Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets

Matthew D Hurteau*, George W Koch, and Bruce A Hungate

Management of forests for carbon uptake is an important tool in the effort to slow the increase in atmospheric CO_2 and global warming. However, some current policies governing forest carbon credits actually promote avoidable CO_2 release and punish actions that would increase long-term carbon storage. In fire-prone forests, management that reduces the risk of catastrophic carbon release resulting from stand-replacing wildfire is considered to be a CO_2 source, according to current accounting practices, even though such management may actually increase long-term carbon storage. Examining four of the largest wildfires in the US in 2002, we found that, for forest land that experienced catastrophic stand-replacing fire, prior thinning would have reduced CO_2 release from live tree biomass by as much as 98%. Altering carbon accounting practices for forests that have historically experienced frequent, low-severity fire could provide an incentive for forest managers to reduce the risk of catastrophic fire and associated large carbon release events.

Front Ecol Environ 2008; 6(9): 493–498, doi:10.1890/070187

Over the past century, fire suppression has been the primary management tool in many fire-prone forests worldwide, and has increased tree density and forest fuels, resulting in an increased risk of stand-replacing wildfire (Veblen and Lorenz 1988; Covington *et al*. 1994; Fulé *et al*. 2004; Ohlson *et al*. 2006; Wang *et al*. 2007). A number of factors, including fire suppression, have probably contributed to increased carbon (C) storage in US forests during recent decades (Hurtt *et al*. 2002). Carbon in fire-suppressed forests is now more vulnerable to catastrophic release than is C in forests of pre-settlement condition (Breshears and Allen 2002).

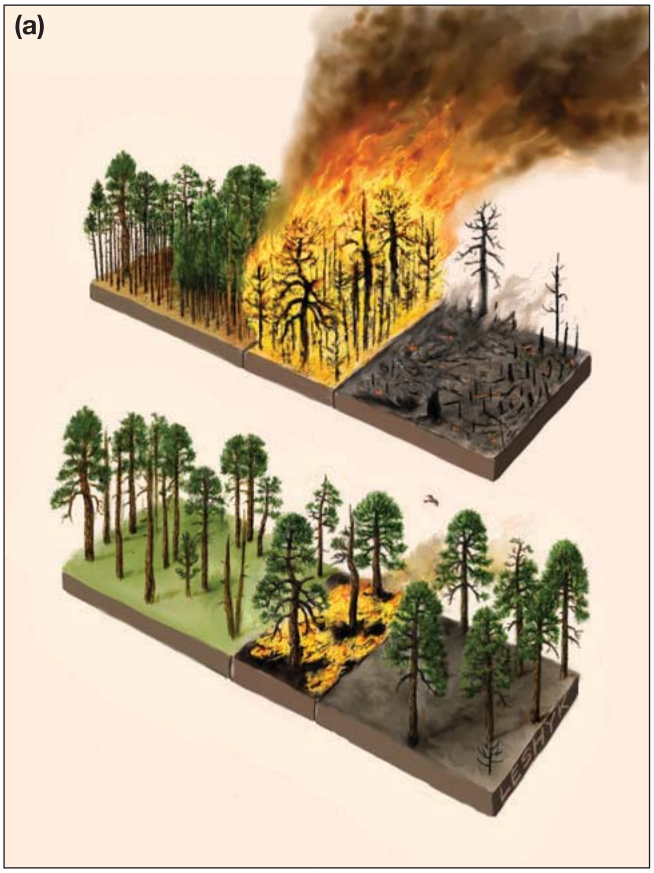
In forests of the western US, fire frequency and severity historically ranged from high-frequency, low-severity fires in ponderosa pine and Sierran mixed-conifer forests (Covington and Moore 1994; McKelvey *et al*. 1996) to low-frequency, high-severity fires in forests at higher elevations, such as spruce–fir and northern latitude coastal forests (Agee 1993; Schoennagel *et al*. 2004). The frequency with which large and severe wildfires have occurred has increased in recent decades, a pattern attributed to both land-use changes (Covington *et al*. 1994; McKelvey *et al*. 1996) and climatic shifts (Westerling *et al*. 2006). Wildfires release massive amounts of CO_2 to the atmosphere (van der Werf *et al*. 2006; Wiedinmyer and Neff 2007). Models indicate that even if current fire suppression success is maintained, the US carbon sink is predicted to decline through the 21st century, because harvesting removes carbon and mortality occurs at rates equal to sequestration resulting from tree regeneration (Hurtt *et al*. 2002). Increased frequency and intensity of wildfires will further contribute to this decline.

