treatments and wildfire on C pools. Relatively few studies measured C flux, and even fewer directly measured total ecosystem C exchange via eddy covariance techniques. C flux studies capture fine-scale processes affecting ecosystem C exchange, such as 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 forests and has significant short-term effects on forest C balance (IPCC, 2007). However, long-term effects of disturbance on C dynamics are more uncertain and are influenced by 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 effects on C dynamics, but are limited by assumptions of constancy in biogeochemical cycles across multiple disturbances and successional pathways (Johnson and Miyanishi, 2008; Walker et al., 2010). Attempts to reduce uncertainty in ecosystem C exchange observations include techniques designed to combine plot-scale data and flux observations in process-based models (Kramer et al., 2002; Rayner et al., 2005; Williams et al., 2004). New methods for upscaling C and water fluxes to regional and continental scales include integration of eddy covariance measurements and remotely sensed data (Wylie et al., 2007; Holifield Collins et al., 2008; Xiao et al., 2008, 2010, 2011). However, uncertainty is still highest during extreme climatic events and large disturbances (Xiao et al., 2010, 2011).

## 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 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 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). However, 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 these studies and compare the relative effects of fuel treatments and wildfire on C dynamics.

Finkral and Evans (2008) use FFE–FVS to estimate the increase in crowning index (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). Thinned stands reduce the likelihood of active crown fire, even in extreme fire conditions, and releases less C (2.41 Mg C $ha^{-1}$) than unthinned stands. By examining three different wood utilization scenarios (firewood, paper, pallets/construction materials), it was concluded that utilization of long-lasting wood products may provide greater C benefit than the magnitude of reduced wildfire emissions.

Hurteau and North (2009) compare the response of six different fuel treatments to wildfire by simulating extreme wildfire conditions using FFE–FVS (17.9 m $s^{-1}$ wind, 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 are highest in the untreated stand (36.3 Mg C $ha^{-1}$), and fuel treatments that include prescribed fire result in lower emissions than thinning-only treatments (Table 4). In summary, fuel treatments release less C (12.7–26.3 Mg C $ha^{-1}$) than untreated stands.

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. More C is released at the time of wildfire from untreated stands than treated stands in all habitat types. However, to sharpen comparisons across studies, we focus on the authors' discussion of warm, dry ponderosa pine habitat (Fischer and Bradley, 1987). Wildfire was simulated 5 years after the implementation of alternative fuel treatments (thin and burn, burn only). Wildfire conditions were designed to represent dry, late summer wildfires (8.9 m $s^{-1}$ wind, 21 °C temperature, "low" moisture content of surface fuels). Mean reduction in wildfire emissions was 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

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

| Study | Region | Treatment type | Wildfire emissions in untreated stands | Wildfire emissions in treated stands | 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 | 36.3 | 24.49 | 11.8 |
| Hurteau and North (2009) | Sierra Nevada, California | Restoration thin | 36.3 | 23.59 | 12.7 |
| Hurteau and North (2009) | Sierra Nevada, California | 1865 Reconstruction thin | 36.3 | 20.87 | 15.4 |
| Hurteau and North (2009) | Sierra Nevada, California | Burn only | 36.3 | 16.33 | 20.0 |
| Hurteau and North (2009) | Sierra Nevada, California | Understory thin and burn | 36.3 | 18.14 | 18.2 |
| Hurteau and North (2009) | Sierra Nevada, California | Restoration burn | 36.3 | 15.42 | 20.9 |
| Hurteau and North (2009) | Sierra Nevada, California | 1865 Reconstruction thin bur | 36.3 | 12.70 | 23.6 |
| Reinhardt and Holsinger (2010) | Northern Rockies | Thin from below and burn | 12.64 | 7.57 | 5.1 |
| Reinhardt and Holsinger (2010) | Northern Rockies | Burn only | 12.64 | 9.71 | 2.9[a] |
| 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 bur | 67.8[b] | 29.71 | 40.1 |

[a] Significant difference not detected (Reinhardt and Holsinger, 2010).
[b] Value represents the mean of 19 paired treated/untreated sites.

fire severity, crown fire occurrence, and tree mortality from wildfire while reducing subsequent wildfire emissions. However, the treatments themselves produce emissions that exceed simulated reductions in wildfire emissions.

