Short- and Long-term Effects of Fire on Carbon in US Dry Temperate Forest Systems

MATTHEW D. HURTEAU AND MATTHEW L. BROOKS

Forests sequester carbon from the atmosphere, and in so doing can mitigate the effects of climate change. Fire is a natural disturbance process in many forest systems that releases carbon back to the atmosphere. In dry temperate forests, fires historically burned with greater frequency and lower severity than they do today. Frequent fires consumed fuels on the forest floor and maintained open stand structures. Fire suppression has resulted in increased understory fuel loads and tree density; a change in structure that has caused a shift from low- to high-severity fires. More severe fires, resulting in greater tree mortality, have caused a decrease in forest carbon stability. Fire management actions can mitigate the risk of high-severity fires, but these actions often require a trade-off between maximizing carbon stocks and carbon stability. We discuss the effects of fire on forest carbon stocks and recommend that managing forests on the basis of their specific ecologies should be the foremost goal, with carbon sequestration being an ancillary benefit.

Keywords: carbon stability, climate change, fire severity, forest management, fuels management

Fire is one of the oldest tools used by humans, and it remains a major factor in land management worldwide (Bowman et al. 2009). Fire management is typically performed for two primary purposes: (1) managing fuels and controlling fire to protect human life and infrastructure; and (2) using fire and fire surrogates (e.g., selective forest thinning) to promote desired future conditions of ecosystems services, biodiversity, wildlife habitat, and commodity production, among others. Carbon management has recently emerged as an additional focus of land management because of growing concerns about the climate effects of rising atmospheric greenhouse gas concentrations.

It would appear that fire is a threat to carbon stocks that should be suppressed if maximizing carbon stocks is an important objective. However, the reality is much more complicated, given the multivariate interactions between vegetation and fire regimes and the potential for changing climatic conditions to influence the prevalence of fire at both regional and global scales (Westerling and Bryant 2008, Liu et al. 2009). A policy of full fire suppression also runs counter to many other important land-management considerations involving sensitive species, biodiversity, and watershed function, among others. It is therefore critical to understand the full implications of alternative fire-management actions, from the perspective of carbon management and other land-management goals, before policy is established or revised.

Carbon sequestration in forests is one of a range of strategies that can be used to mitigate human-caused climate change (Pacala and Socolow 2004). The climate change mitigation potential of forests can be improved by reducing deforestation, increasing the land area that is forested (afforestation and reforestation), and enhancing forest carbon density (Canadell and Raupach 2008). However, sequestering carbon in forests is not without potential risks and drawbacks. Forests can influence biophysical feedbacks within the climate system by reducing the amount of light energy reflected back to the atmosphere from a given land area, thereby causing more solar radiation to be absorbed by the earth (Bonan 2008, Jackson et al. 2008). Forest carbon can also be returned to the atmosphere as a result of both natural and human-caused disturbances (Gullison et al. 2007, Galik and Jackson 2009, Hurteau et al. 2009). In the case of wildfire, the reversal risk can be large. Carbon emissions from fire in the United States are equivalent to 4% to 6% of annual human-caused carbon emissions (Wiedinmyer and Neff 2007). At the state level, the contribution of fire emissions to total annual carbon emissions can be even larger. Campbell and colleagues (2007) reported that carbon emissions from the 2002 Biscuit Fire in Oregon were equivalent to approximately one-third of the fossil fuel–based emissions in the entire state during that year.

Although some risks to sequestered forest carbon are largely beyond the control of humans (e.g., lightning), others are completely manageable (e.g., land-use conversion). Fire management falls in the middle of this continuum, because the manipulation of fuels and the suppression or promotion of fire can influence the frequency, severity, and ultimate effects of fire. Fire regimes and fire effects vary significantly across ecosystem and vegetation types, and the risk of fires to

BioScience 61: 139–146. ISSN 0006-3568, electronic ISSN 1525-3244. doi:10.1525/bio.2011.61.2.9

carbon stocks and the potential for humans to mitigate this risk are largely dependent upon the particular forest system and the prevailing climatic conditions.

In this article we describe the potential short- and long-term effects of fire on above- and belowground carbon stocks in US dry temperate forests. We also examine the trade-offs between management approaches focused on maximizing versus stabilizing aboveground carbon stocks in this ecosystem type. We define carbon stock stabilization as reducing the risk of carbon being returned to the atmosphere through combustion. Dry temperate forests occur worldwide; in North America they are most prevalent in the southwestern part of the continent. Before the implementation of fire-suppression policy during the early 1900s, this forest type experienced frequent fires (on the order of several years to several decades) that maintained lower fuel loads and tree densities (Agee and Skinner 2005). We chose to focus on dry temperate forests because worldwide, much of the forested area burned and fire-management actions implemented occur in seasonally dry vegetation types with intermediate productivity levels (Bowman et al. 2009). In addition, warming climatic conditions are predicted to interact with patterns of fire frequency and have potentially significant effects on carbon stocks in these forests (Westerling et al. 2006).