In a nutshell:

- In current carbon accounting, forest management impacts on potentially catastrophic disturbances are typically ignored
- In forests where fire suppression has caused fuel accumulation, forest fuel-reduction treatments can diminish the risk of stand-replacing fire, thereby promoting carbon storage
- Carbon accounting should recognize the value of management actions that reduce the risk of carbon loss through stand-replacing fire

Currently, forest managers are implementing fuel-reduction treatments in fire-prone forests historically characterized by high-frequency, low-severity fire regimes. In the Sierra Nevada Mountains of California and the southwestern US, these treatments typically involve removing small-diameter trees that have established since the advent of fire suppression policy (Covington *et al*. 1997; North *et al*. 2005). Figure 1 depicts the C storage consequences of two potential forest management strategies for dry western forests that have historically experienced high-frequency, low-severity fire events. While the unthinned option stores more C in the absence of fire, it is more likely to experience a stand-replacing fire that results in a large C release, both during the event and post-fire (Auclair and Carter 1993; Kashian *et al*. 2006). Moreover, depending on the forest type, the area burned by a stand-replacing fire does not recover its pre-fire C stock for decades (Schulze *et al*. 2000; Wirth *et al*. 2002). The thinned scenario effectively increases the "rotation length", placing forest car-

Department of Biological Sciences and Merriam–Powell Center for Environmental Research, Northern Arizona University, Flagstaff, AZ *(matthew.hurteau@nau.edu)



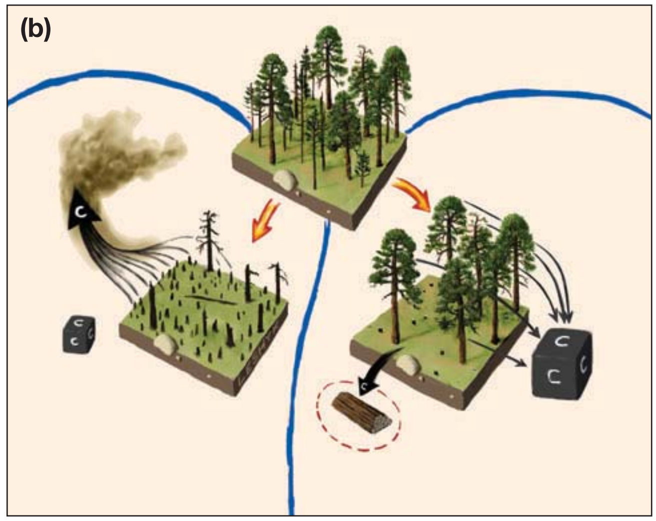


Figure 1. *(a) Two options for a given forest stand and the resultant tree survivorship following a wildfire event. (b) The carbon accounting consequences of two possible options for a given forest stand and the results following a wildfire event. The cubes represent the amount of carbon remaining in the ecosystem after wildfire.*

bon in a longer residence-time pool (the remaining trees continue to grow for a longer time period; Schulze *et al.* 2000). Additionally, as evidenced by wildfire simulations in Sierran mixed-conifer forest, thinning not only reduces the risk of a catastrophic C release, but also results in C being concentrated in fewer, larger trees that approximate the old-growth structure of pre-fire suppression forests (Hurteau and North 2009).

Carbon registry groups require that forest managers determine a baseline above which additional C stored counts as a carbon credit. This one-size-fits-all methodology is fundamentally flawed, because it does not fully account for the effect of variation in stand structure and forest biomass on the risk of stand-replacing fire and albedo (ie the ratio of outgoing to incoming radiation). Under the regulations of the Kyoto Protocol and groups such as the California Climate Action Registry, forest thinning is considered a carbon source to the atmosphere, because the amount of C stored on a given unit of land area is reduced, at least temporarily (Penman *et al.* 2003; California Climate Action Registry 2007), even though thinning reduces the risk of substantially greater C losses during stand-replacing wildfire. If, by contrast, that same forest is not thinned and instead experiences a catastrophic fire, the C stock baseline is simply *reduced*, as if no CO_2 emissions had occurred during the fire. This accounting is justified by the argument that management actions have little influence on the risk of fire, especially in forests that were historically typified by high-severity, stand-replacing fire. Even though CO_2 emissions do occur, they are considered to be uncontrollable and, thus, no land owner or manager's carbon account is debited. In many of the drier forest types of the western US, fires historically burned with high frequency and low severity (Covington *et al.* 1994). For example, southwestern ponderosa pine forests had a historic fire return interval that ranged from 2 to 12 years, and the mean fire return interval for Sierran mixed-conifer forests was 12 to 17 years (McKelvey *et al.* 1996). In these systems, fire maintained both the overstory structure and system processes (McKelvey *et al.* 1996). Here, the current C accounting methodology is faulty, because it penalizes actions that reduce avoidable carbon release to the atmosphere and that have been shown to restore ecosystem function in forests historically maintained by fire (Carey 2005; Zausen *et al.* 2005). Failure to recognize the threat of catastrophic C release amounts to a perverse incentive to increase fire risk through continued fire suppression.

