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The carbon costs of mitigating high-severity wildfire in southwestern ponderosa pine

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Abstract

Forests provide climate change mitigation benefit by sequestering carbon during growth. This benefit can be reversed by both human and natural disturbances. While some disturbances such as hurricanes are beyond the control of humans, extensive research in dry, temperate forests indicates that wildfire severity can be altered as a function of forest fuels and stand structural manipulations. The purpose of this study was to determine if current aboveground forest carbon stocks in fire-excluded southwestern ponderosa pine forest are higher than prefire exclusion carbon stocks reconstructed from 1876, quantify the carbon costs of thinning treatments to reduce high-severity wildfire risk, and compare posttreatment (thinning and burning) carbon stocks with reconstructed 1876 carbon stocks. Our findings indicate that prefire exclusion forest carbon stocks ranged from 27.9 to 36.6 $\mathrm{Mg\,C\,ha^{-1}}$ and that the current fire-excluded forest structure contained on average 2.3 times as much live tree carbon. Posttreatment carbon stocks ranged from 37.9 to 50.6 $\mathrm{Mg\,C\,ha^{-1}}$ as a function of thinning intensity. Previous work found that these thinning and burning treatments substantially increased the 6.1 m wind speed necessary for fire to move from the forest floor to the canopy (torching index) and the wind speed necessary for sustained crown fire (crowning index), thereby reducing potential fire severity. Given the projected drying and increase in fire prevalence in this region as a function of changing climatic conditions, the higher carbon stock in the fire-excluded forest is unlikely to be sustainable. Treatments to reduce high-severity wildfire risk require trade-offs between carbon stock size and carbon stock stability.

Keywords: climate change, forest carbon, mitigation, ponderosa pine, wildfire

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Introduction

Carbon sequestration is one of a suite of ecosystems services that forests provide. Climate change mitigation using forests can be accomplished by reducing emissions from deforestation and degradation, increasing forested land area via afforestation or reforestation, increasing carbon density through improved forest management, and sustainably using forest biomass to replace fossil fuel-based energy sources (Canadell & Raupach, 2008). Estimates of the US forest carbon sink indicate that forest carbon stocks have been increasing in part as a result of in-growth due to fire exclusion and regrowth due to land abandonment (Hurtt *et al*., 2002) and now sequester approximately 10% of annual anthropogenic emissions (Woodbury *et al*., 2007).

However, storing carbon in forests is not without risk. Disturbances that kill trees represent a risk to forest carbon sequestration because the carbon stored in trees can be released back to the atmosphere (Galik & Jackson, 2009; Hurteau *et al*., 2009). While regulations can limit the risk of human-caused reversals, natural disturbances continue to pose a risk to forest carbon projects even in the well-regulated US forest sector. Some of these disturbances, such as storms and insect outbreaks, are difficult if not impossible to manage for. Chambers *et al*. (2007) estimated that Hurricane Katrina resulted in a loss of 105 Tg of forest carbon. Kurz *et al*. (2008) estimate that the mountain pine beetle outbreak in British Columbia will result in a 270 Tg loss of forest carbon over a 20-year period. However, some reversal risks, such as wildfire, can be managed for. Fires emitted approximately 4–6% of US carbon emissions from 2001 to 2007 (Wiedinmyer & Neff, 2007), a relatively small fraction compared with anthropogenic fossil-fuel derived carbon emissions, but still a substantial amount of C.

Wildfires release carbon to the atmosphere as a function of direct biomass combustion and indirectly through the decomposition of fire-killed trees. Indirect emissions from wildfire have been estimated at as much as three times the amount of carbon lost in direct emissions (Auclair & Carter, 1993). Measurements of postwildfire net ecosystem productivity indicate that

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following wildfire, forests can continue to be a source of carbon to the atmosphere for a number of years (Dore *et al*., 2008; Meigs *et al*., 2009).

In the western United States, forest fires have been increasing in size and severity as a result of past management actions to exclude fire and on-going climate change (Covington *et al*., 1997; Fulé *et al*., 1997; Westerling *et al*., 2006; Miller *et al*., 2009). Area burned by wildfire in the western United States correlates with climate variability and in forested systems is mechanistically explained by low fuel moisture and fuel quantity and continuity (Keane *et al*., 2008; Littell *et al*., 2009). Future climate projections for southwestern North America suggest that increasing temperature will likely result in regional drying (Seager *et al*., 2007). Regardless of the atmospheric carbon dioxide emission scenario, the occurrence of large fires is projected to increase (Westerling & Bryant, 2008). Thus, taking action to mitigate high-severity wildfire risk is not solely in the domain of forest conservation, but also pertinent to climate change adaptation.

