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Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions?

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Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions?

John L Campbell[1*], Mark E Harmon[1], and Stephen R Mitchell[2]

It has been suggested that thinning trees and other fuel-reduction practices aimed at reducing the probability of high-severity forest fire are consistent with efforts to keep carbon (C) sequestered in terrestrial pools, and that such practices should therefore be rewarded rather than penalized in C-accounting schemes. By evaluating how fuel treatments, wildfire, and their interactions affect forest C stocks across a wide range of spatial and temporal scales, we conclude that this is extremely unlikely. Our review reveals high C losses associated with fuel treatment, only modest differences in the combustive losses associated with high-severity fire and the low-severity fire that fuel treatment is meant to encourage, and a low likelihood that treated forests will be exposed to fire. Although fuel-reduction treatments may be necessary to restore historical functionality to fire-suppressed ecosystems, we found little credible evidence that such efforts have the added benefit of increasing terrestrial C stocks.

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Various levels of tree removal, often paired with prescribed burning, are a management tool commonly used in fire-prone forests to reduce fuel quantity, fuel continuity, and the associated risk of high-severity forest fire. Collectively referred to as "fuel-reduction treatments", such practices are increasingly used across semiarid forests of the western US, where a century of fire suppression has allowed fuels to accumulate to levels deemed unacceptably hazardous. The efficacy of fuel-reduction treatments in temporarily reducing fire hazard in forests is generally accepted (Agee and Skinner 2005; Ager *et al.* 2007; Stephens *et al.* 2009a) and, depending on the prescription, may serve additional management objectives, including the restoration of native species composition, protection from insect and pathogen outbreaks, and provision of wood products and associated employment opportunities.

In a nutshell:

- Carbon (C) losses incurred with fuel removal generally exceed what is protected from combustion should the treated area burn
- Even among fire-prone forests, one must treat about ten locations to influence future fire behavior in a single location
- Over multiple fire cycles, forests that burn less often store more C than forests that burn more often
- Only when treatments change the equilibrium between growth and mortality can they alter long-term C storage

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Recently, several authors have suggested that fuel-reduction treatments are also consistent with efforts to sequester C in forest biomass, thus reducing atmospheric carbon dioxide (CO_2) levels (Frinkral and Evans 2008; Hurteau *et al.* 2008; Hurteau and North 2009; Stephens *et al.* 2009b). It is argued that short-term losses in forest biomass associated with fuel-reduction treatments are more than made up for by the reduction of future wildfire emissions, and thinning practices aimed at reducing the probability of high-severity fire should therefore be given incentives rather than be penalized in C-accounting programs. This is an appealing notion that aligns the practice of forest thinning with four of the most pressing environmental and societal concerns facing forest managers in this region today – namely, fire hazard, economic stimulus, so-called forest health, and climate-change mitigation. However, we believe that current claims that fuel-reduction treatments function to increase forest C sequestration are based on specific and sometimes unrealistic assumptions regarding treatment efficacy, wildfire emissions, and wildfire burn probability.

In this paper, we combine empirical data from various fire-prone, semiarid conifer forests of the western US (where issues of wildfire and fuel management are most relevant) with basic principles of forest growth, mortality, decomposition, and combustion. Our goal is to provide a complete picture of how fuel treatments and wildfires affect aboveground forest C stocks by examining these disturbance events (1) for a single forest patch, (2) across an entire forest landscape, (3) after a single disturbance, and (4) over multiple disturbances. Finally, we consider how wildfire and/or fuel treatments could initiate alternate equilibrium states

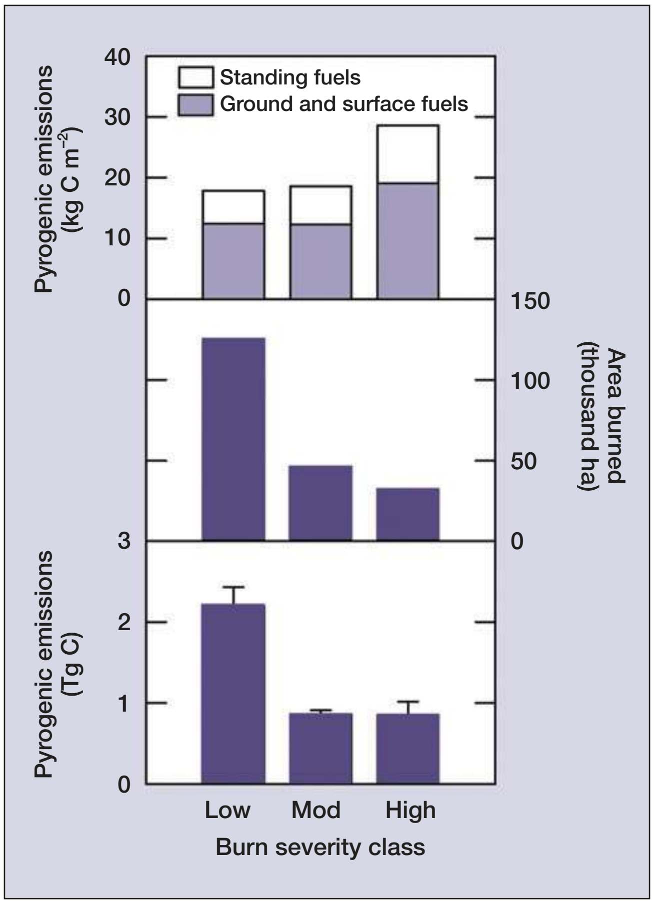

Figure 1. *Sources of pyrogenic emissions across the 2002 Biscuit Fire in southwestern Oregon and northern California. Because most emissions arise from the combustion of ground and surface fuels, pyrogenic emissions from high-severity fires were only one-third higher than those in low-severity fires. Moreover, because most of the fire burned with low severity, the contribution of high-severity fire to total emissions was only about 20%. The Biscuit Fire burned over a mosaic of young, mature, and old-growth stands of mixed conifer growing across a climate gradient ranging from mesic to semiarid. Methods are described in Campbell* et al. *(2007).*

and change the long-term capacity of a forest to accumulate biomass.

