Questions from Representative Ruben Gallego

1. Dr. Law, you've testified that intact natural landscapes are critical for carbon sequestration, biodiversity, and climate resilience. Do you believe that National Forest areas as protected by the Roadless Area Conservation Rule should qualify as conservation lands based on those criteria?

A: Yes, however grazing, mining, and drilling can still be a threat, even in Roadless Areas. Permanent protections at the highest levels are needed for carbon, biodiversity and climate resilience. For President Biden's 30x30 goal, land and waters should have the same level of protection as Wilderness areas without grazing, and be permanently protected from roads, logging and other development. This type of permanence is best met on federal lands.

2. Last year, the Trump administration issued a rule to exempt the Tongass National Forest from all protections provided by the Roadless Rule. Do you believe that the Alaska Roadless Exemption supports carbon, biodiversity, or climate goals?

A: No, the exemption is at odds with and does not support carbon, biodiversity and/or climate goals. The Tongass National Forest is probably our most important forest for its carbon storage and accumulation, biodiversity and climate resilience. The Tongass contains the most intact forest area in the US. Intact forests are essential for carbon storage, biodiversity, and climate resilience. Congress should reinstate the previous protections under the roadless rule, and establish additional protections for all remaining intact, old forests in the Tongass. Old forests in the Tongass need to be permanently protected from any future logging.

3. With nearly 100 of my Democratic colleagues, I have introduced the Roadless Area Conservation Act. The bill would reverse the Alaska Roadless Exemption and permanently protect nearly 60 million acres of Roadless National Forest System land. Do you believe this bill could support climate and conservation goals? Do you think there are opportunities to strengthen Roadless or other climate-smart protections.

A: Yes, this bill is an essential step. I will note that in my home state of Oregon, we have ~4.7 million acres of un-inventoried roadless areas that are candidates for protection (federal lands generally in natural condition and over 1,000 acres in size). I encourage you to consider requiring updated inventories and protecting the mature and old forest areas within them that are not necessarily accounted for in the inventoried Roadless Areas. The role of these systems in carbon storage and climate resilience is tremendous. This bill should be enacted into law as soon as possible, and any additional logging in remaining intact old growth forests should be prohibited.

Our preliminary analysis in the western US suggests that beyond the ~43 M acres forest area currently protected permanently at the highest levels (GAP status 1 and 2), substantial additional acres of existing mature and old forests in the region with relatively high carbon and rich biodiversity could be protected at these levels. Approximately half of this forest area is area is on federal land, while a quarter is on private land. Thus, protections of more natural areas, especially those on federal lands would be a low cost way to simultaneously meeting climate goals and protecting threatened biodiversity. Paying private forestland owners for the service of storing carbon has the potential to store even more carbon in a cost effective manner during the next critical 30 years. Additional funding for permanent conservation easements on private lands would help achieve this goal. This is the lowest cost option for atmospheric CO_2 removal and it is available and working now.

Land use strategies to mitigate climate change in carbon dense temperate forests

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Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but ecosystem process-based integration of climate change, enhanced CO_2, disturbance from fire, and management actions at regional scales are extremely limited. Here, we examine the relative merits of afforestation, reforestation, management changes, and harvest residue bioenergy use in the Pacific Northwest. This region represents some of the highest carbon density forests in the world, which can store carbon in trees for 800 y or more. Oregon's net ecosystem carbon balance (NECB) was equivalent to 72% of total emissions in 2011–2015. By 2100, simulations show increased net carbon uptake with little change in wildfires. Reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increase NECB 56% by 2100, with the latter two actions contributing the most. Resultant cobenefits included water availability and biodiversity, primarily from increased forest area, age, and species diversity. Converting 127,000 ha of irrigated grass crops to native forests could decrease irrigation demand by 233 billion $m^3 \cdot y^{-1}$. Utilizing harvest residues for bioenergy production instead of leaving them in forests to decompose increased emissions in the short-term (50 y), reducing mitigation effectiveness. Increasing forest carbon on public lands reduced emissions compared with storage in wood products because the residence time is more than twice that of wood products. Hence, temperate forests with high carbon densities and lower vulnerability to mortality have substantial potential for reducing forest sector emissions. Our analysis framework provides a template for assessments in other temperate regions.