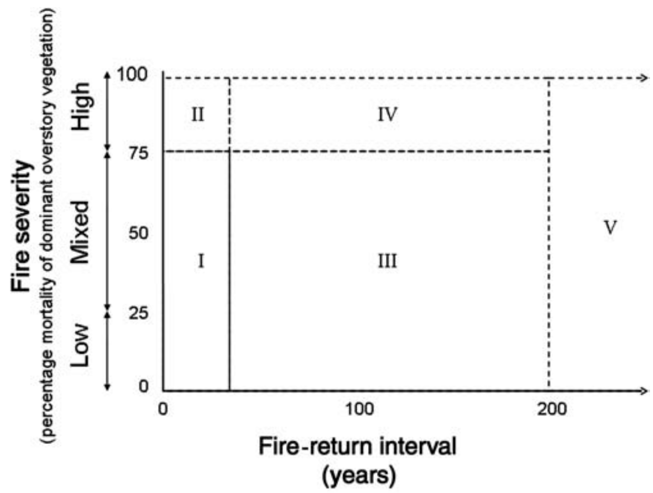

Figure 1. As the interval between fire events in dry temperate forests in the western United States increases, fire severity, defined as the percent mortality of the dominant overstory vegetation, increases. Roman numerals represent a range of fire-return intervals and severities (I, 0 to 35 years, low to mixed severity; II, 0 to 35 years, high severity; III, 35 to 200 years, low to mixed severity; IV, 35 to 200 years, high severity; V, 200 or more years, any severity). Adapted from figure 3-4 of the FRCC Guidebook v. 1.3.0, June 2008.

Short-term effects of fire

The transfer of carbon from the atmosphere to plants occurs through photosynthesis. Plants take in carbon dioxide from the atmosphere, energy from the sun, and nutrients and water from the soil and then assimilate the carbon into tissue. The carbon stored in plants can follow multiple subsequent pathways such as herbivory, harvest, and decomposition, among others. All pathways typically result in carbon being cycled through the decomposition process at some point. The primary source of carbon dioxide to the atmosphere from decomposition is mineralization, a process by which fungi and bacteria break down plant material (Schlesinger 1997).

In the short term, fire can influence the carbon cycle in a number of ways. Fire can affect plant growth directly by killing plants, thereby preventing them from sequestering additional carbon. When smoldering combustion occurs, it can produce charcoal or black carbon, which is the result of incomplete fuel combustion. Carbon in this form is relatively stable and can remain in the system for considerable periods of time (DeLuca and Aplet 2008). When fire consumes vegetation and detritus it releases carbon back to the atmosphere, and it can release nutrients to the soil—potentially increasing postfire vegetation growth. Fire can also provide a competitive advantage for some species, which may have implications for postdisturbance productivity as a function of fire frequency and severity.

Fire intensity tends to correlate with fire severity. As the energy released during combustion (intensity) increases, the effect that fire has on the system, such as plant mortality (severity), also increases (Keeley 2009). The time between fire events (fire-return interval) in part determines fire severity as well (figure 1). When fires are frequent, there is less fuel buildup and fire intensity is lower; when fire is infrequent, fuel buildup can be substantial, resulting in greater fire intensities (van Wagtendonk 1984). Accordingly, when fire intensity is low and frequency is high, plant mortality rates tend to be low. In this scenario, the remaining live plants experience reduced competition from neighboring plants, which can enhance their rates of photosynthesis and carbon assimilation above the rate that carbon is released back to the atmosphere through decomposition of dead plant material. In contrast, when fire intensity is high and frequency is low, plant mortality is high and the balance between carbon assimilation and carbon emission can become negative, making the site a net source of carbon to the atmosphere (Dore et al. 2008, Meigs et al. 2009).

During the combustion process, biomass may be heated in the absence of oxygen, leading to the formation of black carbon (DeLuca and Aplet 2008). Approximately 1% to 3% of the biomass in a burn area is converted to black carbon (Preston 2009). Although this represents a relatively small proportion of the carbon balance of fire, black carbon is a fairly stable form of carbon that can accumulate in soils (DeLuca and Aplet 2008). However, recent research on black carbon in boreal forest soils indicates that the concentration in the soil is quite variable in space and tends to decrease over time (Ohlson et al. 2009). This reduction may be due to the susceptibility of black carbon to microbial breakdown into water-soluble compounds that can be leached from the soil (Hockaday et al. 2007).

Fire is a combustion process that directly or indirectly releases carbon to the atmosphere as biomass is consumed. Direct fire emissions represent a short-term release. A large proportion of direct fire emissions results from the consumption of surface fuels, which comprise leaves, branches, coarse woody debris, and other organic material on the forest floor (Campbell et al. 2007, Meigs et al. 2009). The quantity of direct emissions is in part a function of fire intensity. High-intensity fires, such as some wildfires, produce more carbon emissions than do low-intensity fires, such as prescribed fires (Wiedinmyer and Hurteau 2010).