An example from the 2002 fire season

During the 2002 fire season in the western US, four major fires – the Rodeo–Chediski, Hayman, Biscuit, and McNally – burned some 508 000 hectares of forest land, 92 000 hectares of which experienced high-severity, stand-replacing fire (Subirge and Lovely 2003; USDA Forest Service 2003; Azuma *et al.* 2004; Odion and Hanson 2006). To estimate pre-fire C storage for unmanaged forest, we used field data presented in Kuenzi (2006) for the 189 000-ha Rodeo–Chediski fire that occurred in Arizona in 2002. For the other three fires, we used stand characteristics from a dry, western forest comprised of two *Abies* species, two *Pinus* species, and *Pseudotsuga menziesii*, modeled by Agee and Skinner (2005). To calculate individual tree C storage, we utilized allometric models developed by Kaye *et al.* (2005) for *Pinus ponderosa*, which allowed us to use the modeled tree diameters from each fire event to calculate C storage by tree component. Using models derived from pine results in a more conservative estimate of C stocks compared to the fir and Douglas-fir allometric equations in the California Carbon Action Registry Forest Sector Protocol (2007). The per hectare C values were then multiplied by the number of hectares that experienced high-severity fire at each site.

To estimate hypothetical pre-fire C storage for the forests that had been thinned, we used stand characteristics from Agee and Skinner's (2005) "low-thin" treatment. This treatment is a thin-from-below harvest that most closely matches a forest restoration and wildfire risk reduction prescription; it involves removing a majority of the small-diameter trees (which accounted for 18% of the total biomass) and retaining all of the larger diameter trees.

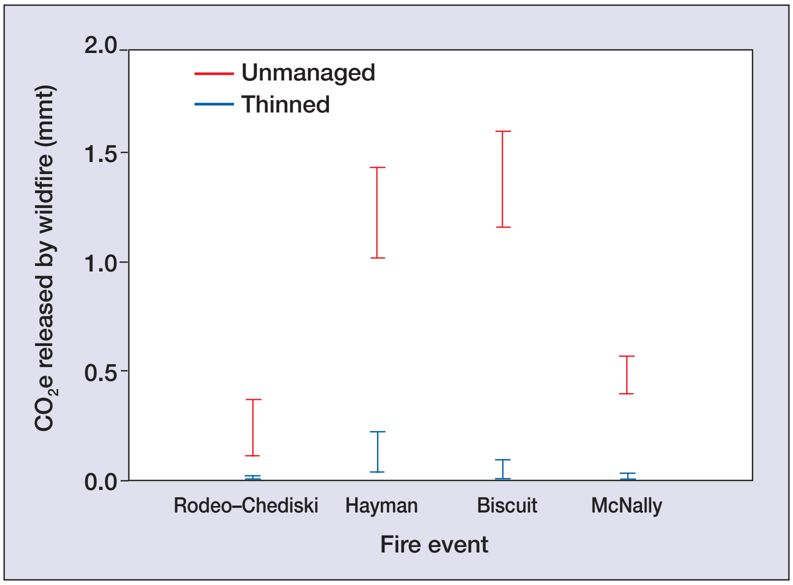


Figure 2. *Millions of metric tons of CO_2 equivalent released from wildfire. Red bars show the range of CO_2 released in each fire event in high-severity burn areas using the two different combustion efficiencies. Blue bars show the range of CO_2 that would have been released had the areas been thinned before wildfire, again assuming the two different combustion efficiencies.*