forests | carbon balance | greenhouse gas emissions | climate mitigation

Strategies to mitigate carbon dioxide emissions through forestry activities have been proposed, but regional assessments to determine feasibility, timeliness, and effectiveness are limited and rarely account for the interactive effects of future climate, atmospheric CO_2 enrichment, nitrogen deposition, disturbance from wildfires, and management actions on forest processes. We examine the net effect of all of these factors and a suite of mitigation strategies at fine resolution (4-km grid). Proven strategies immediately available to mitigate carbon emissions from forest activities include the following: (*i*) reforestation (growing forests where they recently existed), (*ii*) afforestation (growing forests where they did not recently exist), (*iii*) increasing carbon density of existing forests, and (*iv*) reducing emissions from deforestation and degradation (1). Other proposed strategies include wood bioenergy production (2–4), bioenergy combined with carbon capture and storage (BECCS), and increasing wood product use in buildings. However, examples of commercial-scale BECCS are still scarce, and sustainability of wood sources remains controversial because of forgone ecosystem carbon storage and low environmental cobenefits (5, 6). Carbon stored in buildings generally outlives its usefulness or is replaced within decades (7) rather than the centuries possible in forests, and the factors influencing product substitution have yet to be fully explored (8). Our analysis of mitigation strategies focuses on the first four strategies, as well as bioenergy production, utilizing harvest residues only and without carbon capture and storage.

The appropriateness and effectiveness of mitigation strategies within regions vary depending on the current forest sink, competition with land-use and watershed protection, and environmental conditions affecting forest sustainability and resilience. Few process-based regional studies have quantified strategies that could actually be implemented, are low-risk, and do not depend on developing technologies. Our previous studies focused on regional modeling of the effects of forest thinning on net ecosystem carbon balance (NECB) and net emissions, as well as improving modeled drought sensitivity (9, 10), while this study focuses mainly on strategies to enhance forest carbon.

Our study region is Oregon in the Pacific Northwest, where coastal and montane forests have high biomass and carbon sequestration potential. They represent coastal forests from northern California to southeast Alaska, where trees live 800 y or more and biomass can exceed that of tropical forests (11) (Fig. S1). The semiarid ecoregions consist of woodlands that experience frequent fires (12). Land-use history is a major determinant of forest carbon balance. Harvest was the dominant cause of tree mortality (2003–2012) and accounted for fivefold as much mortality as that from fire and beetles combined (13). Forest land ownership is predominantly public (64%), and 76% of the biomass harvested is on private lands.

Significance

Regional quantification of feasibility and effectiveness of forest strategies to mitigate climate change should integrate observations and mechanistic ecosystem process models with future climate, CO_2, disturbances from fire, and management. Here, we demonstrate this approach in a high biomass region, and found that reforestation, afforestation, lengthened harvest cycles on private lands, and restricting harvest on public lands increased net ecosystem carbon balance by 56% by 2100, with the latter two actions contributing the most. Forest sector emissions tracked with our life cycle assessment model decreased by 17%, partially meeting emissions reduction goals. Harvest residue bioenergy use did not reduce short-term emissions. Cobenefits include increased water availability and biodiversity of forest species. Our improved analysis framework can be used in other temperate regions.

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The authors declare no conflict of interest.

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Data deposition: The CLM4.5 model data are available at Oregon State University (terraweb.forestry.oregonstate.edu/FMEC). Data from the >200 intensive plots on forest carbon are available at Oak Ridge National Laboratory (https://daac.ornl.gov/NACP/guides/NACP_TERRA-PNW.html), and FIA data are available at the USDA Forest Service (https://www.fia.fs.fed.us/tools-data/).