Altering forest structure and reducing surface fuels have proven effective at reducing fire severity (Agee & Skinner, 2005; Stephens & Moghaddas, 2005; Roccaforte *et al*., 2008; Collins *et al*., 2009). In addition to reducing fire severity, these treatments can also reduce wildfire related carbon emissions (Hurteau *et al*., 2008; Hurteau & North, 2009; Mitchell *et al*., 2009; Stephens *et al*., 2009a; Wiedinmyer & Hurteau, 2010). However, the reduction in risk comes at a carbon cost because of the associated reduction in tree biomass, waste as a function of mill efficiency, and fossil fuel emissions from harvesting (Finkral & Evans, 2008; Mitchell *et al*., 2009; North *et al*., 2009). The prefire exclusion structure of these forests, which was maintained by frequent, low-severity fire, is often the reference or target condition used as a marker against which treatments are measured. The purpose of this research was threefold (1) to quantify the live tree carbon in the reconstructed forest in 1876, before the exclusion of frequent surface fires; (2) to quantify aboveground forest carbon both before and after wildfire risk mitigation treatments; (3) to compare pre- and posttreatment live tree carbon stocks with the reconstructed 1876 forest carbon stock for four levels of thinning in a ponderosa pine (*Pinus ponderosa*) forest in northern Arizona. Specifically, we sought to determine if postfire exclusion, pretreatment live tree carbon stocks were larger or smaller than prefire exclusion carbon stocks.

Materials and methods

Study area

This study was conducted within and adjacent to the Fort Valley Experimental Forest, a 2003 ha mixed-age ponderosa pine reserve located approximately 15 km northwest of Flagstaff, Arizona in the Coconino National Forest (35°16′19″N, 111°41′22″W). The study area elevation is approximately 2250 m with average slopes of 5–10%. Mean annual precipitation is 57 cm of which roughly half falls as rain in July and August and half as snow during winter months (National Oceanic and Atmospheric Administration, 2005). The forest overstory is dominated by mature ponderosa pine, intermixed with dense thickets of smaller ponderosa pine and a primarily herbaceous understory. Before fire exclusion, this area was characterized by an open forest canopy, dominated by large trees, and maintained by frequent surface fires (Covington *et al*., 1997).

Treatments and data collection

In 1998, three experimental blocks were established, consisting of four 14 ha treatment units. Each treatment unit was randomly assigned one of four thinning treatments. Tree thinning treatments were based on a site-specific reconstruction of prefire exclusion forest structure using principles described by Moore *et al*. (1999) and included a no-thin control, and three levels of thinning that used evidence of old-growth individuals as a basis for retention. In addition to retaining all trees that had established before 1876, the thinning treatments had variable levels of retention per evidence (e.g. snags, logs, stumps) of an old-growth individual (Covington & Moore, 1994; Fulé *et al*., 2001). The three levels of retention are based on the number of trees retained for each piece of evidence (e.g. snag, log, etc.) of an old-growth individual, with the retained trees essentially serving as replacements for the old-growth individual. The heaviest thinning treatment retained 1.5–3 trees, the moderate thinning treatment retained 2–4 trees, and the lightest thinning treatment retained 3–6 trees per evidence of each old-growth individual. The number of young trees retained depended on tree size; when the young trees were small [$<$40.6 cm diameter at breast (dbh)], the higher retention rate in each range was used. Tree retention was also a function of proximity to the evidence of an old-growth individual, whereby trees that were closest to each piece of evidence were retained to approximate the prefire exclusion spatial pattern. Thinning treatments were implemented in 1999 and prescribe burned in either spring 2000 or spring 2001.