Stephens et al. (2009) used Fuels Management Analyst Plus (FMA) to estimate tree mortality from simulated wildfire under extreme burning conditions. In the study area, approximately 70% of total aboveground C was contained in live trees. In untreated stands, approximately 90% of live-tree C was determined to be at high risk (>75% mortality) during severe wildfire. Untreated stands contain approximately 145 Mg C $ha^{-1}$ of live-tree C at high risk for wildfire mortality. Thin-only treatments have 18 Mg C $ha^{-1}$ of live-tree C at high risk for mortality, whereas thin-and-burn treatments reduce the value to 4 Mg C $ha^{-1}$. Treatments that combine moderate to heavy thinning from below with prescribed fire are most effective at preventing the transfer of C from live to dead pools during wildfire. Although tree mortality is an important metric for assessing stand resilience to disturbance, most C stored in forest biomass (stem wood, branches, and coarse woody debris) usually remains unconsumed even by high-severity wildfires. Combustion of surface and ground fuels is usually the largest proportion of emissions from wildfires (Campbell et al., 2007; Meigs et al., 2009), so tree mortality may be an incomplete metric for estimating C emissions from wildfire.

Meigs et al. (2009) report 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, and used clip plots to assess understory vegetation. Estimates of wildfire consumption were calculated with Consume 3.0 (http://www.fs.fed.us/pnw/fera/research/smoke/consume/index.shtml) augmented with local data. Ponderosa pine stands in the burned area emitted an average of 19.7–30.2 Mg C $ha^{-1}$. Grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.) – 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 grand fir–Douglas-fir. Fuel treatments were not present in the study area, but the study provides context, via empirical evidence across a range of burn severity, for pyrogenic emissions in ponderosa pine. Estimates are consistent with 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 the forest ecosystem model, STANDCARB, to examine the effects of fuel reduction on fire severity and long-term C dynamics in east Cascade Range ponderosa pine, west Cascade Range western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), and Coast Range western hemlock-Sitka spruce (*Picea sitchensis* (Bong.) Carrière) forest (Oregon) STANDCARB integrates climate-driven growth and decomposition processes with species-specific rates of senescence, mortality, and competition that characterize forest gap dynamics. Results indicate that fuel treatments in ponderosa pine forest reduce fire severity. However, the quantity of C removed during treatment exceeded the avoided C losses from mitigated fire severity because much of the C stored in forest biomass remains unconsumed in wildfires. The direction of the simulated net C response (+, −) varied according to pre-treatment conditions, and fuel treatments produced C gains in only the driest ecosystems in the east Cascades with high levels of fuel accumulation.

Sorensen et al. (2011) quantified the effects of fuel treatments and wildfire on the C budget of five ponderosa pine stands in northern Arizona, using FFE–FVS to simulate long-term effects of wildfire, thinning, and repeated prescribed fire. 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 $s^{-1}$ wind, 29 °C temperature, "very dry" surface fuels). Wildfire emissions range from 17.0 to 24.4 Mg C $ha^{-1}$ in the HF100 scenario and 13.9–21.5 Mg C $ha^{-1}$ in the HF50 scenario. Simulations for scenarios of prescribed fire applied every 10 and 20 years over 100 years calculated prescribed fire emissions 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 (torching index and crowning index), but did not estimate emissions from wildfire. The authors simulated 95th percentile burning conditions for the study area using Fire Family Plus (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 five 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 landscape risk analysis. A probabilistic framework for wildfire occurrence and a fire spread algorithm from FlamMap (Finney, 2002) were used to simulate 30,000 burn periods at random ignition locations in a watershed in southern Oregon dominated by ponderosa pine. Fire growth calculations were generated at 90 × 90 m grids, and parameters (temperature and fuel moisture) for wildfire simulations were derived from 97th percentile August fire weather conditions. Wind speed was 11.1 m $s^{-1}$ in all simulations and wind direction was randomly simulated across three azimuths representing dominant wind patterns. Results indicate a decrease in average burn probability (treated 0.017 vs. untreated 0.021) across all forested areas. Wildfire emissions were generated from FFE–FVS in all scenarios. Stands were selected for simulated fuel treatment based on criteria from the Fremont-Winema National Forest and

**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- | Low | 16.6 | 13 |
| Meigs et al. (2009) | Eastern Oregon (Grand fir/Douglas- | Moderate | 25.3 | 19 |
| Meigs et al. (2009) | Eastern Oregon (Grand fir/Douglas- | High | 32.3 | 24 |
| Meigs et al. (2009) | Eastern Oregon (Pacific ponderosa \| | Low | 19.7 | 23 |
| Meigs et al. (2009) | Eastern Oregon (Pacific ponderosa \| | Moderate | 25.6 | 29 |
| Meigs et al. (2009) | Eastern Oregon (Pacific ponderosa \| | High | 30.2 | 35 |

comprise 7180 ha of 45,192 forested acres in the study watershed (~15%). In the areas designated for fuel treatment (7180 ha), emissions from wildfire are 0.02 Mg C $ha^{-1}$ (treatment) and 1.38 Mg C $ha^{-1}$ (non-treatment). 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%) constitute a much higher proportion of treatment-related emissions.