In fire-prone forested systems, fire was more prevalent in the past than it is today (Stephens et al. 2007). These fires, whether natural or human caused, released considerable amounts of carbon to the atmosphere. Estimated pre-1800 fire emissions of carbon dioxide from forests in California range from 23.1 to 62.6 teragrams (Tg) carbon dioxide per year (Stephens et al. 2007). The average annual estimate of carbon dioxide emissions from fire in California from 2001 to 2008 was 17.8 Tg carbon dioxide per year. However, 2008 had substantially higher emissions (54.5 Tg carbon dioxide) as a result of a large number of lightning-ignited fires (Wiedinmyer and Hurteau 2010). Although recent annual fire emissions are either below or within the range of historical emissions of carbon dioxide in California, they are well below the upper bound.

Fire can also influence belowground carbon stocks. Although some studies suggest that the impacts of fire on the soil carbon pool are relatively small (Wirth et al. 2002, Kashian et al. 2006), burn severity and soil drainage can influence the soil carbon stock (Harden et al. 2000). A recent study of the impacts of high-severity wildfire on the soil carbon pool in Oregon found that approximately 60% of the carbon contained in the mineral horizons was released by the Biscuit Fire there in 2002 (Bormann et al. 2008). This soil carbon loss is thought to be largely the result of soil erosion from significant vegetation removal and steep slopes, and this has implications for future productivity (Bormann et al. 2008).

Although understanding the short-term effects of fire on a system and the emissions associated with fire is important for informing management decisions and managing air quality, these effects must be viewed over the long term to better account for the effects of fire on carbon stocks.

Long-term effects of fire

Over the long term, fire effects on terrestrial carbon stocks are a function of the balance between carbon loss from direct fire emissions and decomposition and carbon gain from vegetation regrowth. Indirect fire emissions result from the decomposition of vegetation killed but not consumed by fire; this source can be as much as three times the size of direct carbon emissions (Auclair and Carter 1993). The amount of dead biomass that remains following a fire event is largely a function of fire severity. Low-severity fire consumes less fuel and kills few trees (Agee and Skinner 2005, Hurteau and North 2009, Meigs et al. 2009); in contrast, when fire severity is high, more fuel is consumed and tree mortality rates are higher (Agee and Skinner 2005, Meigs et al. 2009). Tree mortality rates influence indirect emissions because high tree mortality transfers carbon that was stored in live trees to the dead tree pool, which is subject to decomposition (Kashian et al. 2006).

If the successional pathway that resulted in the prefire forest remains unchanged, the recovering forest will transition from a carbon source to a carbon sink, and with sufficient time the forest will resequester all of the carbon lost from both direct and indirect sources (figure 2; Kashian et al. 2006). However, Meigs and colleagues (2009) reported that four to five years postfire, high-severity burned areas in mixed-conifer forest and moderate- and high-severity burned areas in ponderosa pine forests of the eastern Cascades, in Oregon, continued to be net sources of carbon to the atmosphere. Also, in a ponderosa pine forest that burned under high-severity conditions in Arizona, Dore and colleagues (2008) reported that the site remained a net source of carbon to the atmosphere 10 years postfire, and that it is unlikely the site will become a net sink in the near future, a result of slow vegetation recovery. The potential also exists for type conversion from forest to a different vegetation type (e.g., shrubland or grassland) following some high-severity fires (figure 2). Savage and Mast (2005) surveyed 10 sites where stand-replacing wildfire had occurred in southwestern ponderosa pine forest. In 50% of these sites, the lack of tree regeneration indicated that the sites had transitioned from a forest to a grassland or shrubland with a diminished capacity to sequester carbon. Thus, the long-term effects of high-severity fire and the potential for type conversion have substantial implications for the carbon balance of dry temperate forests (figure 2).

Rising temperatures and the associated earlier spring snowmelt correlate with increasing wildfire size in the western United States, in part because these factors lengthen the fire season (Westerling et al. 2006). Higher temperatures are also thought to exacerbate vegetation mortality rates during severe drought conditions (Breshears et al. 2005, van Mantgem et al. 2009) and increase carbon emissions from the decomposition of dead plant material (Kirschbaum 1995). Climate change projections for southwestern North America suggest that regardless of precipitation trends, the region will become more water stressed because of the effects of higher temperatures on evaporation rates (Seager et al. 2007). This combination of factors has the potential to reduce carbon stocks and net ecosystem productivity if the successional pathway of forested systems is altered by stand-replacing fires and those forest stands more frequently transition into more drought-tolerant grassland and shrubland vegetation types (figure 2).

Maximizing versus stabilizing carbon stocks

Broadly speaking, there are two approaches for carbon sequestration in dry temperate forests: carbon maximization and carbon stabilization. Carbon maximization can be achieved by increasing the carbon density, on a relative scale, per unit of land area (figure 3a). However, the carbon maximization approach neglects the influence of changing climatic conditions and stand density on fire weather, fire behavior, fire severity, and tree mortality, and ultimately the potential for (a) a very slow forest recovery that would approximate the shape of the carbon stock curve in figure 3a (but drawn out over a longer period of time), or (b) vegetation-type conversion (figure 3b).