Immediate stand-level C losses attributed to wildfire and fuel-reduction treatments

Because fuel-reduction treatments are generally designed to reduce subsequent wildfire severity, rather than to preclude fire entirely, it is important to compare the C losses incurred under both high- and low-severity fire scenarios. The amount of biomass combusted in a high-severity crown fire is unquestionably greater than the amount combusted in a low-severity surface fire. The difference, however, is smaller than that suggested by some authors (eg Hurteau *et al.* 2008). Even under the most extreme fuel-moisture conditions, the water content of live wood frequently prohibits combustion beyond surface char; this is evident in the retention of even the smallest canopy branches after high-severity burns (Campbell *et al.* 2007). Moreover, the consumption of fine surface fuels (ie leaf litter, fallen branches, and understory vegetation), though variable, can be high even in low-severity burns. As shown in Figure 1, Campbell *et al.* (2007) found that patches of mature mixed-conifer forest in southwestern Oregon that were subject to low-severity fire (ie 0–10% overstory mortality) released 70% as much C per unit area as did locations experiencing high-severity fire (ie > 80% overstory mortality). When scaled over an entire wildfire perimeter, the importance of high-severity fire in driving pyrogenic emissions is further diminished because crown fires are generally patchy while surface fires are nearly ubiquitous (Meigs *et al.* 2009).

According to Campbell *et al.* (2007), less than 20% of the estimated 3.8 teragrams of C released to the atmosphere by the 2002 Biscuit Fire in the Siskiyou National Forest of southern Oregon and northern California (Figure 1) arose from overstory combustion. Simply put, because most pyrogenic emissions arise from the combustion of surface fuels, and most of the area within a typical wildfire experiences surface-fuel combustion, efforts to minimize overstory fire mortality and subsequent necromass decay are limited in their ability to reduce fire-wide pyrogenic emissions.

The total amount of biomass combusted, or taken offsite, during a fuel treatment is, by definition, a prescribed quantity and can vary widely depending on the specific management objective and techniques used. A review of fuel-reduction treatments carried out in semiarid conifer forests in the western US reveals that aboveground C losses associated with treatment averaged approximately 10%, 30%, and 50% for prescribed fire only, thinning only, and thinning followed by prescribed fire, respectively (WebTable 1). By comparison, wildfires burning over comparable fire-suppressed forests consume an average 12–22% of the aboveground C (total fire-wide averages reported by Campbell *et al.* [2007] and Meigs *et al.* [2009], respectively).

Given that both fuel-reduction treatments and wildfire remove C from a forest, to what degree does the former reduce the impact of the latter? To test this question, Mitchell *et al.* (2009) simulated wildfire combustion following a wide range of fuel-reduction treatments for three climatically distinct conifer forest types in Oregon. As illustrated in Figure 2, fuel treatments were effective in reducing combustion in a subsequent wildfire, and the greater the treatment intensity, the greater the reduction in future combustion. However, even in the mature, fire-suppressed ponderosa pine (*Pinus ponderosa*) forest, protecting one unit of C from wildfire combustion typically came at the cost of removing three units of C in treatment. The reason for this is simple: the efficacy of fuel-reduction treatments in reducing future wildfire emissions comes in large part by removing or combusting surface fuels ahead of time. Furthermore, because removing fine canopy fuels (ie leaves and twigs) practically necessitates removing the branches and boles to which they are attached, conventional fuel-reduction treatments usually remove more C from a forest stand than would a wildfire burning in an untreated stand. In an extreme modeling scenario, wherein only fine-surface fuels were removed, subsequent avoided combustion did slightly exceed treatment removals (Figure 2, circles). However, this marginal gain amounted to less than 0.03% of the total C stores, which is, practically speaking, a zero-sum game.

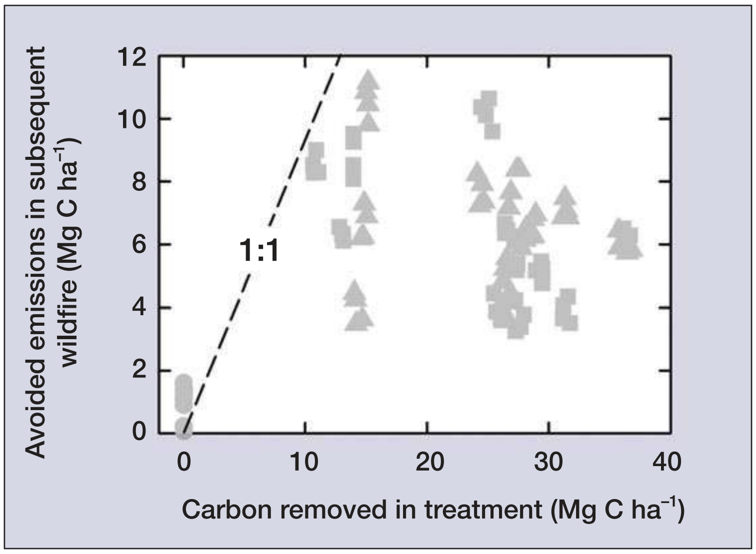

Figure 2. *Simulated effectiveness of various fuel-reduction treatments in reducing future wildfire combustion in a ponderosa pine (*Pinus ponderosa*) forest. In general, protecting one unit of C from wildfire combustion came at the cost of removing approximately three units of C in treatment. At the very lowest treatment levels, more C was protected from combustion than removed in treatment; however, the absolute gains were extremely low. Circles show understory removal, squares show prescribed fire, and triangles show understory removal and prescribed fire. Simulations were run for 800 years with a treatment-return interval of 10 years and a mean fire-return interval of 16 years. Forest structure and growth were modeled to represent mature, semiarid ponderosa pine forest growing in Deschutes, Oregon. Further descriptions of these simulations are given in Mitchell* et al. *(2009).*

Wildfire probability, treatment life span, and treatment efficacy across a landscape

Any approach to C accounting that assumes a wildfire burn probability of 100% during the effective life span of a fuel-reduction treatment is almost certain to overestimate the ability of such treatments to reduce pyrogenic emissions on the future landscape. Inevitably, some fraction of the land area from which biomass is thinned will not be exposed to any fire during the treatment's effective life span and therefore will incur no benefits of reduced combustion (Rhodes and Baker 2008). On the other hand, assuming that landscape-wide burn probabilities apply to all of the treated area is almost certain to underestimate the influence of treatment on future landscape combustion. This is because doing so does not account for managers' ability to target treatments toward probable ignition sources or the capacity of treated areas to reduce burn probability in adjacent untreated areas (Ager *et al.* 2010).