To estimate post-fire C storage for thinned forest, we utilized model outputs from Agee and Skinner (2005), which had an 8% tree mortality rate. To estimate post-fire C storage for unthinned forest, we utilized the Kaye *et al.* (2005) allometric models, coupled with two published combustion factors for emissions per unit of aboveground biomass, to bracket a range of possible emissions. Specifically, we used the average combustion factors for high-severity areas in the Biscuit fire (which burned greater than 200 000 ha in southern Oregon and northern California in 2002; Campbell *et al.* in press) and the 30% combustion efficiency used by Wiedinmyer *et al.* (2006) for woody fuels. Our calculated emissions values are quite conservative, since we account only for emissions from live trees. As noted by Campbell *et al.* (in press), 57% of the Biscuit fire emissions were the result of combustion of the litter and duff layers. Our analysis did not account for the energy costs of thinning and transporting biomass and their associated CO_2 emissions.

Of the approximately 92 000 ha that experienced high-severity, stand-replacing fire in these four wildfires, approximately 4.2–6.1 million metric tons of CO_2 equivalent ($MMTCO_2e$) were released from live tree biomass (Figure 2). Thinning these same forests before they burned would have removed 3.9 $MMTCO_2e$ and reduced live tree fire emissions to only 0.07–0.3 $MMTCO_2e$.

Both strategies – forest thinning and fire suppression – cost money. In the case of thinning to reduce fuels, small-diameter trees are typically non-merchantable and require that the manager pay for their removal. The Healthy Forest Restoration Act (2003), the legislation driving much of the recent fuel-reduction effort, allows for project funding through operations revenue. This approach could diminish forest C storage by requiring the removal and sale of large trees to offset the cost of removing the smaller ones.

Fire suppression and the inevitable wildfires that follow are also expensive. The direct suppression costs for fighting the four fires we assessed were approximately \$277 million (Graham 2003; Snider *et al.* 2003; The Wilderness Society 2003; Azuma *et al.* 2004). This figure includes only expenditures for personnel and equipment actually employed in fighting the fires, and does not include other costs, such as property loss, land rehabilitation, and reforestation. A conservative estimate of these other costs for the Rodeo–Chediski fire is \$250 million (Snider *et al.* 2003). This puts the cost of that fire at \$1586 ha^{-1}. If we assume similar direct and indirect costs for the other fires, the high-severity areas of these four fire events cost approximately \$145.8 million. An analysis of forest thinning costs for nine national forests in the western US found that treatments can range from \$344–1097 ha^{-1} (Fight and Barbour 2006). For the four fires considered here, this would have totaled \$100.9 million (using \$1097 ha^{-1}) and would have resulted in 16.5 million metric tons of CO_2e sequestered on the site in live tree biomass, had these areas been thinned prior to the fire event. This sequestered CO_2 has a current market value of \$8.5 million (based on \$1.90 per ton of carbon; Chicago Climate Exchange, November 30, 2007). The revenue generated from the sale of offsets could potentially be used by forest managers to pay for wildfire risk reduction operations (see Figure 3 for costs by event). Allowing carbon credits for protecting forest carbon would provide revenue that could be used in lieu of operations revenue, thereby reducing reliance on larger diameter trees to pay for fuel reduction operations under the Healthy Forest Restoration Act.