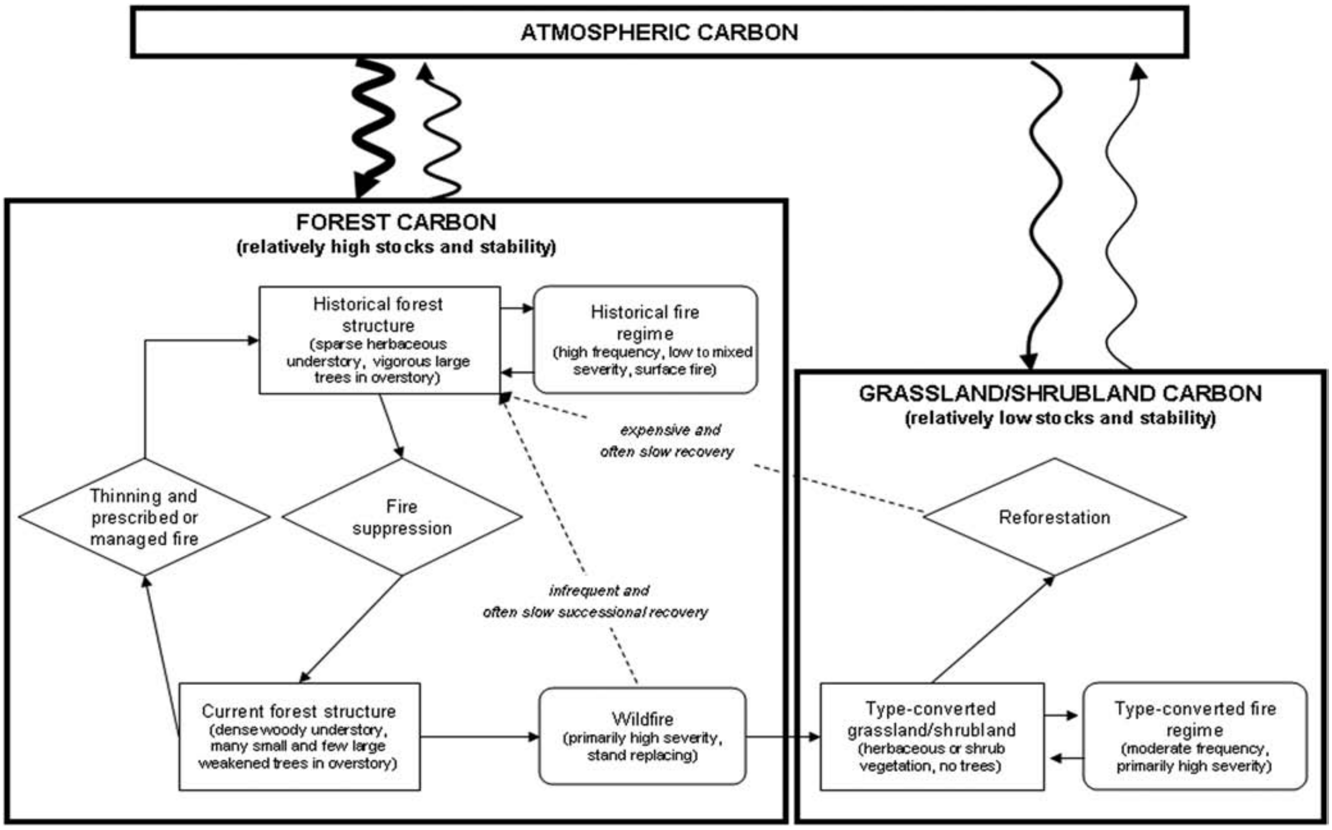

Figure 2. Dry temperate forest structures, characterized by low understory fuel loads and larger trees with a heterogeneous distribution, were historically maintained by a frequent, low- or mixed-severity surface-fire regime, resulting in relatively stable carbon stocks. The implementation of fire-suppression policy has shifted forests to a dense structure with a more homogenous distribution of the forest overstory dominated by smaller trees and high fuel loads, which is conducive to high-severity, stand-replacing fire and lessened aboveground carbon stock stability (e.g., maintaining the aboveground carbon stock over time). Wildfire in these altered forests can be followed by slow successional recovery to a forested condition or by type conversion to grassland or shrubland vegetation. Three management alternatives are currently available: (1) continue fire suppression, which will likely result in additional type conversion of forest to grassland or shrubland, with smaller and less stable aboveground carbon stocks; (2) implement thinning and prescribed burning or managed fire to restore historical forest structure and fire regimes that maintain aboveground carbon stocks and maximize their stability; or (3) reforest landscapes already converted to grassland or shrubland, restoring forest condition and carbon sequestration capacities.