Twenty 400 m^2 (11.28 m radius) fixed area monitoring plots were established on a 60 m $\times$ 60m grid in each of the 12 treatment units (240 total plots). Plot centers were permanently marked with iron stakes and all trees were tagged to ensure exact relocation for sampling in subsequent years. Sampling took place in 1998 (pretreatment) and 2006 (posttreatment). Overstory trees taller than breast height (137 cm) were measured on each plot, including species, condition (living or snag/log condition class), dbh and a preliminary field classification of pre- or postfire exclusion (1876) origin. Trees likely to have originated before 1876 were identified based on size ($>$40 cm diameter at stump height) or yellow bark (White, 1985). All potentially prefire exclusion trees, as well as a 10% random subsample of other live trees, were cored with an increment borer at 40 cm in height. Dead woody biomass and forest floor fuel loads were measured on a 15.2 m planar transect in a random direction from each plot center. Woody debris biomass was calculated using procedures in Brown (1974) and Sackett (1980). Forest floor fuel loading (e.g. surface organic matter comprised of litter and duff) was calculated using equations from Ffolliot *et al.* (1968). Understory vegetation was sampled in each plot on a 50 m point-line intercept transect oriented along the plot aspect and centered on plot center. Herbaceous plants were recorded every 30 cm along each transect for a total of 166 points per plot. Plant foliar cover (%) was estimated by dividing the number of plant occurrences along the point-intercept transect by 166 points.

Collecting vegetation data using quadrats has been shown to provide a more accurate estimate of understory cover compared with the point-line intercept method in southwestern ponderosa pine forest (Korb *et al.*, 2003; Abella & Covington, 2004). Conversions of ocular cover estimates to plant biomass for the general location have been developed using 1 m^2 quadrats (Laughlin, 2009). Therefore, in 2006, we additionally sampled ten 1 m^2 quadrats per plot arranged along the center of each point-line intercept transect. Understory cover data for 1998 were estimated using a linear regression equation between 2006 quadrat cover and point-line cover data (quadcover $= 8.9227 + 3.7040 \times$ point-line cover, $P < 0.0001$, $r^2 = 0.80$). Plant biomass was then calculated using equations from Laughlin (2009).

Forest reconstruction and carbon concentration

Increment cores were surfaced and cross-dated with local chronologies. Rings were counted on cores that could not be cross-dated. For cores with missing pith, additional years to the center were estimated with a pith locator consisting of concentric circles matched to the curvature and density of the inner rings. Prefire exclusion forest structure was reconstructed at the time of disruption of the frequent fire regime, *circa*, 1876, following dendroecological methods described in detail in Fulé *et al.* (1997). Tree diameters for 1876 were reconstructed for both living and dead trees as a function of inventoried diameter and/or radial increment using methods described in Bakker *et al.* (2008). Dendroecological reconstructions in this forest type and locality have been shown to be accurate within ± 3 trees ha^{-1} as compared with historical forest measurement plot data (Moore *et al.*, 2004).

We used allometric equations developed locally by Kaye *et al.* (2005) to calculate total above ground carbon biomass for pretreatment (1998), posttreatment (2006) and reconstructed (1876) live trees and snags. To quantify carbon in coarse (>7.62 cm in diameter) and fine woody debris (<7.62 cm in diameter), we used a biomass-to-carbon conversion factor of 50% (Penman *et al.*, 2003). Forest floor fuel loads (e.g. litter and duff) were converted to carbon biomass assuming a carbon concentration of 37% (Smith & Heath, 2002). We assumed the carbon concentration in understory plant biomass to be 43% (Laughlin, 2009).

Results

Reconstructed 1876 aboveground live tree carbon (C) stocks ranged from 27.9 to 36.6 Mg C ha^{-1}. On average, pretreatment aboveground live tree C was 2.3 times greater than the reconstructed 1876 live tree carbon and ranged from 69.5 to 75.1 Mg C ha^{-1} (Fig. 1a). Posttreatment mean aboveground live tree C values ranged from 37.9 Mg C ha^{-1} in the 1.5–3 tree retention prescription, to 50.6 Mg C ha^{-1} in the 3–6 tree retention prescription (Fig. 1b). Paired *t*-test comparisons of the posttreatment and reconstructed 1876 live tree C indicated that only the 3–6 leave tree prescription was significantly greater than the reconstructed live tree C stock. The difference between pre- and posttreatment control C stock values is due to live tree growth and a reduction in snags over the 8 years between measurement periods.

Tree frequency by diameter class tended to be more evenly distributed in the 1876 reconstruction (Fig. 2). In pretreatment stands, smaller diameter classes were disproportionally represented. Posttreatment diameter distributions approximated the 1876 reconstruction, because smaller trees were preferentially harvested (Fig. 2b–d). However, all thinning levels had a greater number of individuals in each diameter class as compared with the reconstruction because the treatments were designed to leave greater than one tree for evidence of each tree that was present in 1876.

Before treatment, litter and duff, fine woody debris, coarse woody debris, and understory vegetation comprised between 21% and 28% of the aboveground carbon stock (Fig. 1a). Posttreatment, the contributions of these pools to the total carbon stock was a function of thinning intensity, ranging from 23% in the 1.5–3 leave tree prescription to 17% in the 3–6 leave tree prescription (Fig. 1b).