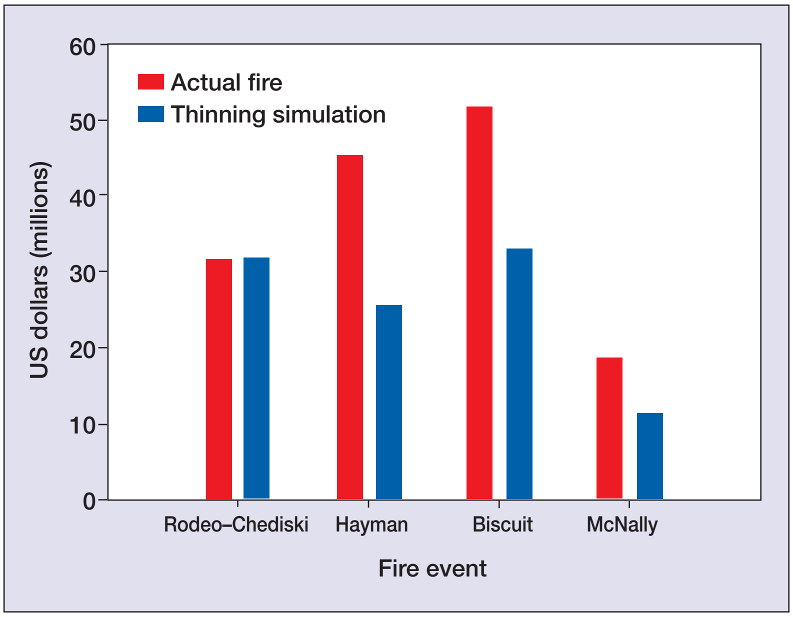


Figure 3. *Total estimated cost of each fire event plus the cost of offsetting the CO_2 release (red) and total cost of thinning the same land area minus the market value of the offsets gained from protecting the carbon stock (blue).*

Conclusions

Our "back-of-the-envelope" calculations indicate that massive CO_2 emissions from wildfire are avoidable in forests that have historically been characterized by frequent, low-severity fire (Figure 2). Forests thinned to approach pre-settlement tree density and stand structure harbor substantially more C after wildfire than adjacent dense stands that have not been thinned (Wirth *et al.* 2002; Figures 4 and 5). Moreover, the biomass removed by thinning is available for wood products or energy generation, the latter replacing fossil-fuel emissions (Pacala *et al.* 2001). Thinning forests for C protection also achieves many of the ecological goals of forest restoration (Covington 2000). One of the ancillary benefits of thinning these forests is a reduction in resource competition that increases the growth of the remaining trees (Sheriff 1996). This increase in growth rates could potentially offset part of the predicted decline in the US carbon sink, while concurrently storing carbon in fewer, larger trees per hectare, thereby reducing the risk of loss to catastrophic fire.

Under the Kyoto Protocol (for unmanaged lands) and the California Climate Action Registry carbon accounting policies, emissions of CO_2e from stand-replacing fires are ignored, and the C stock baseline is simply recalculated based on the new carbon stock level (Penman *et al.* 2003; California Climate Action Registry 2007). In our analysis, the carbon protection leverage of forest thinning is strong. Assuming a 30% combustion efficiency, removing 3.9 $MMTCO_2$e of the original forest C by thinning prevents the direct loss of 5.7 $MMTCO_2$e of the carbon by catastrophic wildfire, if the same wildfire burns over the thinned area after treatment. Thus, even if all of the thinned forest biomass is burned, there is still a net benefit of protecting 1.8 $MMTCO_2$e (5.7 $MMTCO_2$e–3.9 $MMTCO_2$e) from release during wildfire. If the thinned forest biomass is used for wood products or energy generation, the benefits are correspondingly greater.

Stand-replacing fires are a natural part of the disturbance regime for some forest types, and resetting the carbon baseline after fire may be appropriate in these situations. However, our analysis calls into question the application of current carbon accounting practices in systems historically characterized by a low-severity and high-frequency fire regime. Carbon accounting guidelines should allow us to maintain the disturbance regimes that shaped the diverse forest types in the western US, particularly in cases where the natural disturbance regimes also favor long-term carbon storage.