Alternatively, carbon stabilization is focused on minimizing the potential fire-induced loss of carbon from the system by altering stand structure to reduce the risk of high-severity, stand-replacing fire (figure 3c). Although carbon maximization and stabilization may be mutually exclusive in a fire-prone forest, they should be thought of as end points on a spectrum of options rather than as two dichotomous objectives. The range of options is continuous and the role of fire management ranges from active fire suppression to an intensive burning program, depending on other natural resource or fuels-management objectives.

Maximizing carbon stocks by protecting them from fire

Fire is generally thought to pose a threat to carbon stocks, and fire suppression is thought to have contributed to growth in forest carbon stocks during the 20th century (Hurtt et al. 2002). Before the implementation of fire suppression policy during the early 1900s, dry temperate forests were maintained by frequent, low-severity fires, and forest structure was dominated by fewer larger trees at lower densities (Covington et al. 1997, Stephens and Fulé 2005, North et al. 2007). Fire suppression in these forests has led to an ingrowth of trees that would seem to lead to larger carbon stocks. However, in some dry temperate forests, such as mixed-conifer stands in the Sierra Nevada Mountains, reductions in the number of large trees have resulted in an overall reduction in the amount of carbon stored in live trees (Fellows and Goulden 2008, North et al. 2009). Thus, simply protecting forests from burning may not be a sustainable approach for maximizing carbon stocks.

Even if fire suppression efforts continue to be successful in the future, the sink strength of forests in the United States is projected to decline because of an equilibration among vegetation growth, harvesting practices, and tree mortality (Hurtt et al. 2002). In addition, warming climatic conditions

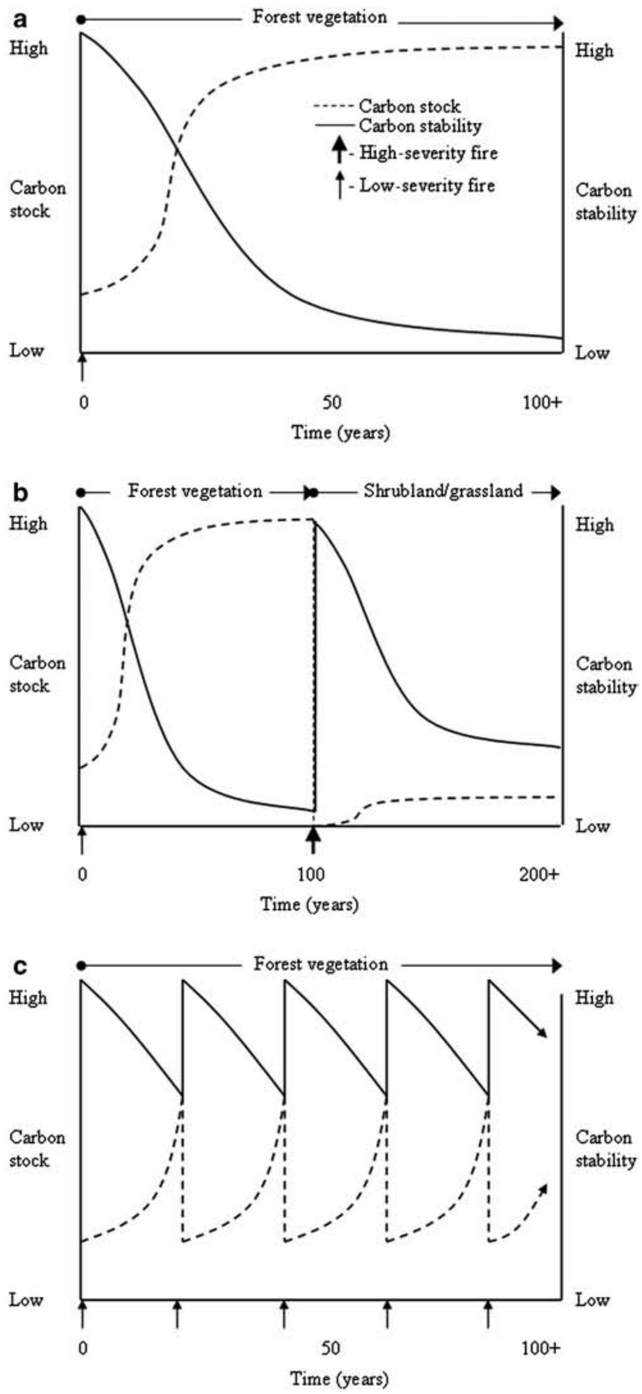

Figure 3. Theoretical relationship between aboveground carbon stock and carbon stability in a dry temperate forest with no fire (a), where an increase in the aboveground carbon stock in the absence of frequent fires increases the potential for high-severity fire followed by carbon-depleting vegetation type conversion from forest to shrubland or grassland (b), and frequent low-severity fires, consistent with many current fire-management prescriptions and achieving maximized carbon stability (c). Note the longer time scale on panel (b) compared with panels (a) or (c). Also note that all panels have the same initial condition, beginning with a low-severity fire, with each representing a different subsequent fire scenario.

and the forest fuel conditions created by a century of fire suppression have led to greater fire size and severity, a trend predicted to continue in the future (Westerling et al. 2006, North et al. 2007, Westerling and Bryant 2008, Miller et al. 2009). Thus, it seems that fire suppression will become increasingly difficult as we make our way through the 21st century. So how do we balance the trade-offs associated with maintaining forest carbon stocks with managing fire risk at acceptable levels?