Discussion

The carbon carrying capacity of a forest represents the amount of C that can be maintained in the system given climatic conditions and natural disturbance regimes, and barring human disturbance (Keith *et al.*, 2009, 2010). Fire is a natural disturbance in the ponderosa pine forests of the southwestern United States. However, the frequency and intensity of fire in these systems has been fundamentally altered by human intervention, causing a transition from frequent, low-severity fire to infrequent, high-severity fire (Covington *et al.*, 1997; Fulé *et al.*, 1997). As a result, live tree C stocks at the Fort Valley study site have increased on average by 231% since fire exclusion in 1876. The results of this study contrast with research in the Sierra Nevada mountains of California that indicates while tree density has increased as a result of fire exclusion, live tree carbon stocks have decreased because of a reduction in the number of large trees which contain a

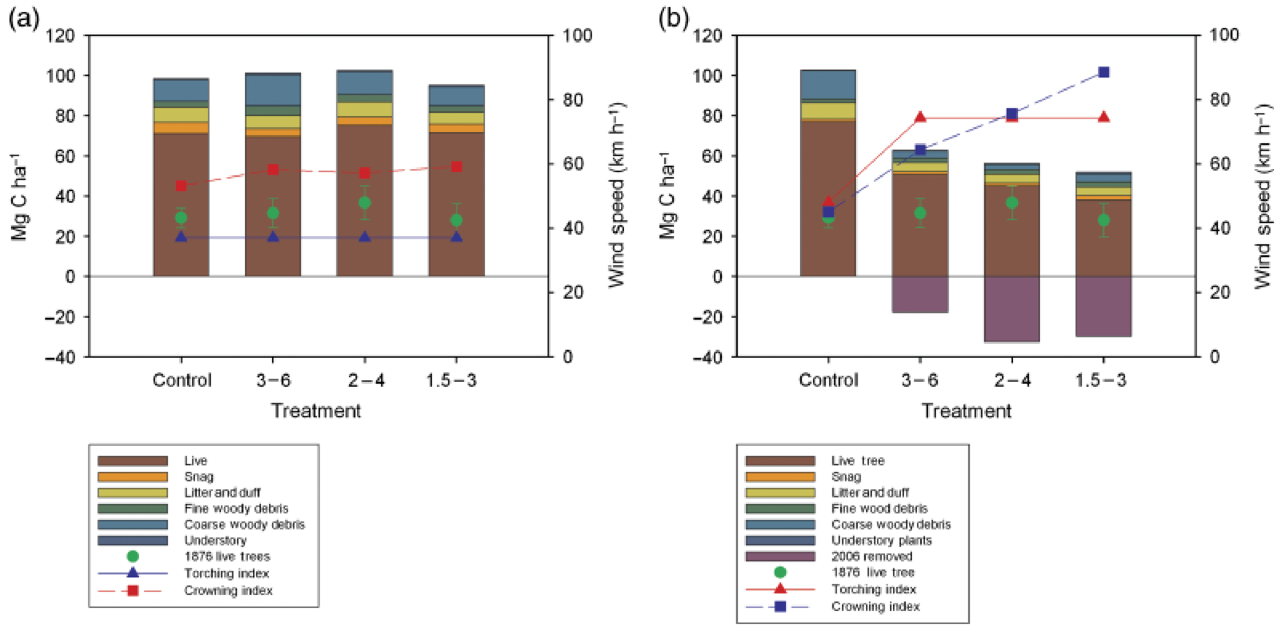

Fig. 1. The stacked bars are the pretreatment (a) and posttreatment (b) carbon stocks in live trees, litter and duff, fine woody debris, coarse woody debris, and understory plants for the control 1.5–3, 2–4, and 3–6 leave tree prescriptions. Green dots are the mean reconstructed 1876 live tree carbon stocks with standard errors. The pre- and posttreatment torching indices and crowning indices from Fulé *et al.* (2001) are represented by blue triangles and red squares, respectively.

disproportionate amount of the live tree C (Fellows & Goulden, 2008; North *et al.*, 2009).