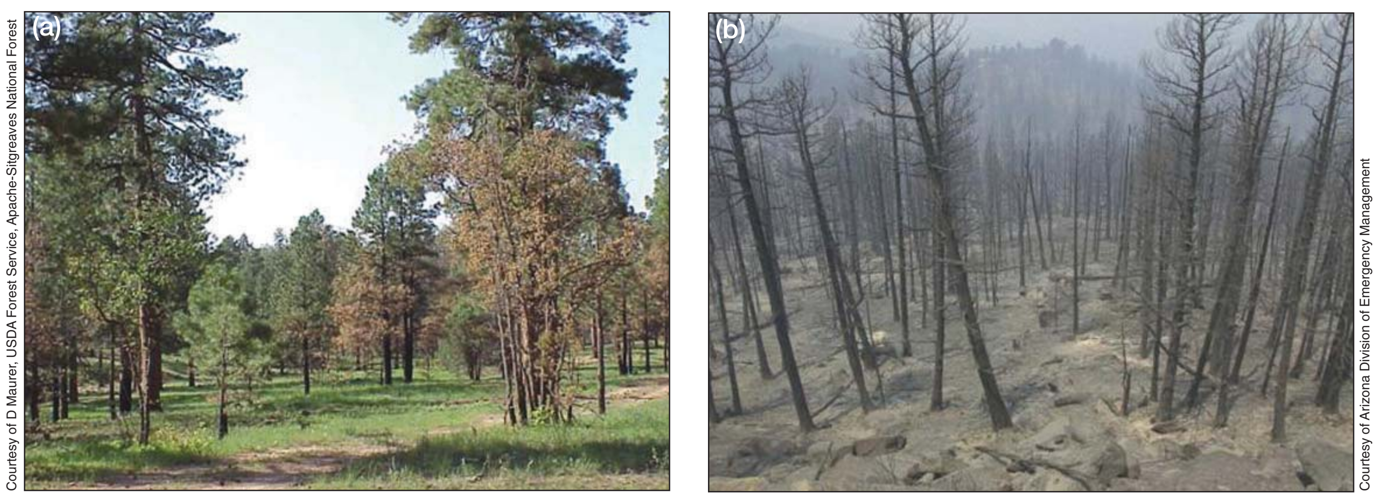


Figure 4. *Post-fire forest conditions following the Rodeo–Chediski fire. (a) Photo taken approximately one year after the fire in an area that had been thinned (to a basal area of about 15 m^2 ha^{-1}) and pile burned in 1997. (b) Photo taken immediately post-fire, in an area that was not thinned prior to the fire.*

Carbon accounting guidelines may also need to be revised to consider more broadly the feedbacks from forest management and disturbance to the climate system (eg the influence of forest treatments on albedo). In addition to promoting C storage, forest thinning probably contributes a net cooling effect by increasing surface reflectance: removing small trees reduces leaf area index (McDowell *et al.* 2007), exposing the grass, soil, or snow below (depending on season and latitude), each of which has a higher albedo than coniferous trees (Liu *et al.* 2005). On a regional scale, increased albedo could cause net cooling (Liu *et al.* 2005; Randerson *et al.* 2006; Figure 6), creating another possible benefit of fuel-reduction treatments. Evaluating this hypothesis will require measuring changes in albedo with fuel-reduction treatments, as well as regional scaling of the consequences for climate. It seems likely that, in some situations, fuel-reduction treatments would offer climate benefits, both by increasing C storage and by increasing surface reflectance.

Current carbon accounting practices for forest systems that are characterized by frequent, low-severity fire ignore the influence of management actions on fire risk. Instead of being punished for reducing C stocks, forest managers who take action to reduce the risk of catastrophic wildfire in systems that historically experienced frequent, low-severity fire could be rewarded with carbon credits. Allowing credits for wildfire risk reduction would provide another means of generating revenue to cover the cost of thinning forests, especially in regions such as the Southwest, where a majority of the wood biomass that needs to be removed is not currently merchantable.

Acknowledgements

The authors thank V Leshyk from the Bilby Research Center at Northern Arizona University for illustrations. This research was supported by the US Department of Energy's Office of Science (BER) through the Western Regional Center of the National Institute for Climatic Change Research at Northern Arizona University.

M Jablonski, USDA Forest Service, Lassen National Forest

Figure 5. *Stand conditions following the 2002 Cone fire at the Blacks Mountain Experimental Forest. The white line approximates the border between the treated and untreated areas prior to the wildfire. The area in the upper left was left untreated and the remaining area was thinned and prescribe-burned prior to the Cone fire.*

References

Agee JK. 1993. Fire ecology of Pacific Northwest forests. Washington, DC: Island Press.

Agee JK and Skinner CN. 2005. Basic principles of forest fuel reduction treatments. *Forest Ecol Manag* **211**: 83–96.

Auclair AND and Carter TB. 1993. Forest wildfires as a recent source of CO_2 at northern latitudes. *Can J Forest Res* **23**: 1528–36.

Azuma DL, Donnegan J, and Gedney D. 2004. Southwest Oregon Biscuit fire: an analysis of forest resources and fire severity. Portland, OR: USDA Forest Service. Res Paper PNW-RP-560.

Breshears DD and Allen CD. 2002. The importance of rapid, disturbance-induced losses in carbon management and sequestration. *Global Ecol Biogeogr* **11**: 1–5.

California Climate Action Registry. 2007. Forest Sector Protocol. www.climateregistry.org/PROTOCOLS. Viewed 5 Mar 2008.