Stabilizing carbon stocks using fire and other tools

It is undeniable that individual fires consume biomass and release carbon into the atmosphere. However, these instantaneous effects can often be balanced or exceeded by subsequent compensatory regrowth of vegetation (Kashian et al. 2006). The net change of carbon contained in vegetation relative to prefire levels depends on the time since burning, fire severity, weather, topographic position, the type of vegetation that actually grows back, and postfire management actions. It may take several centuries for a forest to recover from a high-severity fire (Kashian et al. 2006), but forests that burn at lower severities may be able to replace biomass lost to fire over decadal time scales (figure 2; Hurteau and North 2009, 2010). Frequent fire in dry temperate forests appears to select for lower-density forest stands with larger-diameter trees (Stephens and Gill 2005, North et al. 2007). In some systems, these stands store a greater volume of carbon per unit area than the stands they replace, whereas in others, the fire-suppressed structure contains a larger volume of carbon per unit area (Fellows and Goulden 2008, North et al. 2009, Hurteau et al. 2010). Even more important, forests with larger-diameter trees of fire-resistant species have a complex structure that often includes a high height-to-live-crown ratio, making them less susceptible to stand-replacing crown fires and type conversions to other vegetation types, thus promoting long-term carbon stability (figure 2; Stephens et al. 2008, Hurteau and North 2009).

Reducing the density of trees may initially require mechanical thinning before prescribed burning. This structural manipulation typically involves thinning from below, or removing smaller-diameter trees and leaving larger-diameter trees. In some forest types, silvicultural prescriptions may also preferentially reduce the abundance of certain species. Although reducing the risk of high-severity fire by thinning does result in an initial reduction in the live-tree carbon stock (Finkral and Evans 2008, North et al. 2009, Stephens et al. 2009, Dore et al. 2010), thinning a forest and then carrying out prescribed burning can reduce future tree mortality rates and carbon emissions caused by wildfire (Agee and Skinner 2005, Hurteau and North 2009). In addition, surviving trees continue to sequester carbon following wildfire, which must also be factored into the net carbon balance equation.

There are several carbon management issues to consider when implementing mechanical thinning treatments to reduce the risk of high-severity fire. Central to these issues is the natural role that fire plays in a particular system. For example, thinning treatments to reduce high-severity fire generally are not warranted in vegetation types such as the wet coastal forests of the Pacific Northwest, where stand-replacing fire is a natural occurrence, tree species composition is largely unaffected by disturbance, and reducing carbon emissions from wildfire requires a much larger removal of carbon from the system than is lost during a fire event (Ohmann et al. 2007, Mitchell et al. 2009). In addition, the fire-return intervals in these systems are naturally longer than the period of fire suppression that began in the early 1900s, so most wet forest stands may be well within their historical fuel conditions and capable of full recovery following a fire event. In contrast, thinning treatments may garner a carbon management benefit in the form of avoided emissions from wildfire and greater tree survivorship in vegetation types such as dry temperate forests, where low- or mixed-severity fires were historically the primary fire type (Hurteau et al. 2008). In these systems, thinning beneath the forest canopy to remove small-diameter trees, which act as ladder fuels, and reduce surface fuels provides the greatest carbon management benefit, as most of the tree carbon is stored in larger trees. This type of structural manipulation typically involves removing between 26% and 34% of the live-tree carbon (Finkral and Evans 2008, North et al. 2009, Stephens et al. 2009). More intensive tree removal is counterproductive from a carbon management perspective and adds little value in terms of reducing high-severity fire risk (North et al. 2009, Hurteau and North 2010).

Another consideration is the level of thinning treatment. Incomplete treatments, such as those that neglect surface fuels or insufficiently reduce canopy bulk density or ladder fuels, have little impact on reducing fire severity (Safford et al. 2009). Furthermore, regular prescribed fires or other management fires are necessary following thinning treatments to manage surface fuels and maintain high-severity fire resistance (Hurteau and North 2009). The final considerations relate to fossil-fuel use for treatment implementation and the fate of the carbon removed during thinning treatments. Fossil fuel used for mechanical treatment and hauling logs to the mill equates to a small fraction (0.4% to 0.5%) of the carbon stored in the posttreatment forest (Finkral and Evans 2008, North et al. 2009). The fate of the harvested tree carbon can be central to the carbon balance. For example, using thinned trees for firewood and accounting for the reduction in fossil fuel used for home heating can result in a net carbon loss of 3.11 megagrams (Mg) carbon per hectare (ha), whereas using the thinned material for longer-lived wood products results in a net gain of 3.35 Mg carbon per ha in southwestern ponderosa pine forests (Finkral and Evans 2008). In Sierra Nevada mixed-conifer forest, understory tree removal can yield a substantial number of trees that are appropriate for dimensional lumber production, with lumber being equivalent to 6.4% to 8.9% of the total posttreatment carbon pool (North et al. 2009). North and colleagues (2009) reported that the waste associated with milling inefficiency is second only to prescribed fire emissions in understory thinning. If the milling waste is used as biofuel to generate electricity, the carbon contained in this material can be used to offset fossil fuel–based energy.