Among fire-prone forests of the western US, the combination of wildfire starts and suppression efforts result in current burn probabilities of less than 1% (WebTable 2). Given a fuel-treatment life expectancy of 10–25 years, only 1–20% of treated areas will ever have the opportunity to affect fire behavior. Such approximations are consistent with a similar analysis reported by Rhodes and Baker (2008), who suggested that only 3% of the area treated for fuels is likely to be exposed to fire during their assumed effective life span of 20 years. Extending treatment efficacy by repeated burning of understory fuels could considerably increase the likelihood of a treated stand to affect wildfire behavior, but such efforts come at the cost of more frequent C loss.

A more robust, though more complicated, evaluation of fuel-treatment effect on landscape burn probability is achieved through large-scale, spatially explicit fire spread simulations (Miller 2003; Syphard *et al.* 2011). In one such simulation, representing both the topography and distribution of fuels across a fire-prone and fire-suppressed landscape in western Montana, Finney *et al.* (2007) showed how strategically treating as little as 1% of the forest annually for 20 years reduced the area impacted by a single large wildfire (expected to occur about once on this landscape in that 20-year period) by half, and how strategically treating 4% of the forest annually reduced the area impacted by a single large wildfire by > 95% (Figure 3). However, even when the treatment effect was highest, the protection of each hectare of forest from fire came at the cost of treating nearly 10 hectares (note the axis scales in Figure 3). Such inefficiencies come not from the treatments' efficacy in curtailing fire spread; rather, they stem from the rarity of wildfire. Put another way, the treatment of even modest areas may lead to high fractional reductions in the area impacted by high-severity wildfire, but because such fires rarely affect much of the landscape, the absolute change in area burned is small.

Carbon dynamics through an entire disturbance cycle

Although there is a body of literature that separately quantifies the decomposition of standing dead trees, dead tree fall rate, and the decomposition of downed woody debris, there are surprisingly few empirical studies that integrate these processes to estimate the overall longevity of fire-killed trees. Combining disparate estimates of standing and downed wood decay with tree-fall rates sug-

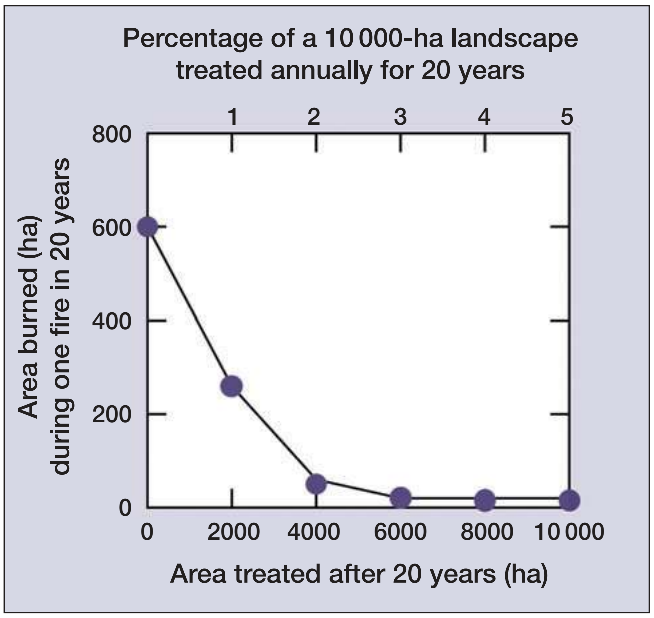

***Figure 3.** Simulated effects of strategically placed fuel treatments on wildfire spread across a fire-prone ponderosa and lodgepole pine* (Pinus contorta) *landscape in western Montana. Treating only 1% of the forest annually for 20 years reduced the area impacted by a single large wildfire (assumed to occur about once in 20 years) by more than half. However, across this entire treatment response, the protection of one hectare of forest from fire required the treatment of about 10 hectares. Adapted from Finney* et al. *(2007).*

gests that the overall rate at which fire-killed trees decompose in a semiarid conifer forest likely ranges between 1–9% annually (ie a half-life of 8–70 years). These values are consistent with the observations of Donato (unpublished data), who found that 52% of the biomass killed in a forest-replacing wildfire in southwestern Oregon was still present after 18 years.

It is reasonable to expect that in the first decade or two after a forest-replacing fire, the decomposition of fire-killed trees may exceed the net primary production (NPP) of re-establishing vegetation, thus driving net ecosystem production (NEP) below zero. This expectation is supported by eddy covariance flux measurements (Dore *et al.* 2008) and other empirical studies of post-fire vegetation (Irvine *et al.* 2007; Meigs *et al.* 2009). However, despite a protracted period of negative NEP following a fire event, total C stocks integrated over the entire disturbance cycle may be similar for a forest subject to a fuel-reduction treatment and one subject to a stand-replacing fire. This can easily be shown with a simple C model that simulates growth, mortality, decomposition, and combustion for ponderosa pine forests (Figure 4). How can this be? Simply put, biomass recovery may be slower in the wildfire scenario than in the fuel-reduction scenario, but initial biomass losses may be greater in the fuel-reduction scenario than in the wildfire scenario. Although the parameters used to generate Figure 4 (ie 30% live basal-area removal in the treatment scenario, 100% tree mortality in the wildfire scenario, and rapid post-fire regeneration) are reasonable, real-world responses may not exhibit such parity in integrated C stocks between disturbance types. The point of this simulation is to demonstrate how marked differences in post-disturbance NEP do not necessarily translate into differences in C stocks integrated over time. The quantification of NEP over short intervals is extremely valuable in teasing apart ecosystem C dynamics; however,