Torching and crowning indices are commonly used metrics for quantifying the 6.1 m wind speed required for fire to move from the surface into the forest canopy (torching) and for the occurrence of an active crown fire (crowning) (Scott & Reinhardt, 2001). Previous fire modeling work conducted by Fulé *et al.* (2001) at this site indicated that increased tree density due to fire exclusion had resulted in a decrease in torching and crowning indices, suggesting that this system has exceeded the carbon carrying capacity (Fig. 1). Fulé *et al.* (2001) found that these treatments were effective at increasing both torching and crowning indices over pretreatment values, with the greatest posttreatment increases in the 1.5–3 tree retention prescription (Fig. 1b). Achieving these increases in torching and crowning indices required a reduction in the live tree C pool. Mean live tree C reductions ranged from 17.7 to 32.6 Mg C ha^{-1} and were equivalent to 41.5%, 43.4%, and 25.6% of pretreatment live tree C in the 1.5–3, 2–4, and 3–6 tree retention prescriptions, respectively.

The concept of surpassing the carbon carrying capacity in this system is further supported by two other recent studies in the region. A survey of 10 ponderosa pine sites that burned under stand-replacing fire conditions across Arizona and New Mexico found that only 50% of the sites had sufficient tree regeneration to lead to forest recovery (Savage & Mast, 2005). The remaining sites showed limited or no regeneration indicating a shift toward a nonforest type. Dore *et al.* (2008) report that 10 years following a stand-replacing wildfire in northern Arizona, the site continues to be a source of C to the atmosphere, with little evidence of transition to a carbon sink in the near future.

Before recent human intervention, high-severity fires in southwestern ponderosa pine forests were rare (Swetnam *et al.*, 1999). Given the recent evidence of increasing fire size in the western United States, coupled with projections of further regional drying due to changing climatic conditions (Westerling *et al.*, 2006; Seager *et al.*, 2007), large fires are likely to become more prevalent on the landscape (Westerling & Bryant, 2008). As a result, we need to view the climate change mitigation potential of these forests in the context of the potential for C loss due to wildfire. The results of this research and work by Fulé *et al.* (2001) indicate that high-severity wildfire resistance can be restored to southwestern ponderosa pine forests. However, this increased resistance comes with a C stock reduction cost.

Are the forest management goals of high-severity wildfire risk reduction, carbon sequestration to mitigate climate change, and capacity building for system level climate change adaptation mutually exclusive? Forest structure can be manipulated to reduce the risk of stand-replacing wildfire, but at the cost of a reduction in C stocks (Finkral & Evans, 2008; Hurteau & North, 2009; Mitchell *et al.*, 2009; North *et al.*, 2009; Stephens