The concept of carbon carrying capacity, the amount of carbon that can be stored in a system as a function of prevailing climatic conditions and natural disturbance regimes, has been proposed as a potential foundation for carbon management plans (Keith et al. 2009, 2010, Hurteau et al. 2010). Managing within the carbon carrying capacity for dry temperate forests requires incorporating an understanding of fire and stand dynamics (North et al. 2009). Altering forest structure by thinning smaller trees and then carrying out prescribed burning aggregates carbon into fewer larger trees and reduces the potential for high-severity fire (Stephens and Moghaddas 2005, Finkral and Evans 2008, Hurteau and North 2009, North et al. 2009). These actions may reduce the amount of standing carbon in trees, but they will improve the stability of these carbon stocks over time. Management objectives in this context should be focused on achieving a balance between carbon stock size and carbon stabilization that falls within the carbon carrying capacity of the forest.

Conclusions

Forests provide a suite of ecosystem services, including carbon sequestration for mitigating human-caused climate change. However, even if forest-based carbon sequestration were maximized to achieve the 1 gigaton of carbon per year required to mitigate one-seventh of the global emissions projected by Pacala and Socolow (2004), reduced fossil-fuel consumption would still be required to lower atmospheric carbon dioxide concentration. Thus, forests offer a bridging strategy and are only part of the climate change mitigation portfolio (McCarl and Sands 2007). Although forest carbon sequestration does carry a risk of reversal, even impermanent carbon offsets generated by increasing aboveground forest carbon stocks can serve to reduce compliance costs in a cap-and-trade system, and in the case of fire, this risk can be reduced (Hurteau et al. 2009, Mignone et al. 2009). However, mitigating fire risk in dry temperate forests requires periodic carbon emissions from prescribed burning or allowing natural fires to burn under certain circumstances (i.e., managed fire). In addition to improving aboveground forest carbon stability, managing these forests in ways that maximize their resilience to fire also provides for a fully functioning ecosystem, which is consistent with a wide array of other land-management goals. As such, we recommend managing forests on the basis of their specific ecologies, with the view that carbon sequestration is one of many ancillary ecosystem services.

Acknowledgments

The authors thank Karen Phillips, Nathan Stephenson, Leland Tarnay, and four anonymous reviewers for helpful comments on previous versions of this manuscript. MDH acknowledges support from Cooperative Agreement 08-CA-11272170-102 with the US Department of Agriculture Forest Service Pacific Southwest Research Station, using funds provided by the Bureau of Land Management through the sale of public lands as authorized by the Southern Nevada Public Land Management Act. MLB acknowledges support from the

US Geological Survey Terrestrial, Freshwater, and Marine Ecosystems Program, and the National Park Service Fire and Aviation Management Program (Interagency Agreement F8803090011) and Climate Change Response Program (Interagency Agreement F8803100033).

References cited

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.

Bonan GB. 2008. Forests and climate change: Forcings, feedbacks, and the climate benefits of forests. Science 320: 1444–1449.

Bormann BT, Homann PS, Darbyshire RL, Morrissette BA. 2008. Intense forest wildfire sharply reduces mineral soil C and N: The first direct evidence. Canadian Journal of Forest Research 38: 2771–2738.

Bowman DMJS, et al. 2009. Fire in the Earth system. Science 324: 481–484.

Breshears DD, et al. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences 102: 15144–15148.

Campbell J, Donato D, Azuma D, Law B. 2007. Pyrogenic carbon emission from a large wildfire in Oregon, United States. Journal of Geophysical Research 112: G04014.

Canadell JG, Raupach MR. 2008. Managing forests for climate change mitigation. Science 320: 1456.

Covington WW, Fulé PZ, Moore MM, Hart SC, Kolb TE, Mast JN, Sackett SS, Wagner MR. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. Journal of Forestry 95: 23–29.

DeLuca TH, Aplet GH. 2008. Charcoal and carbon storage in forest soils of the Rocky Mountain West. Frontiers in Ecology and the Environment 6: 18–24.

Dore S, Kolb TE, Montes-Helu M, Sullivan BW, Winslow WD, Hart SC, Kaye JP, Koch GW, Hungate BA. 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, Eckert SE, Sullivan BW, Hungate BA, Kaye JP, Hart SC, Koch GW, Finkral AJ. 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.

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.

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

Gullison RE, et al. 2007. Tropical forests and climate policy. Science 316: 985–986.