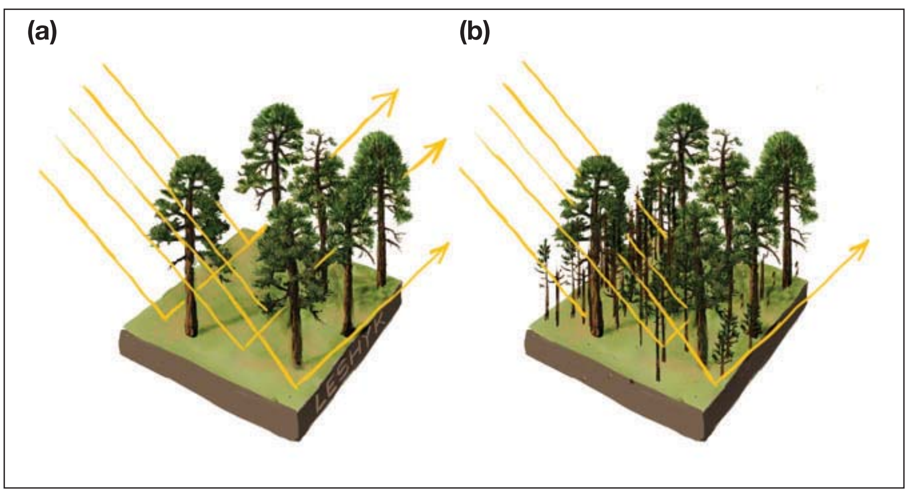


Figure 6. *Albedo of forest land under (a) thinned and (b) unmanaged conditions. Thinning increases albedo by revealing more of the "brighter" grass, soil surface, or snow.*

Campbell J, Donato D, Azuma D, and Law B. Pyrogenic carbon emissions from a large wildfire in Oregon, USA. *J Geophys Res Biogeosci*. In press.

Carey AB. 2005. Active intentional management for biodiversity and other forest values. In: Peterson CE and Maguire DA (Eds). Balancing ecosystem values: innovative experiments for sustainable forestry. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. General Technical Report, PNW-GTR-635.

Covington WW and Moore MM. 1994. Southwestern ponderosa pine forest structure: changes since Euro-American settlement. *J Forest* **92**: 39–47.

Covington WW, Everett RL, Steele R, *et al.* 1994. Historical and anticipated changes in forest ecosystems of the inland west of the United States. *J Sustain Forest* **2**: 13–63.

Covington WW, Fule PZ, Moore MM, *et al.* 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. *J Forest* **95**: 23–29.

Covington WW. 2000. Helping western forests heal. *Nature* **408**: 135–36.

Fight RD and Barbour RJ. 2006. Financial analysis of fuel treatments on national forests in the western United States. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. Research Note, PNW-RN-555.

Fule PZ, Crouse JE, Cocke AE, *et al.* 2004. Changes in canopy fuels and potential fire behavior 1880–2040. *Ecol Model* **175**: 231–48.

Graham RT (Ed). 2003. Hayman fire case study. Ogden, UT: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. General Technical Report, RMRS-GTR-114.

Healthy Forest Restoration Act. 2003. http://frwebgate.access.gpo.gov/cgibin/getdoc.cgi?dbname=108_cong_bills&docid=f:h1904enr.txt.pdf. Viewed 6 Feb 2008.

Hurteau M and North M. 2009. Fuel treatment effects on tree-based carbon storage and emissions under modeled wildfire scenarios. *Front Ecol Environ* **7**. doi:10.1890/080049.

Hurtt GC, Pacala SW, Moorcroft PR, *et al.* 2002. Projecting the future of the US carbon sink. *P Natl Acad Sci USA* **99**: 1389–94.

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

Kaye JP, Hart SC, Fule PZ, *et al.* 2005. Initial carbon, nitrogen, and phosphorous fluxes following ponderosa pine restoration treatments. *Ecol Appl* **15**: 1581–93.

Kuenzi AM. 2006. Pre-fire treatment effects and understory plant community response on the Rodeo–Chediski fire, Arizona (MS thesis). Flagstaff, AZ: Northern Arizona University.

Liu H, Randerson JT, Lindfors J, and Chapin III FS. 2005. Changes in the surface energy budget after fire in boreal ecosystems of interior Alaska: an annual perspective. *J Geophys Res* **110**: D13101.

McDowell NG, Adams HD, Bailey JD, and Kolb TE. 2007. The role of stand density on growth efficiency, leaf area index, and resin flow in southwestern ponderosa pine forests. *Can J Forest Res* **37**: 343–55.