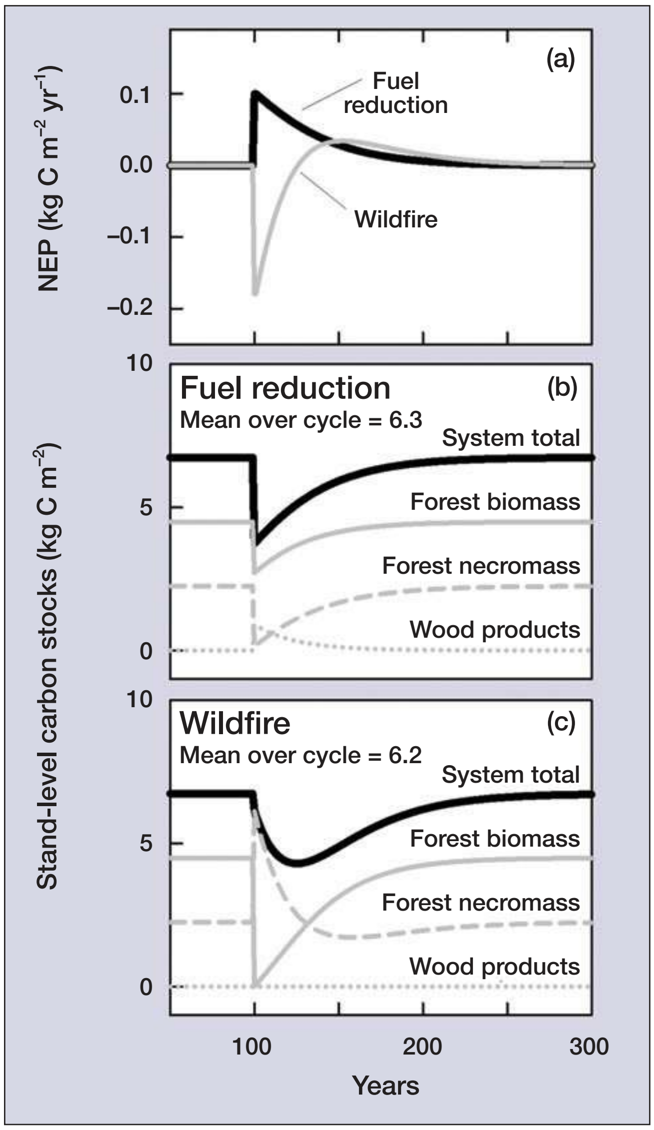

***Figure 4.** (a) Simulated net ecosystem production and (b–c) C stocks throughout an entire disturbance interval, initiated by either wildfire or fuel-reduction treatment. Unlike the stand subject to fuel reduction via thinning, the combination of low biomass and high necromass after wildfire functions to drive NEP below zero. Nevertheless, although initial losses associated with wildfire were much lower than those in the fuel-reduction treatment, the two scenarios achieved parity in C stocks over the entire disturbance interval. The model used to generate these simulations was parameterized for a ponderosa pine forest representative of the eastern Cascades and is fully described in WebFigure 1.*

simply comparing C flux rates immediately following different disturbances can give a misleading picture of how disturbances dictate long-term C balance.

■ Fire frequency and C stocks over multiple disturbance cycles

The C stocks of an ecosystem in a steady state are inversely proportional to the rate constants related to losses, such as those that occur through respiration or combustion (Olson 1963). Whereas Olson (1963) considered ecosystems in steady state, the same phenomenon occurs for the average ecosystem stocks over time or over broad areas (Smithwick *et al.* 2007). As fire frequency increases, the absolute and relative amount of C combusted per individual fire decreases, suggesting that as fire frequency increases, so too will average C stocks. However, using a model that simulates forest growth, mortality, decomposition, and fuel-dependent combustion, researchers can show that a low-frequency, high-severity fire regime stores substantially more C over time than a high-frequency, low-severity fire regime (mean C stocks increased by 40% as the mean fire-return interval was increased from 10 to 250 years; Figure 5). The reason for this is explained by the first principles outlined by Olson (1963). Fractional combustion is, by nature, more constrained than fire frequency. In our example, although fire interval increased from 10 years to 250 years, fractional combustion of ecosystem C for a semiarid ponderosa pine forest only increased from 9% to 18% (Figure 5). To have parity in C stocks across these different fire intervals, fractional combustion per event would, at times, have to exceed 250% – clearly violating the conservation of mass. As long as wildfire does not cause lasting changes in site productivity or non-fire mortality, no forest system is exempt from this negative relationship between fire frequency and average landscape C storage. Although we chose to illustrate the response for a semiarid ponderosa pine forest typical of those considered for fuel reduction, the same relative response was observed when the simulations were run for mesic Douglas fir (*Pseudotsuga menziesii*) forests parameterized for higher production and decomposition rates.

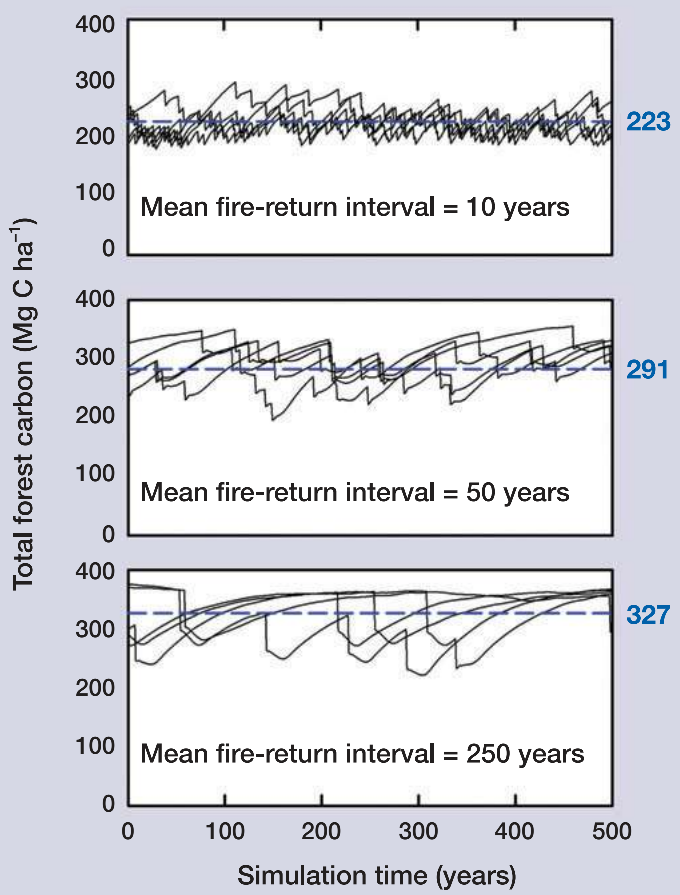

Figure 5. *Total forest C stores simulated for a ponderosa pine forest in the eastern Cascades of Oregon experiencing three different hypothetical fire regimes. Black lines depict the C stores of five individual stands subject to random fire events. Blue lines mark the 500-year average of all five stands. As mean fire-return interval increases, the variation of C stores over time (or space by extension) increases, but so does the long-term average. For simplicity, we show the results of only five stands per fire regime; however, the mean trends do not change with additional simulations. Nearly identical patterns result when alternate forest types are used. We performed simulations using STANDCARB, as described in WebFigure 2 and in Harmon* et al. *(2009).*

Although stability of C stocks is desirable, stability is a function of spatial extent. In the case of a single forest stand, C stocks under the frequent, low-severity fire regime are more stable than those under an infrequent, high-severity fire regime. However, the fluctuations in C stocks exhibited by a single stand become less relevant as one scales over time or over populations of stands experiencing asynchronous fire events (Smithwick *et al.* 2007). In other words, forests experiencing frequent fires lose less C per fire event than forests experiencing infrequent fires, but the former do not store more C over time or across landscapes.