Harden JW, Trumbore SE, Stocks BJ, Hirsch A, Gower ST, O'Neill KP, Kasischke ES. 2000. The role of fire in the boreal carbon budget. Global Change Biology 6 (suppl. 1): 174–184.

Hockaday WC, Grannas AM, Kim S, Hatcher PG. 2007. The transformation and mobility of charcoal in a fire-impacted watershed. Geochimica et Cosmochimica Acta 71: 3432–3445.

Hurteau MD, 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.

———. 2010. Carbon recovery rates following different wildfire risk mitigation treatments. Forest Ecology and Management 260: 930–937.

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, Hungate BA, Koch GW. 2009. Accounting for risk in valuing forest carbon offsets. Carbon Balance and Management 4: 1.

Hurteau MD, Stoddard MT, Fulé PZ. 2010. The carbon costs of mitigating high-severity wildfire in southwestern ponderosa pine. Global Change Biology. doi:10.1111/j.1365–2486.2010.02295.x

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.

Jackson RB, et al. 2008. Protecting climate with forests. Environmental Research Letters 3: 044006.

Kashian DM, Romme WH, Tinker DB, Turner MG, Ryan MG. 2006. Carbon storage on landscapes with stand-replacing fires. BioScience 56: 598–606.

Keeley JE. 2009. Fire intensity, fire severity and burn severity: A brief review and suggested usage. International Journal of Wildland Fire 18: 116–126.

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.

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.1365–2486.2009.02146.x

Kirschbaum MUF. 1995. The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic carbon storage. Soil Biology and Biochemistry 27: 753–760.

Liu Y, Stanturf JA, Goodrick SL. 2009. Trends in global wildfire potential in a changing climate. Forest Ecology and Management 259: 685–697.

McCarl BA, Sands RD. 2007. Competitiveness of terrestrial greenhouse gas offsets: Are they a bridge to the future? Climatic Change 80: 109–126.

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.

Mignone BK, Hurteau MD, Chen Y, Sohngen B. 2009. Carbon offsets, reversal risk and US climate policy. Carbon Balance and Management 4: 3.

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.

North M, Innes J, Zald H. 2007. Comparison of thinning and prescribed fire restoration treatments to Sierran mixed-conifer historic conditions. Canadian Journal of Forest Research 37: 331–342.

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.

Ohlson M, Dahlberg B, Okland T, Brown KJ, Halvorsen R. 2009. The charcoal carbon pool in boreal forest soils. Nature Geoscience 2: 692–695.

Ohmann JL, Gregory MJ, Spies TA. 2007. Influence of environment, disturbance, and ownership on forest vegetation of coastal Oregon. Ecological Applications 17: 18–33.

Pacala S, Socolow R. 2004. Stabilization wedges: Solving the climate problem for the next 50 years with current technologies. Science 305: 968–972.

Preston CM. 2009. Biogeochemistry: Fire's black legacy. Nature Geoscience 2: 674–675.

Safford HD, Schmidt DA, Carlson CH. 2009. Effects of fuel treatments on fire severity in an area of wildland-urban interface, Angora Fire, Lake Tahoe Basin, California. Forest Ecology and Management 258: 773–787.

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

Schlesinger WH. 1997. Biogeochemistry: An Analysis of Global Change. Academic Press.

Seager R, et al. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. Science 316: 1181–1184.

Stephens SL, Fulé PZ. 2005. Western pine forests with continuing frequent fire regimes: Possible reference sites for management. Journal of Forestry 103: 357–362.

Stephens SL, Gill SJ. 2005. Forest structure and mortality in an old-growth Jeffrey pine–mixed conifer forest in north-western Mexico. Forest Ecology and Management 205: 15–28.

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, Martin RE, Clinton NE. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management 251: 205–216.

Stephens SL, Fry DL, Franco-Vizcaino E. 2008. Wildfire and spatial patterns in forests in northwestern Mexico: the United States wishes it had similar fire problems. Ecology and Society 13: 10.

Stephens SL, Moghaddas JJ, Hartsough BR, Moghaddas EEY, Clinton NE. 2009. 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.

van Mantgem PJ, et al. 2009. Widespread increase of tree mortality rates in the western United States. Science 323: 521–524.

van Wagtendonk JW. 1984. Fire suppression effects on fuels and succession in short-fire-interval wilderness ecosystems. Pages 119–126 in Proceeding, Symposium, and Workshop on Wilderness Fire, November 15–18, 1983. US Department of Agriculture Forest Service. General Technical Report INT-182.

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.

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 and Management 2: 10.

Wirth C, Czimczik CI, Schulze E-D. 2002. Beyond annual budgets: Carbon flux at different temporal scales in fire-prone Siberian Scots pine forests. Tellus 54: 611–630.

Matthew Hurteau (matthew.hurteau@nau.edu) is a forest ecologist at Northern Arizona University. He studies climate change mitigation and adaption in fire-prone forests. Matthew Brooks is a research botanist at the US Geological Survey, Western Ecological Research Center, Yosemite Field Station. He studies fire ecology and the effects of land-management actions in shrubland and forest ecosystems.