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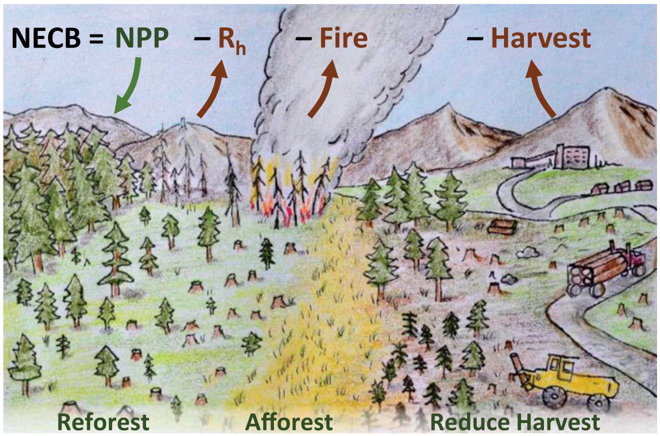

Fig. 1. Approach to assessing effects of mitigation strategies on forest carbon and forest sector emissions. NECB is productivity (NPP) minus Rh and losses from fire and harvest (red arrows). Harvest emissions include those associated with wood products and bioenergy.

Many US states, including Oregon (14), plan to reduce their greenhouse gas (GHG) emissions in accordance with the Paris Agreement. We evaluated strategies to address this question: How much carbon can the region's forests realistically remove from the atmosphere in the future, and which forest carbon strategies can reduce regional emissions by 2025, 2050, and 2100? We propose an integrated approach that combines observations with models and a life cycle assessment (LCA) to evaluate current and future effects of mitigation actions on forest carbon and forest sector emissions in temperate regions (Fig. 1). We estimated the recent carbon budget of Oregon's forests, and simulated the potential to increase the forest sink and decrease forest sector emissions under current and future climate conditions. We provide recommendations for regional assessments of mitigation strategies.

Results

Carbon stocks and fluxes are summarized for the observation cycles of 2001–2005, 2006–2010, and 2011–2015 (Table 1 and Tables S1 and S2). In 2011–2015, state-level forest carbon stocks totaled 3,036 Tg C (3 billion metric tons), with the coastal and montane ecoregions accounting for 57% of the live tree carbon (Tables S1 and S2). Net ecosystem production [NEP; net primary production (NPP) minus heterotrophic respiration (Rh)] averaged 28 teragrams carbon per year (Tg C y^{-1}) over all three periods. Fire emissions were unusually high at 8.69 million metric tons carbon dioxide equivalent (tCO_2e y^{-1}, i.e., 2.37 Tg C y^{-1}) in 2001–2005 due to the historic Biscuit Fire, but decreased to 3.56 million tCO_2e y^{-1} (0.97 Tg C y^{-1}) in 2011–2015 (Table S4). Note that 1 million tCO_2e equals 3.667 Tg C.

Our LCA showed that in 2001–2005, Oregon's net wood product emissions were 32.61 million tCO_2e (Table S3), and 3.7-fold wildfire emissions in the period that included the record fire year (15) (Fig. 2). In 2011–2015, net wood product emissions were 34.45 million tCO_2e and almost 10-fold fire emissions, mostly due to lower fire emissions. The net wood product emissions are higher than fire emissions despite carbon benefits of storage in wood products and substitution for more fossil fuel-intensive products. Hence, combining fire and net wood product emissions, the forest sector emissions averaged 40 million tCO_2e y^{-1} and accounted for about 39% of total emissions across all sectors (Fig. 2 and Table S4). NECB was calculated from NEP minus losses from fire emissions and harvest (Fig. 1). State NECB was equivalent to 60% and 70% of total emissions for 2001–2005 and 2011–2015, respectively (Fig. 2, Table 1, and Table S4). Fire emissions were only between 4% and 8% of total emissions from all sources (2011–2015 and 2001–2004, respectively). Oregon's forests play a larger role in meeting its GHG targets than US forests have in meeting the nation's targets (16, 17).

Historical disturbance regimes were simulated using stand age and disturbance history from remote sensing products. Comparisons of Community Land Model (CLM4.5) output with Forest Inventory and Analysis (FIA) aboveground tree biomass (>6,000 plots) were within 1 SD of the ecoregion means (Fig. S2). CLM4.5 estimates of cumulative burn area and emissions from 1990 to 2014 were 14% and 25% less than observed, respectively. The discrepancy was mostly due to the model missing an anomalously large fire in 2002 (Fig. S3*A*). When excluded, modeled versus observed fire emissions were in good agreement ($r^2 = 0.62$; Fig. S3*B*). A sensitivity test of a 14% underestimate of burn area did not affect our final results because predicted emissions would increase almost equally for business as usual (BAU) management and our scenarios, resulting in no proportional change in NECB. However, the ratio of harvest to fire emissions would be lower.