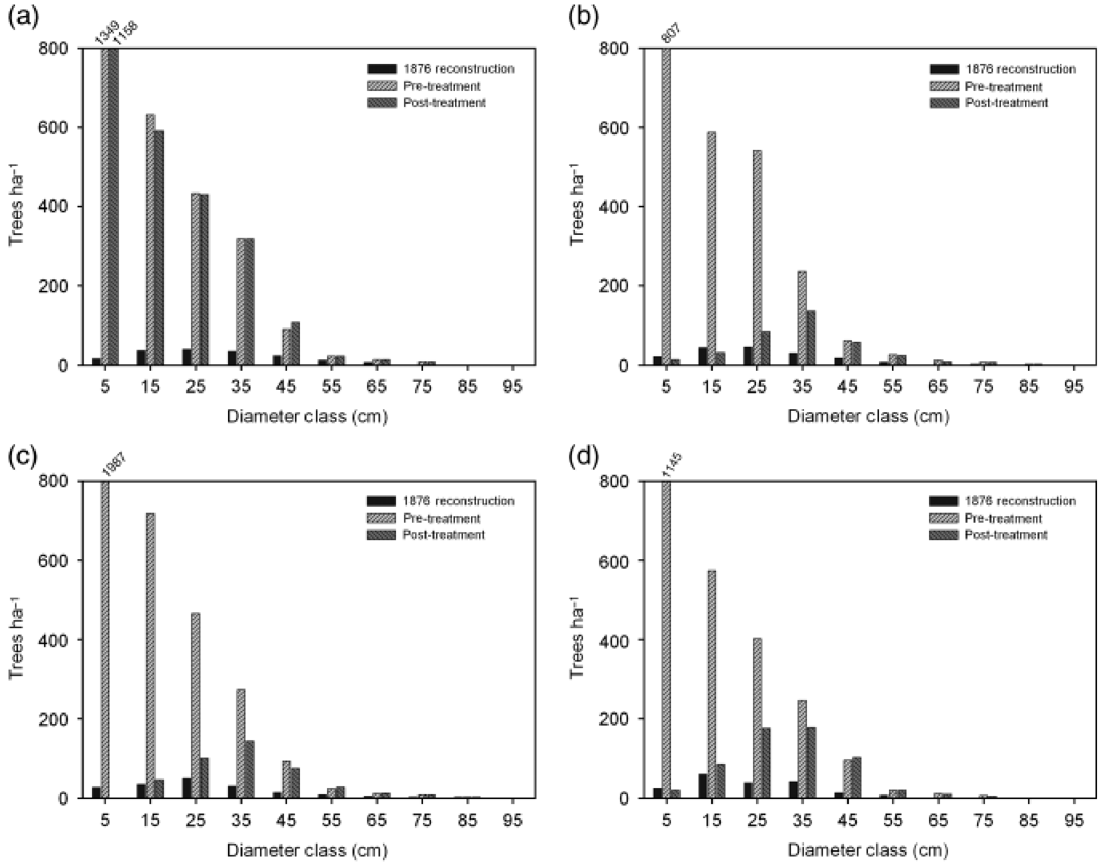

Fig. 2. The 1876 reconstruction, pretreatment, and posttreatment tree frequency by 10 cm diameter classes are shown for the control (a), 1.5–3 (b), 2–4 (c), and 3–6 (d) leave tree prescriptions.

et al., 2009a, b; Reinhardt & Holsinger, 2010). However in dry forest types, such as ponderosa pine, these carbon stock reductions can have a net carbon benefit if wildfire emissions reductions are larger than the carbon removed during treatment (Hurteau & North, 2009; Mitchell *et al.*, 2009). Furthermore, the trees retained during treatment continue to grow and sequester carbon (Hurteau & North, 2010). Climate change adaptation adds another level of uncertainty to management. Recent increases in western US wide tree mortality have been attributed to increasing temperature and the associated water stress (van Mantgem *et al.*, 2009). Work in northern Arizona indicates that forest thinning increased the carbon sink strength compared with unthinned forest during the driest summer months (Dore *et al.*, 2010), suggesting that this management option adds system level resilience to drought.

Given the potential for high-severity fire in southwestern ponderosa pine forests to yield a vegetation type conversion and an associated reduction in the carbon stock and carbon stock potential, wildfire risk mitigation treatments could be viewed in the context of reducing emissions from deforestation and degradation, whereby high-severity fire is the agent of change. While the carbon stock per unit area arising from fire exclusion may be higher in postfire exclusion forest, it is likely to be unsustainable in the face of changing climatic conditions and fire, and therefore counterproductive for climate change mitigation. In contrast, managing the system to be within the natural range of variability for carbon stock size offers the opportunity to restore fire as a natural process, improve carbon stock stability, and potentially to build climate change adaptation capacity.

References

Abella SR, Covington WW (2004) Monitoring an Arizona ponderosa pine restoration: sampling efficiency and multivariate analysis of understory vegetation. *Restoration Ecology*, **12**, 359–367.

Agee JK, Skinner CN (2005) Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*, **211**, 83–96.

Auclair AND, Carter TB (1993) Forest wildfires as a recent source of CO_2 at northern latitudes. *Canadian Journal of Forest Research*, **23**, 1528–1536.

Bakker JD, Meador-Sanchez AJ, Fule PZ, Huffman DW, Moore MM (2008) Growing trees backwards: description of a stand reconstruction model. In: *Fort Valley Experimental Forest – A Century of Research 1908–2008. Proceeding RMRS-P-53CD* (eds Olberding SD, Moore MM), USDA Forest Service, Rocky Mountain Research Station, Flagstaff, AZ.

Brown JK (1974) *Handbook for Inventorying Downed Woody Material. Gen. Tech. Rep. INT-16*. US Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.

Canadell JG, Raupach MR (2008) Managing forests for climate change mitigation. *Science*, **320**, 1456–1457.

Chambers JQ, Fisher JI, Zeng H, Chapman EL, Baker DB, Hurtt GC (2007) Hurricane Katrina's carbon footprint on U.S. gulf coast forests. *Science*, **318**, 1107.

Collins BM, Miller JD, Thode AE, Kelly M, Van Wagtendonk JW, Stephens SL (2009) Interactions among wildland fires in a long-established Sierra Nevada natural fire area. *Ecosystems*, **12**, 114–128.

Covington WW, Fule PZ, Moore MM *et al.* (1997) Restoring ecosystem health in ponderosa pine forests of the southwest. *Journal of Forestry*, **95**, 23–29.