Pearson et al. (2010) and Goslee et al. (2010) developed methodologies to evaluate C dynamics associated with fuel treatment projects in low to mid-elevation forest in northern California and Oregon. The authors, with consultation from teams of scientists, quantify C storage and release within the context of a six-point conceptual framework: annual fire risk, treatment emissions, fire emissions, forest growth and re-growth, re-treatment, and the shadow effect (i.e., treatment effect outside the treated area). Results indicate that the mean annual probability of wildfire for the study region is less than 0.76%/year, and treatments reduce C stocks by an average of 19%. Where timber is removed, 30% of extracted biomass is stored in long-lasting wood products. Wildfire emissions in treated stands, quantified with the Fuel Characteristic Classification System, are reduced by 6% relative to untreated stands. Growth estimates for a 60-year simulation horizon, derived from FVS, indicate that in the absence of wildfire, untreated stands sequester 17% more C than treated stands. However, in simulations that include wildfire, treated stands sequester 63% more C than untreated stands. The shadow effect is unlikely to be large enough to affect net GHG emissions. In summary, initial reductions in C stocks (e.g., thinning), combined with low annual probability of wildfire, preclude C benefits associated with fuel treatments, even if harvest residues are used for biomass energy.

Campbell et al. (2007) determined combustion factors in mixed conifer forest in the Biscuit Fire (Oregon) in 25 different C pools, assessing variation as a function of remotely sensed burn severity data. To estimate pyrogenic emissions (stratified by burn severity) the authors combined combustion factors with pre-burn fuel densities derived from FIA plots ($n$ = 180) and species-specific allometric equations. The combustion factor increases monotonically in nearly all C pools as burn severity increases. Total emissions for the Biscuit Fire are 3.5–4.4 Tg C (12.4–28.6 Mg C $ha^{-1}$, depending on severity), with 60% of emissions from 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 short residence time (10–20 years).

North and Hurteau (2011) quantified wildfire emissions and short-term changes in C stocks in treated and untreated stands burned in 12 wildfires in the central and southern Sierra Nevada (19 associated fuel treatments). 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). Average emissions from wildfire in treated stands (29.7 Mg C $ha^{-1}$) are 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, mean net C loss is higher in treated (80.2 Mg C $ha^{-1}$) than untreated (67.8 Mg C $ha^{-1}$) stands. Tree mortality differs in treated (53%) and untreated (97%) stands, as does survivorship of trees >50 cm d.b.h. (87% and 6%, respectively). Overall, 70% of total ecosystem C in untreated stands is transferred to decomposing C with significant effects on long-term C balance.

Campbell et al. (2011) used empirical data from 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 the effects of fire-return intervals (10, 50, 250 years) for ponderosa pine forest in eastern Oregon. Results indicate that total C is less variable in a high-frequency, low-severity fire regime, but that long-term, total forest C is higher in a low-frequency, high-severity fire regime (simulation period = 500 years). The authors report a modest increase in long-term average fractional combustion of ecosystem C (9%) between high- and low-frequency fire regimes. 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% (violate the conservation of mass) in order to equal combustion achieved in the high-frequency disturbance regime. Therefore, it is concluded that fuel reduction treatments result in lower C storage in all disturbance regimes.

Stephens et al. (2012) reported C dynamics from six sites in four western US states in the Fire and Fire Surrogate study, a network of long-term studies designed to quantify the effects of fire and fire surrogate treatments in multiple forest types and conditions. The study design includes thin-only, thin-and-burn, burn-only, and control treatments at each site. The authors used the First Order Fire Effects Model (Reinhardt, 2003) to simulate fuel consumption and emissions from wildfire. Canopy fraction burned was modeled under the upper 97.5 percentile fire weather conditions using FMA. Wildfire emissions range from 10 to 80 Mg C $ha^{-1}$ and are lowest in treatments that include prescribed fire. C removed during treatment ranges from 30 to 60 Mg C $ha^{-1}$. Variability in C dynamics across sites suggests that trade-offs between treatment removals and wildfire emissions are highly site-specific.