■ The capacity of fire and fuel-reduction treatments to alter equilibrium states

In the sections above, we have assumed that forests eventually succeed toward a site-specific dynamic equilibrium of growth and mortality. Although the concept of a site-specific carrying capacity usefully underlies many of the models of forest development, it is worth considering situations where disturbances might initiate alternate steady states by effecting changes in growth, mortality, or combustibility that persist through to the next disturbance.

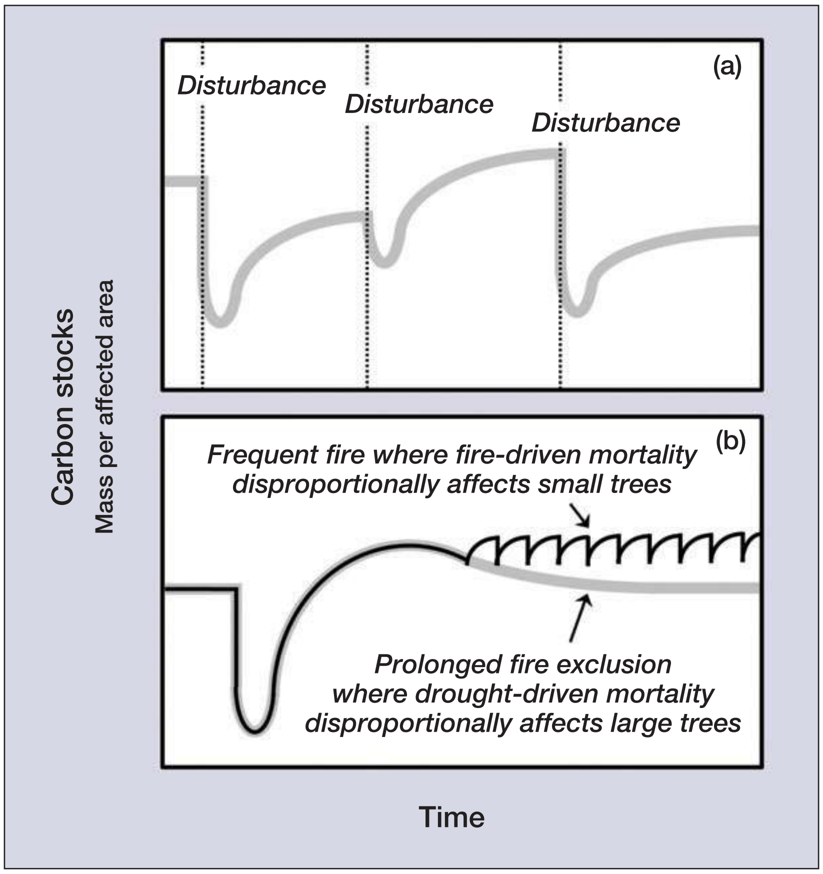

Figure 6. *Hypothetical examples of how disturbances could initiate alternate steady-state C stocks. (a) Illustration of what C stocks might look like if long-term successional trajectories were contingent more on seed availability at the time of fire than they were on fixed site conditions, as suggested by Kashian* et al. *(2006). (b) Illustration of how frequent fires could shift mortality away from larger trees and toward smaller trees, thus increasing steady-state C stocks, as suggested by North* et al. *(2009).*

A simple example of disturbance-altering, long-term forest growth involves the loss in soil fertility that can accompany certain high-severity fires (Johnson and Curtis 2001; Bormann *et al.* 2008). Another mechanism by which disturbance can initiate changes in steady-state C stocks involves the persistent changes in tree density that may follow some disturbance events. For instance, Kashian *et al.* (2006) determined that forest biomass in the lodgepole pine (*Pinus contorta*) forests of Yellowstone National Park was relatively insensitive to changes in fire frequency but very dependent on the density to which forests grew after fire. In a system where long-term successional trajectories are contingent more on forest condition at the time of disturbance (eg serotinous seed availability) than on permanent site conditions, C stocks could well stabilize at different levels after different disturbances, as illustrated in Figure 6a.

A final example of how changes in disturbance regime could persistently alter equilibrium between growth and mortality involves size-dependent mortality in the semiarid conifer forests of the Sierra Nevada (Smith *et al.* 2005). Both Fellows and Goulden (2008) and North *et al.* (2009) found fewer large trees and lower overall biomass in current fire-excluded forests than were believed to exist at these locations before fire exclusion. These authors suggest that small trees are disproportionally vulnerable to fire mortality, and large trees are disproportionally vulnerable to pathogen- and insect-based mortality; therefore, as biological agents replace fire as the primary cause of mortality, the number of large trees decreases accordingly. Under such scenarios, the thinning of small trees combined with frequent burning could, over time, increase biomass by maintaining a greater number of larger trees (see Figure 6b). However, not all studies support the notion that fire exclusion reduces stand-level biomass (Bouldin 2008; Hurteau *et al.* 2010). Specifically, another study conducted by North *et al.* (2007) in the Sierra Nevada found that net losses in large-diameter trees between 1865 and 2007 were more than compensated for by the infilling of small-diameter trees, such that total live-wood volume remained unchanged over this period of fire suppression. Furthermore, Hurteau and North (2010) reported that fire-suppressed control plots aggraded as much C over 7 years as did comparable thinned plots.

Presuming that maximum steady-state C stocks are not dictated entirely by permanent site qualities and depend, at least in some part, on the nature and timing of disturbance, it is conceivable that prescriptions such as fuel reduction and prescribed fire could eventually elevate (or reduce) C stocks at a single location slightly beyond what they would be under a different disturbance regime (Hurteau *et al.* 2010). However, exactly how stable or self-reinforcing this alternate state is remains unknown.