Covington WW, Moore MM (1994) Southwestern ponderosa forest structure: changes since Euro-American settlement. *Journal of Forestry*, **92**, 39–47.

Dore S, Kolb TE, Montes-Helu M *et al*. (2008) Long-term impact of a stand-replacing fire on ecosystem CO_2 exchange of a ponderosa pine forest. *Global Change Biology*, **14**, 1–20.

Dore S, Kolb TE, Montes-Helu M *et al*. (2010) Carbon and water fluxes from ponderosa pine forests disturbed by wildfire and thinning. *Ecological Applications*, **20**, 663–683.

Fellows AW, Goulden ML (2008) Has fire suppression increased the amount of carbon stored in western U.S. forests? *Geophysical Research Letters*, **35**, L12404, doi: 10.1029/2008GL033965.

Ffolliot PF, Clary WP, Davis JR (1968) *Some Characteristics of the Forest Floor Under Ponderosa Pine in Arizona. Research Note Rm-127*. USDA Forest Service Rocky Mountain Forest and Range Experimental Station, Ft. Collins, CO.

Finkral AJ, Evans AM (2008) The effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. *Forest Ecology and Management*, **255**, 2743–2750.

Fulé PZ, Covington WW, Moore MM (1997) Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. *Ecological Applications*, **7**, 895–908.

Fulé PZ, Mchugh C, Heinlein TA, Covington WW (2001) Potential fire behavior is reduced following forest restoration treatments. In: *Ponderosa Pine Ecosystem Restoration and Conservation: Steps Toward Stewardship. Proceedings RMRS-P-22* (eds Vance RK, Edminster CB, Covington WW, Blake JA), pp. 28–35. Rocky Mountain Research Station. USDA Forest Service, Ogden, UT.

Galik CS, Jackson RB (2009) Risks to forest carbon offset projects in a changing climate. *Forest Ecology and Management*, **257**, 2209–2216.

Hurteau M, North M (2009) Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. *Frontiers in Ecology and the Environment*, **7**, 409–414.

Hurteau MD, Hungate BA, Koch GW (2009) Accounting for risk in valuing forest carbon offsets. *Carbon Balance Manag*, **4**, 1.

Hurteau MD, Koch GW, Hungate BA (2008) Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. *Frontiers in Ecology and the Environment*, **6**, 493–498.

Hurteau MD, North M (2010) Carbon recovery rates following different wildfire risk mitigation treatments. *Forest Ecology and Management*, **260**, 930–937.

Hurtt GC, Pacala SW, Moorcroft PR, Caspersen J, Shevliakova E, Houghton RA, Moore B III (2002) Projecting the future of the U.S. carbon sink. *Proceedings of the National Academy of Sciences*, **99**, 1389–1394.

Kaye JP, Hart SC, Fule PZ, Covington WW, Moore MM, Kaye MW (2005) Initial carbon, nitrogen, and phosphorous fluxes following ponderosa pine restoration treatments. *Ecological Applications*, **15**, 1581–1593.

Keane RE, Agee JK, Fulé P *et al.* (2008) Ecological effects of large fires on US landscapes: benefit or catastrophe? *International Journal of Wildland Fire*, **17**, 696–712.

Keith H, Mackey B, Berry S, Lindenmayer D, Gibbons P (2010) Estimating carbon carrrying capacity in natural forest ecosystems across heterogeneous landscapes: addressing sources of error. *Global Change Biology*, doi: 10.1111/j.1365-2486.2010.02178.x.

Keith H, Mackey BG, Lindenmayer DB (2009) Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proceedings of the National Academy of Sciences*, **106**, 11635–11640.

Korb JE, Covington WW, Fule P (2003) Sampling techniques influence understory plant trajectories after restoration: an example from ponderosa pine restoration. *Restoration Ecology*, **11**, 504–515.

Kurz WA, Dymond CC, Stinson G *et al*. (2008) Mountain pine beetle and forest carbon feedback to climate change. *Nature*, **452**, 987–990.

Laughlin DC (2009) *Functional consequences of long-term vegetation dynamics*. PhD dissertation, Northern Arizona University.

Littell JS, Mckenzie D, Peterson DL, Westerling AL (2009) Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. *Ecological Applications*, **19**, 1003–1021.

Meigs GW, Donato DC, Campbell JL, Martin JG, Law BE (2009) Forest fire impacts on carbon uptake, storage, and emission: the role of burn severity in the Eastern Cascades, Oregon. *Ecosystems*, **12**, 1246–1267.