Projections show that under future climate, atmospheric carbon dioxide, and BAU management, an increase in net carbon uptake due to CO_2 fertilization and climate in the mesic ecoregions far outweighs losses from fire and drought in the semiarid ecoregions. There was not an increasing trend in fire. Carbon stocks increased by 2% and 7% and NEP increased by 12% and 40% by 2050 and 2100, respectively.

We evaluated emission reduction strategies in the forest sector: protecting existing forest carbon, lengthening harvest cycles, reforestation, afforestation, and bioenergy production with product substitution. The largest potential increase in forest carbon is in the mesic Coast Range and West Cascade ecoregions. These forests are buffered by the ocean, have high soil water-holding capacity, low risk of wildfire [fire intervals average 260–400 y (18)], long carbon residence time, and potential for high carbon density. They can attain biomass up to 520 Mg C ha^{-1} (12). Although Oregon has several protected areas, they account for only 9–15% of the total forest area, so we expect it may be feasible to add carbon-protected lands with cobenefits of water protection and biodiversity.

Reforestation of recently forested areas include those areas impacted by fire and beetles. Our simulations to 2100 assume regrowth of the same species and incorporate future fire responses to climate and cyclical beetle outbreaks [70–80 y (13)]. Reforestation has the potential to increase stocks by 315 Tg C by 2100, reducing forest sector net emissions by 5% by 2100 relative to BAU management (Fig. 3). The East and West Cascades ecoregions had the highest reforestation potential, accounting for 90% of the increase (Table S5).

Afforestation of old fields within forest boundaries and nonfood/nonforage grass crops, hereafter referred to as "grass crops," had to meet minimum conditions for tree growth, and crop grid cells had to be partially forested (*SI Methods* and Table S6). These crops are not grazed or used for animal feed. Competing land uses may decrease the actual amount of area that can be afforested. We calculated the amount of irrigated grass crops (127,000 ha) that could be converted to forest, assuming success of carbon offset programs (19). By 2100, afforestation increased stocks by

Table 1. Forest carbon budget components used to compute NECB

Flux, Tg C·y^{-1}	2001–2005		2006–2010		2011–2015		2001–2015
NPP	73.64	7.59	73.57	7.58	73.57	7.58	73.60
Rh	45.67	5.11	45.38	5.07	45.19	5.05	45.41
NEP	27.97	9.15	28.19	9.12	28.39	9.11	28.18
Harvest removals	8.58	0.60	7.77	0.54	8.61	0.6	8.32
Fire emissions	2.37	0.27	1.79	0.2	0.97	0.11	1.71
NECB	17.02	9.17	18.63	9.14	18.81	9.13	18.15

Average annual values for each period, including uncertainty (95% confidence interval) in Tg C y^{-1} (multiply by 3.667 to get million tCO_2e).