Additional considerations

The purpose of this paper is to illustrate the basic biophysical relationships that exist between fuel-reduction treatments, wildfire, and forest C stocks over time. Understanding these dynamics is necessary for crafting meaningful forest C policy; however, it is not by itself sufficient. A full accounting of C would also include the fossil-fuel costs of conducting fuel treatments, the longevity of forest products removed in fuel treatments, and the ability of fuel treatments to produce renewable "bioenergy", potentially offsetting combustion of fossil fuels. A detailed consideration of these factors is beyond the scope of this paper, but it is worth pointing out some limits of their contribution. First, the fossil-fuel costs of conducting fuel treatments are relatively small, ranging from 1–3% of the aboveground C stock (Finkral and Evans 2008; North *et al.* 2009; Stephens *et al.* 2009b). Second, only a small fraction of forest products ever enters "permanent" product stocks; this is especially true for the smaller-diameter trees typically removed during fuel treatments. Primarily, half-lives of forest products (7–70 years) are not significantly different than the half-life of the same biomass left in forests (Krankina and Harmon 2006). Third, the capacity of forest biofuels to offset C emissions from fossil-fuel consumption is greatly constrained by both transportation logistics and the lower energy output per unit C emitted as compared with fossil fuel (Marland and Schlamadinger 1997; Law and Harmon 2011).

Conclusions

The empirical data used in this paper derive from semi-arid, fire-prone conifer forests of the western US, which are largely composed of pine, true fir (*Abies* spp), and Douglas fir. These are the forests where management agencies are weighing the costs and benefits of up-scaling fuel-reduction treatments. Although it would be imprudent to insist that the quantitative responses reported in this paper necessarily apply to every manageable unit of fire-prone forest in the western US, our conclusions depend not so much on site-specific parameters but rather on the basic relationships – between growth, decomposition, harvest, and combustion – to which no forest is exempt. To simply acknowledge the following – that (1) forest wildfires primarily consume leaves and small branches, (2) even strategic fuels management often involves treating more area than wildfire would otherwise affect, and (3) the intrinsic trade-off between fire frequency and the amount of biomass available for combustion functions largely as a zero-sum game – leaves little room for any fuel-reduction treatment to result in greater sustained biomass regardless of system parameterization. Only when treatment, wildfire, or their interaction leads to changes in maximum biomass potential (ie system state change) can fuel treatment profoundly influence C storage.

In evaluating the effects of wildfire and fuel-reduction treatments on forest C stocks across various spatial and temporal scales, we conclude that:

(1) Empirical evidence shows that most pyrogenic C emissions arise from the combustion of surface fuels, and because surface fuel is combusted in almost all fire types, high-severity wildfires burn only 10% more of the standing biomass than do the low-severity fires that fuel treatment is intended to promote (Figure 1).
(2) Model simulations support the notion that forests subjected to fuel-reduction treatments experience less pyrogenic emissions when subsequently exposed to wildfires. However, across a range of treatment intensities, the amount of C removed in treatment was typically three times that saved by altering fire behavior (Figure 2).
(3) Fire-spread simulations suggest that strategic application of fuel-reduction treatments on as little as 1% of a landscape annually can reduce the area subject to severe wildfire by 50% over a 20-year period. Even so, the protection of one hectare of forest from wildfire required the treatment of 10 hectares, owing not to the low efficacy of treatment but rather to the rarity of severe wildfire events (Figure 3).
(4) It is reasonable to expect that after a forest-replacing fire, the decomposition of fire-killed trees exceeds NPP, driving NEP below zero. By contrast, the deliberate removal of necromass in fuel-reduction treatments could result in a period of elevated NEP. However, despite marked differences in post-disturbance NEP, it is possible for average C stocks to be identical for these two disturbance types (Figure 4).
(5) Long-term simulations of forest growth, decomposition, and combustion illustrate how, despite a negative feedback between fire frequency and fuel-driven severity, a regime of low-frequency, high-severity fire stores more C over time than a regime of high-frequency, low-severity fire (Figure 5).
(6) The degree to which fuel management could possibly lead to increased C storage over space and time is contingent on the capacity of such treatments to increase maximum achievable biomass through mechanisms such as decreased non-fire mortality or the protection from losses in soil fertility that are sometimes associated with the highest-severity fires (Figure 6).

There is a strong consensus that large portions of forests in the western US have suffered both structurally and compositionally from a century of fire exclusion and that certain fuel-reduction treatments, including the thinning of live trees and prescribed burning, can be effective tools for restoring historical functionality and fire resilience to these ecosystems (Hurteau *et al.* 2010; Meigs and Campbell 2010). Furthermore, by reducing the likelihood of high-severity wildfire, fuel-reduction treatments can improve public safety and reduce threats to the resources provided by mature forests.

On the basis of material reviewed in this paper, it appears unlikely that forest fuel-reduction treatments have the additional benefit of increasing terrestrial C storage simply by reducing future combustive losses and that, more often, treatment would result in a reduction in C stocks over space and time. Claims that fuel-reduction treatments reduce overall forest C emissions are generally not supported by first principles, modeling simulations, or empirical observations. The C gains that could be achieved by increasing the proportion of large to small trees in some forests are limited to the marginal and variable differences in biomass observed between fire-suppressed forests and those experiencing frequent burning of understory vegetation.

Emerging policies aimed at reducing atmospheric CO_2 emissions may well threaten land managers' ability to apply restoration prescriptions at the scale necessary to achieve and sustain desired forest conditions. For this reason, it is imperative that scientists continue research into the processes by which fire can mediate long-term C storage (eg charcoal formation, decomposition, and community state change) and more accurately quantify the unintended consequences of fuel-reduction treatments on global C cycling.