## 7. Conclusions

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 synthesis has focused on the magnitude of initial C losses to fuel treatments via tree removal (thinning), equipment usage, milling waste, and the consumption of surface fuels during prescribed fire. We also discussed various methods for measuring wildfire emissions and fuel treatment C costs, including biometric measurements, eddy covariance techniques, and model simulations.

All studies agree unequivocally that untreated stands release more emissions to the atmosphere during wildfire than treated stands, and that emissions increase monotonically as burn severity increases. Tree mortality from wildfire is also consistently reduced by the presence of fuel treatments. 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 in 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, 2011) and thus must be included in analyses of long-term C storage. Although 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 assume future wildfire probability of 100%, reporting inferences that essentially detail a "best-case scenario" for wildfire

emissions mitigation. Annual probability of wildfire in dry temperate forests for a given stand is approximately 1% (Ager et al., 2010; Pearson et al., 2010; Campbell et al., 2011).

High spatial and temporal variability in the burn environment and complexity in atmosphere–terrestrial ecosystem interactions create uncertainty in estimating 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 C in litter, organic soil, and mineral soil, which together comprise approximately 60% of all wildfire emissions (Campbell et al., 2007). It is also challenging to estimate surface fuels and wildfire emissions from remotely-sensed data (Lachowski et al., 1995; Keane et al., 2001; Wiedinmyer and Neff, 2007). Simulation approaches are commonly used to estimate wildfire emissions, but model inputs are constrained by a paucity of empirical data. Ultimately, improved estimates of surface fuel consumption at large spatial scales will improve management of dry temperate forests to sequester C.

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 through timber harvest, prescribed fire, and milling waste exceed observed and simulated reductions in wildfire emissions. However, if thinned trees are milled into long-lasting wood products, 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). Utilization of residual woody biomass 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; Winford and Gaither, 2012). However, many areas in the western US do not have biomass markets to compensate for the cost of treatments (Reinhardt and Holsinger, 2010), and small diameter trees removed during treatments are often unmerchantable and scattered onsite or burned in piles, increasing overall treatment emissions (Han et al., 2004). Although long-lasting wood products and energy utilization can help offset a portion of harvested C, a substantial contribution to total treatment costs is derived from prescribed fire, which reduces fine surface fuels, potential fire behavior, and crown fire hazard (Graham et al., 2004; Peterson et al., 2005; Raymond and Peterson, 2005).

The ability of fuel treatments to mitigate future fire behavior and move forest structure to a more fire-resistant condition is well documented. However, C costs associated with fuel treatments have can exceed the magnitude of C reduction in wildfire emissions, because 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 larger transfers occur between live and dead pools, where C is released gradually via decomposition over decades.

Fuel treatments may effectively reduce disturbance severity with 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. Burn probabilities in treated stands in southern Oregon are less than 2%, so the probability that a treated stand encounters wildfire and creates C benefits is low (Ager et al., 2010). Strategically located fuel treatments designed to reduce wildfire spread per treatment area may reduce the number of required fuel treatments in a given landscape (Finney et al., 2007), but even optimal fuel treatment placements require 10 ha of treatment to protect 1 ha of forest from wildfire (Finney et al., 2007; Campbell et al., 2011).

The temporal horizon at which trade-offs are examined determines the degree to which fuel treatments affect C dynamics. 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 (emissions, harvest) reflects disturbance intensity and resilience of the disturbed forest and may be shortened by treatments that 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. Assuming that additional disturbance does not occur prior to forest recovery to initial stand age and density, 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., forest to shrubs or grass) following unsuccessful post-fire tree regeneration or resulting from disturbance to the soil may affect C sequestration in a warmer climate. For example, C losses from high-severity wildfire can persist for decades as a result of protracted GPP recovery and vegetation life-form conversion (Savage and Mast, 2005; Deal et al., 2010; Dore et al., 2010). Fuel treatments can benefit long-term C storage by reducing the likelihood of such events.

Overall, inferences about fuel treatment and wildfire effects on C storage are varied and equivocal. Empirical and modeled stand-scale studies report a wide range of treatment effects on C storage, complicating inferences about C benefits. Studies at large spatial scales show that the rarity of high-severity wildfire events in a treated landscape can result in reduced C storage. However, few studies satisfactorily address C response to future disturbance scenarios. Considerable uncertainty exists in modeling C exchange during extreme droughts and large disturbances, events that are expected to increase in frequency in the future. In order to improve the accuracy and usefulness of assessments of fuel treatment wildfire trade-offs for C storage, it will be critical to quantify the entire fuel profile and consider interactions of multiple 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 trade-off analyses.

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