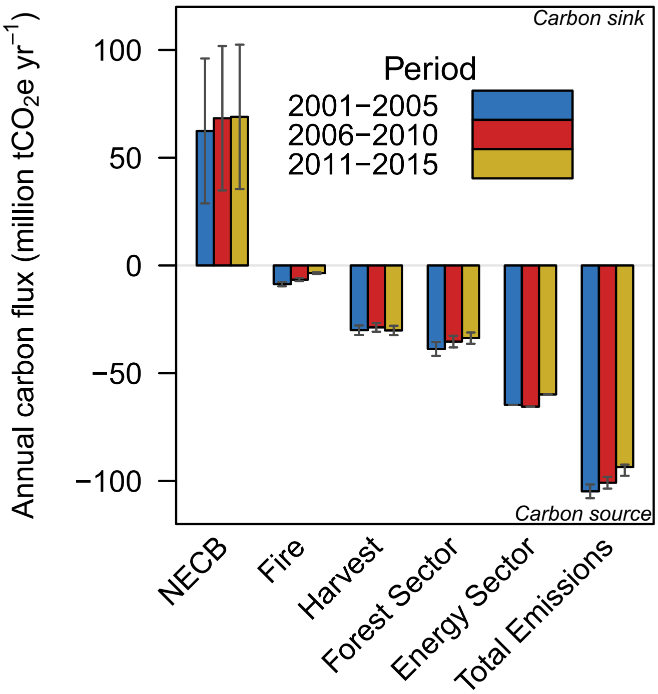

Fig. 2. Oregon's forest carbon sink and emissions from forest and energy sectors. Harvest emissions are computed by LCA. Fire and harvest emissions sum to forest sector emissions. Energy sector emissions are from the Oregon Global Warming Commission (14), minus forest-related emissions. Error bars are 95% confidence intervals (Monte Carlo analysis).

94 Tg C and cumulative NECB by 14 Tg C, and afforestation reduced forest sector GHG emissions by 1.3–1.4% in 2025, 2050, and 2100 (Fig. 3).

We quantified cobenefits of afforestation of irrigated grass crops on water availability based on data from hydrology and agricultural simulations of future grass crop area and related irrigation demand (20). Afforestation of 127,000 ha of grass cropland with Douglas fir could decrease irrigation demand by 222 and 233 billion $m^3 \cdot y^{-1}$ by 2050 and 2100, respectively. An independent estimate from measured precipitation and evapotranspiration (ET) at our mature Douglas fir and grass crop flux sites in the Willamette Valley shows the ET/precipitation fraction averaged 33% and 52%, respectively, and water balance (precipitation minus ET) averaged 910 $mm \cdot y^{-1}$ and 516 $mm \cdot y^{-1}$. Under current climate conditions, the observations suggest an increase in annual water availability of 260 billion $m^3 \cdot y^{-1}$ if 127,000 ha of the irrigated grass crops were converted to forest.

Harvest cycles in the mesic and montane forests have declined from over 120 y to 45 y despite the fact that these trees can live 500–1,000 y and net primary productivity peaks at 80–125 y (21). If harvest cycles were lengthened to 80 y on private lands and harvested area was reduced 50% on public lands, state-level stocks would increase by 17% to a total of ~3,600 Tg C and NECB would increase 2–3 Tg C y^{-1} by 2100. The lengthened harvest cycles reduced harvest by 2 Tg C y^{-1}, which contributed to higher NECB. Leakage (more harvest elsewhere) is difficult to quantify and could counter these carbon gains. However, because harvest on federal lands was reduced significantly since 1992 (NW Forest Plan), leakage has probably already occurred.

The four strategies together increased NECB by 64%, 82%, and 56% by 2025, 2050, and 2100, respectively. This reduced forest sector net emissions by 11%, 10%, and 17% over the same periods (Fig. 3). By 2050, potential increases in NECB were largest in the Coast Range (Table S5), East Cascades, and Klamath Mountains, accounting for 19%, 25%, and 42% of the total increase, whereas by 2100, they were most evident in the West Cascades, East Cascades, and Klamath Mountains.

We examined the potential for using existing harvest residue for electricity generation, where burning the harvest residue for energy emits carbon immediately (3) versus the BAU practice of leaving residues in forests to slowly decompose. Assuming half of forest residues from harvest practices could be used to replace natural gas or coal in distributed facilities across the state, they would provide an average supply of 0.75–1 Tg C y^{-1} to the year 2100 in the reduced harvest and BAU scenarios, respectively. Compared with BAU harvest practices, where residues are left to decompose, proposed bioenergy production would increase cumulative net emissions by up to 45 Tg C by 2100. Even at 50% use, residue collection and transport are not likely to be economically viable, given the distances (>200 km) to Oregon's facilities.