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WebTable 1. Biomass reductions associated with various fuel reduction treatments as prescribed at various fire-prone forests of western North America

Treatment type, forest type, and location	*Fraction of live basal area cut or killed in prescribed burn*	*Fate of logging slash*	*Δ surface fuels (estimated fraction)*	*Δ total aboveground biomass through both combustion and removal (estimated fraction)*
Prescribed fire only				
Sierran mixed conifer (Central Sierras)[a]	0.00	None	–0.70	–0.11
Sierran mixed conifer (Central Sierras)[b]	0.15	None	–0.02	–0.13
Ponderosa pine/Douglas fir (Northern Rockies)[b]	0.11	None	–0.19	–0.12
Ponderosa pine/Douglas fir (Blue Mountains)[b]	0.08	None	–0.32	–0.12
Ponderosa pine (Southwestern Plateau)[b]	0.04	None	–0.50	–0.11
Ponderosa pine/true fir (Southern Cascades)[b]	0.30	None	0.67	–0.16
Thinning only				
Sierran mixed conifer (Central Sierras)[a]	0.36	Left on site	0.96	–0.16
Sierran mixed conifer (Central Sierras)[a]	0.60	Left on site	1.60	–0.27
Ponderosa pine (Southern Rockies)[c]	0.36	Pile burned	0.01	–0.30
Ponderosa pine (Central Sierras)[d]	0.50	Pile burned	0.01	–0.42
Sierran mixed conifer (Central Sierras)[b]	0.34	Left on site	0.92	–0.15
Ponderosa pine/Douglas fir (Northern Rockies)[b]	0.54	Left on site	1.43	–0.24
Ponderosa pine/Douglas fir (Blue Mountains)[b]	0.24	Left on site	0.64	–0.11
Ponderosa pine (Southwestern Plateau)[b]	0.53	Pile burned	0.01	–0.45
Ponderosa pine/true fir (Southern Cascades)[b]	0.58	Removed	0.00	–0.49
Thinning and prescribed fire				
Sierran mixed conifer (Central Sierras)[a]	0.37	Left on site	–0.40	–0.38
Sierran mixed conifer (Central Sierras)[a]	0.66	Left on site	–0.17	–0.59
Ponderosa pine (Central Sierras)[d]	0.50	Pile burned	–0.69	–0.53
Sierran mixed conifer (Central Sierras)[b]	0.42	Left on site	–0.37	–0.41
Ponderosa pine/Douglas fir (Northern Rockies)[b]	0.78	Left on site	–0.08	–0.67
Ponderosa pine/Douglas fir (Blue Mountains)[b]	0.46	Left on site	–0.33	–0.44
Ponderosa pine (Southwestern Plateau)[b]	0.59	Pile burned	–0.68	–0.61
Ponderosa pine/true fir (Southern Cascades)[b]	0.73	Removed	–0.70	–0.72

Notes: Total biomass losses were approximated solely from basal reported area reduction according to the following assumptions: total aboveground biomass was assumed to be composed of 45% live merchantable boles (subject to removal proportional to basal area reduction), 40% live tree branch and foliage (converted to slash proportional to basal area reduction), and 15% surface fuels (both live and dead biomass and subject to combustion in prescribed fire). Prescribed fire was assumed to combust 70% of surface fuels and logging slash; pile burning was assumed to combust 99% of logging slash. [a]North *et al.* (2007); [b]Stephens *et al.* (2009); [c]Finkal and Evans (2008); [d]Campbell *et al.* (2008).

WebTable 2. Burn probability for forests of Oregon, Washington, and California from 1985 to 2005

Forest type (ecoregion)	*Fraction of forest area burned annually* — *Any severity*	*Fraction of forest area burned annually* — *High severity*	*Fuel-treatment life expectancy (range in years)*	*Random probability of a treated stand being exposed to any fire*
Cool–wet conifer (Coast Range)	0.00018	0.00002	5–15	0.0009–0.00274
Cool–mesic conifer (West Cascades, North Cascades)	0.00177	0.00046	5–15	0.00884–0.02651
Cool–dry conifer (East Cascades, North Rockies, Blue Mts)	0.00411	0.00054	10–25	0.04112–0.10279
Warm–mesic conifer (Klamath Mountains)	0.00622	0.00119	10–25	0.06217–0.15542
Warm–dry conifer (Sierra Nevada, South California Mts)	0.00780	0.00178	10–25	0.07798–0.19495

Notes: This simple prediction of wildfire-treatment occurrence by multiplying regional fire probability by fuel treatment life assumes random interaction of wildfire and treatment and does not account for strategic placement of fuel treatments. Area burned annually based on Monitoring Trends in Burn Severity fire perimeter and severity classification maps from 1985 to 2005 (http://mtbs.gov). Total forested area in each ecoregion based on 2005 National Land Cover Dataset land-cover maps (http://landcover.usgs.gov). Ecoregions correspond to Omernik Level 3 classification (Omernik 1978). Treatment life expectancies are crude estimates based on Rhodes and Baker (2008) and Agee and Skinner (2005). Being that these numbers were derived from actual region-wide land-surface-change detection, they include regional fire suppression activities. Natural burn probabilities, as well as those that may result from future management decisions or climate change, are likely to be higher.