Miller JD, Safford HD, Crimmins M, Thode AE (2009) Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. *Ecosystems*, **12**, 16–32.

Mitchell SR, Harmon ME, O'connell KEB (2009) Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. *Ecological Applications*, **19**, 643–655.

Moore MM, Covington WW, Fulé PZ (1999) Reference conditions and ecological restoration: a southwestern ponderosa pine perspective. *Ecological Applications*, **9**, 1266–1277.

Moore MM, Huffman DW, Fulé PZ, Covington WW, Crouse JE (2004) Comparison of historical and contemporary forest structure and composition on permanent plots in southwestern ponderosa pine forests. *Forest Science*, **50**, 162–176.

National Oceanic and Atmospheric Administration (2005) National Oceanic and Atmospheric Administration. Available at http://www.noaa.gov

North M, Hurteau M, Innes J (2009) Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. *Ecological Applications*, **19**, 1385–1396.

Penman JM, Gytarzky M, Hiraishi M *et al*. (2003) *Good Practice Guidance for Land use, Land use Change, and Forestry*. Institute for Global Environmental Strategies for the Intergovernmental Panel on Climate Change, Hayama, Kanagawa, Japan.

Reinhardt E, Holsinger L (2010) Effects of fuel treatments on carbon-disturbance relationships in forests of the northern rocky mountains. *Forest Ecology and Management*, **259**, 1427–1435.

Roccaforte JP, Fule PZ, Covington WW (2008) Landscape-scale changes in canopy fuels and potential fire behaviour following ponderosa pine restoration treatments. *International Journal of Wildland Fire*, **17**, 293–303.

Sackett SS (1980) *Reducing Natural Ponderosa Pine Fuels Using Prescribed Fire: Two Case Studies. Research Note RM-392*. USDA Forest Service, Ft. Collins, CO.

Savage M, Mast JN (2005) How resilient are southwestern ponderosa pine forests after crown fires? *Canadian Journal of Forest Research*, **35**, 967–977.

Scott JH, Reinhardt ED (2001) *Assessing Crown Fire Potential by Linking Models of Surface and Crown Fire Behavior. Res. Pap. Rmrs-Rp-29*. USDA Forest Service RMRS, Fort Collins, CO.

Seager R, Ting M, Held I *et al*. (2007) Model projections of an imminent transition to a more arid climate in southwestern North America. *Science*, **316**, 1181–1184.

Smith JE, Heath LS (2002) *A Model of Forest Floor Carbon Biomass for United States Forest Types. Paper NE-722*. Department of Agriculture, Forest Service, Northeastern Research Station, Newton Square, PA.

Stephens SL, Moghaddas JJ (2005) Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. *Forest Ecology and Management*, **215**, 21–36.

Stephens SL, Moghaddas JJ, Edminster C *et al*. (2009a) Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. *Ecological Applications*, **19**, 305–320.

Stephens SL, Moghaddas JJ, Hartsough BR, Moghaddas EEY, Clinton NE (2009b) Fuel treatment effects on stand-level carbon pools, treatment-related emissions, and fire risk in a Sierra Nevada mixed-conifer forest. *Canadian Journal of Forest Research*, **39**, 1538–1547.

Swetnam TW, Allen CD, Betancourt JL (1999) Applied historical ecology: using the past to manage for the future. *Ecological Applications*, **9**, 1189–1206.

Van Mantgem PJ, Stephenson NL, Byrne JC *et al*. (2009) Widespread Increase of tree mortality rates in the Western United States. *Science*, **323**, 521–524.

Westerling AL, Bryant BP (2008) Climate change and wildfire in California. *Climatic Change*, **87** (Suppl. 1), S231–S249.

Westerling AL, Hidalgo HG, Cayan DR, Swetnam TW (2006) Warming and earlier spring increase western U.S. forest wildfire activity. *Science*, **313**, 940–943.

White AS (1985) Presettlement regeneration patterns in a southwestern ponderosa pine stand. *Ecology*, **66**, 589–594.

Wiedinmyer C, Hurteau MD (2010) Prescribed fire as a means of reducing forest carbon emissions in the western U.S. *Environmental Science and Technology*, **44**, 1926–1932.

Wiedinmyer C, Neff JC (2007) Estimates of CO_2 from fires in the United States: implications for carbon management. *Carbon Balance Manag*, **2**, 10.

Woodbury PB, Smith JE, Heath LS (2007) Carbon sequestration in the U.S. forest sector from 1990 to 2010. *Forest Ecology and Management*, **241**, 14–27.