McKelvey KS, Skinner CN, Chang C, *et al.* 1996. An overview of fire in the Sierra Nevada. In: Sierra Nevada ecosystem project: final report to Congress. Vol II. Assessments and scientific basis for management options. Davis, CA: Centers for Water and Wildland Resources, University of California, Davis.

North M, Hurteau M, Fiegener R, and Barbour M. 2005. Influence of fire and El Niño on tree recruitment varies by species in Sierran mixed conifer. *Forest Sci* **51**: 187–97.

Odion DC and Hanson CT. 2006. Fire severity in conifer forests of the Sierra Nevada, California. *Ecosystems* **9**: 1177–89.

Ohlson DW, Berry TM, Gray RW, *et al.* 2006. Multi-attribute evaluation of landscape-level fuel management to reduce wildfire risk. *Forest Policy Econ* **8**: 824–37.

Pacala SW, Hurtt GC, Baker D, *et al.* 2001. Consistent land- and atmosphere-based US carbon sink estimates. *Science* **292**: 2316–20.

Penman J, Gytarsky M, Hiraishi T, *et al.* (Eds). 2003. Good practice guidance for land use, land-use change and forestry. Kanagawa, Japan: Intergovernmental Panel on Climate Change. www.ipcc-nggip.iges.or.jp/public/gpglulucf/gpglulucf_contents.htm. Viewed 6 Feb 2008.

Randerson JT, Liu H, Flanner MG, *et al.* 2006. The impact of boreal forest fire on climate warming. *Science* **314**: 1130–32.

Schoennagel T, Veblen TT, and Romme WH. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. *BioScience* **54**: 661–76.

Schulze E-D, Wirth C, and Heimann M. 2000. Managing forests after Kyoto. *Science* **289**: 2058–59.

Sheriff DW. Responses of carbon gain and growth of *Pinus radiata* stands to thinning and fertilizing. *Tree Physiol* **16**: 527–36.

Snider GB, Wood DB, and Daugherty PJ. 2003. Analysis of costs and benefits of restoration-based hazardous fuel reduction: treatments vs no treatment. Flagstaff, AZ: Northern Arizona University School of Forestry. Progress Report 1.

Subirge TG and Lovely C. 2003. Environmental impact statement for Rodeo–Chediski fire timber salvage sale area. Springerville, AZ: US Department of Agriculture, Forest Service, Apache-Sigreaves and Tonto National Forests.

The Wilderness Society. 2003. Summary of the McNally fire, California: July 21–August 29, 2002. Washington, DC: Ecology and Economics Research Department.

USDA (US Department of Agriculture) Forest Service. 2003. Environmental assessment: Hayman fire timber salvage sale. Colorado Springs, CO: USDA Forest Service.

van der Werf GR, Randerson JT, Giglio L, *et al.* 2006. Interannual variability in global biomass burning emissions from 1997 to 2004. *Atmos Chem Phys* **6**: 3423–41.

Veblen T and Lorenz D. 1988. Recent vegetation changes along the forest/steppe ecotone of northern Patagonia. *Ann Assoc Am Geogr* **78**: 93–111.

Wang X, He HS, and Li X. 2007. The long-term effects of fire suppression and reforestation on a forest landscape in northeastern China after a catastrophic wildfire. *Landscape Urban Plan* **79**: 84–95.

Westerling AL, Hidalgo HG, and Swetnam TW. 2006. Warming and earlier spring increase western US forest wildfire activity. *Science* **313**: 940–43.

Wiedinmyer C and Neff JC. 2007. Estimates of CO_2 from fires in the United States: implications for carbon management. *Carbon Bal Manage* **2**: 10.

Wiedinmyer C, Quayle B, Geron C, *et al.* 2006. Estimating emissions from fires in North America for air quality modeling. *Atmos Environ* **40**: 3419–32.

Wirth C, Czimczik CI, and Schulze E-D. 2002. Beyond annual budgets: carbon flux at different temporal scales in fire-prone Siberian Scots pine forests. *Tellus* **54B**: 611–30.

Zausen GL, Kolb TE, Bailey JD, and Wagner MR. 2005. Long-term impacts of stand management on ponderosa pine physiology and bark beetle abundance in northern Arizona: a replicated landscape study. *Forest Ecol Manage* **218**: 291–305.