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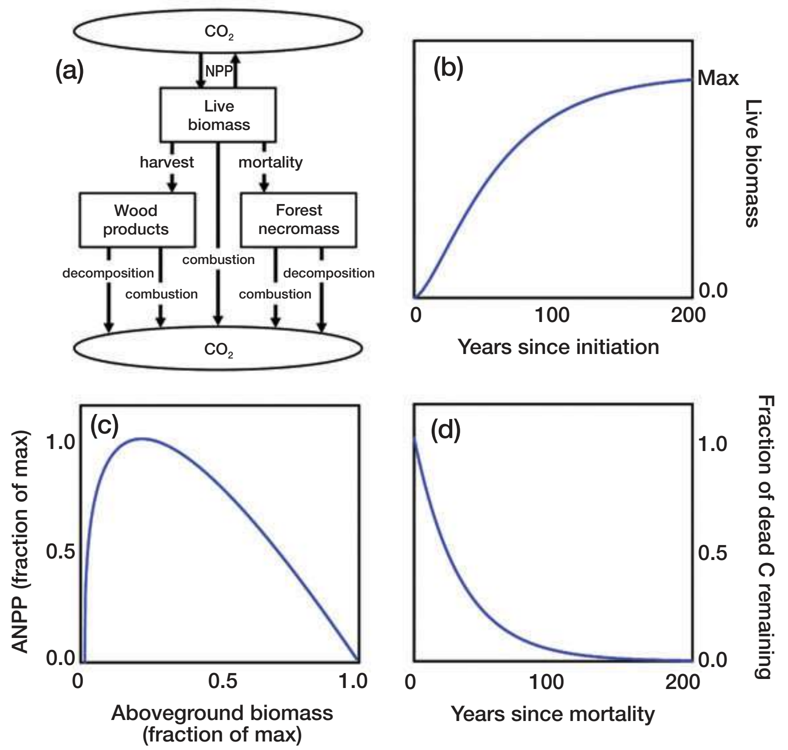
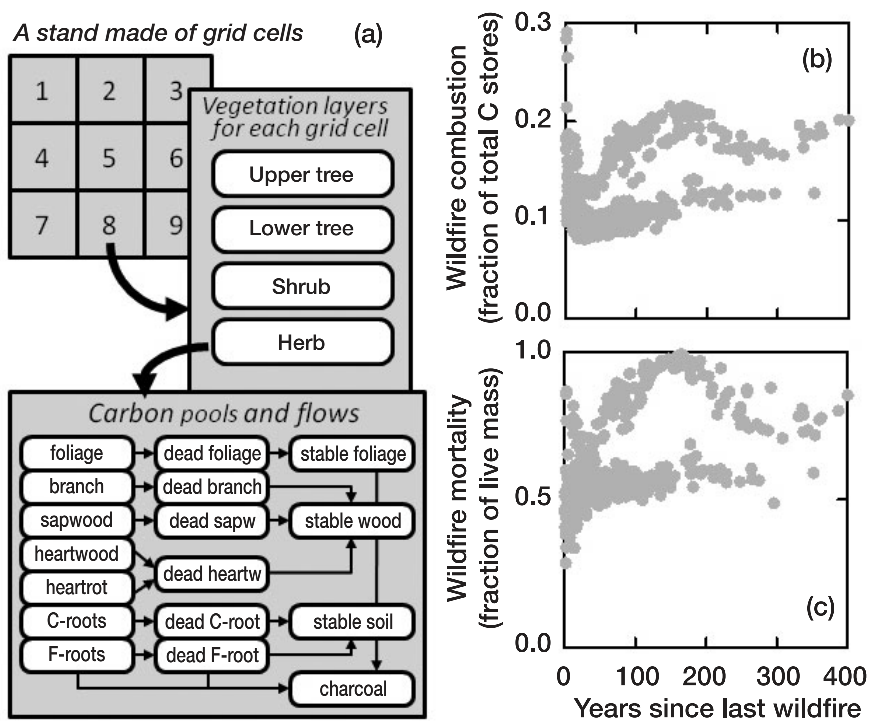


WebFigure 1. *(a) Structure and (c and d) dynamic functions behind the forest carbon model used to produce Figure 4. (b) Live biomass is assumed to aggrade over time according to a Chapman-Richards function* $y_1 = a^*(1 - \exp[-b_1x_1])^c$, *the derivative of which,* $y_2 = c^*b_1^*y_1^*(1 - \exp[\ln\{y_1/a\}/c])/\exp(\ln[y_1/a]/c)$, *allows (c) ANPP (aboveground net primary production) to be calculated annually according to current biomass;* y_1 *is aboveground live biomass in kg C* m^{-2}, y_2 *is ANPP in kg C* m^{-2} yr^{-1}, a *is the maximum aboveground live biomass that the site can sustain,* x_1 *is the time in years since initiation (which can be back-calculated from any assigned biomass),* b_1 *is a constant proportional to the time required to achieve maximum biomass, and* c *is a constant proportional to the initial growth lag. (d) Decomposition, the heterotrophic mineralization of each necromass pool including wood products, is determined according to an exponential loss function* $y_4 = M^* - k$, *where* y_4 *is loss of necromass in kg C* m^{-2} yr^{-1}, M *is the current mass of necromass in kg C* m^{-2}, *and* k *is a pool-specific decomposition constant. For the simulations shown in Figure 4, we used the following parameters to represent growth, harvest, combustion, and decay in a semiarid, fire-prone pine forest of western North America:* a = *4.8 kg C* m^{-2}; b_1 = *0.02;* c = *1.6;* k = *0.005* yr^{-1} *for both forest necromass and wood products; starting* y_1 = *4.5 kg C* m^{-2}; *starting* $M_{necromass}$ = *2.2 kg C* m^{-2}; *starting* $M_{products}$ = *0 Mg C* ha^{-1}; *treatment removals = 2.9 kg C* m^{-2}; *treatment related mortality (uncombusted slash) = 1% of* y_1 *at time of treatment; wildfire mortality = 95% of* y_1 *at time of fire; wildfire combustion = 10% of* y_1 *and 40%* M *of at time of fire.*



WebFigure 2. *(a) Structure and (b and c) disturbance responses behind STANDCARB, the forest carbon model used to produce Figure 5. STANDCARB simulates the accumulation of C over succession in mixed-species and mixed-age forest stands at annual time steps. The growth of vegetation and subsequent transfer of C among the various carbon pools shown in (a) are regulated by user-defined edapho-climatic inputs and species-specific responses. The imposition of wildfire in any given year results in the instantaneous transfer of C from each live pool into its corresponding dead pool (wildfire mortality) and the instantaneous loss of C from each live and dead pool to the atmosphere (wildfire combustion). The exact amount of mortality and combustion incurred in a given wildfire depends on stand-specific species composition, and the amount of biomass in each separate C pool, which at any given time may not be in equilibrium (gray circles in [b] and [c] reflect this variation). For the simulations shown in Figure 5, STANDCARB was parameterized for a semiarid ponderosa pine forest growing in eastern Oregon: max attainable biomass = 210 Mg C ha^{-1}; mean ANPP = 5.1 Mg C ha^{-1} yr^{-1}; non-fire mortality rate constants = 0.37, 0.5, 0.032, 0.017, and 0.013 yr^{-1} for foliage, fine roots, branches, coarse roots, and stems, respectively; decomposition rate constants = 0.21, 0.15, 0.08, 0.11, 0.023, and 0.017 yr^{-1} for foliage, fine roots, branches, coarse roots, stems, and soil C, respectively. It is worth noting that patterns nearly identical to those illustrated in Figure 5 result from STANDCARB parameterized for a mesic Douglas-fir forest having much larger ANPP, potential biomass, and decomposition rates. For a full description of STANDCARB structure and parameterization, see Harmon* et al. *(2009) and http://andrewsforest.oregonstate.edu/lter/pubs/webdocs/models/standcarb2.htm.*