Discussion

Earth system models have the potential to bring terrestrial observations related to climate, vulnerability, impacts, adaptation,

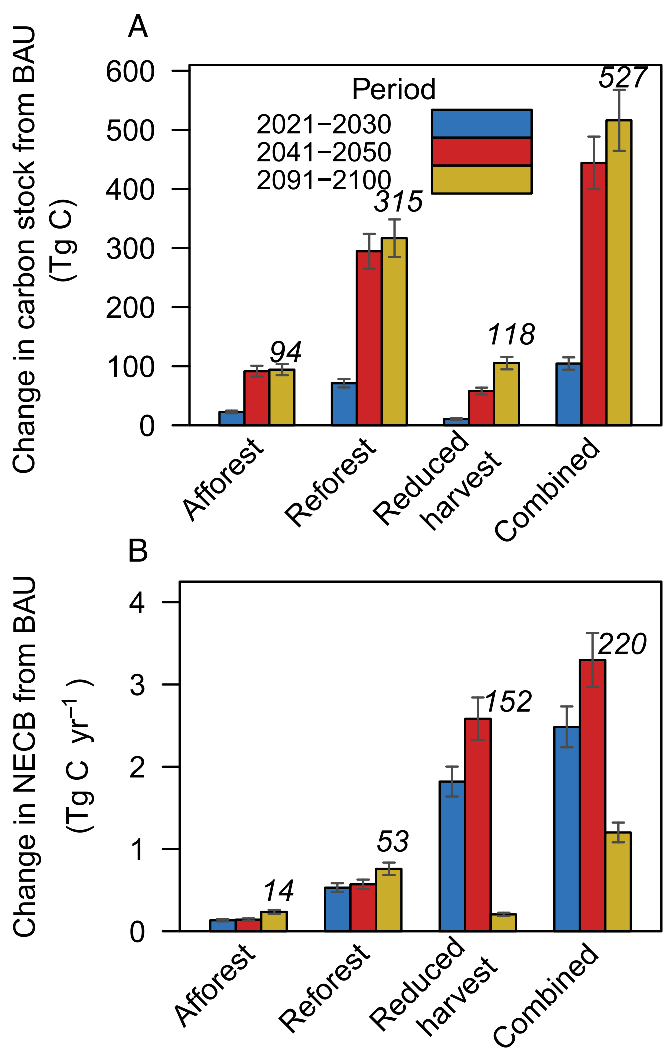

Fig. 3. Future change in carbon stocks and NECB with mitigation strategies relative to BAU management. The decadal average change in forest carbon stocks (*A*) and NECB relative to BAU (*B*) are shown. Italicized numbers over bars indicate mean forest carbon stocks in 2091–2100 (*A*) and cumulative change in NECB for 2015–2100 (*B*). Error bars are ±10%.

and mitigation into a common framework, melding biophysical with social components (22). We developed a framework to examine a suite of mitigation actions to increase forest carbon sequestration and reduce forest sector emissions under current and future environmental conditions.

Harvest-related emissions had a large impact on recent forest NECB, reducing it by an average of 34% from 2001 to 2015. By comparison, fire emissions were relatively small and reduced NECB by 12% in the Biscuit Fire year, but only reduced NECB 5–9% from 2006 to 2015. Thus, altered forest management has the potential to enhance the forest carbon balance and reduce emissions.

Future NEP increased because enhancement from atmospheric carbon dioxide outweighed the losses from fire. Lengthened harvest cycles on private lands to 80 y and restricting harvest to 50% of current rates on public lands increased NECB the most by 2100, accounting for 90% of total emissions reduction (Fig. 3 and Tables S5 and S6). Reduced harvest led to NECB increasing earlier than the other strategies (by 2050), suggesting this could be a priority for implementation.

Our afforestation estimates may be too conservative by limiting them to nonforest areas within current forest boundaries and 127,000 ha of irrigated grass cropland. There was a net loss of 367,000 ha of forest area in Oregon and Washington combined from 2001 to 2006 (23), and less than 1% of native habitat remains in the Willamette Valley due to urbanization and agriculture (24). Perhaps more of this area could be afforested.

The spatial variation in the potential for each mitigation option to improve carbon stocks and fluxes shows that the reforestation potential is highest in the Cascade Mountains, where fire and insects occur (Fig. 4). The potential to reduce harvest on public land is highest in the Cascade Mountains, and that to lengthen harvest cycles on private lands is highest in the Coast Range.

Although western Oregon is mesic with little expected change in precipitation, the afforestation cobenefits of increased water availability will be important. Urban demand for water is projected to increase, but agricultural irrigation will continue to consume much more water than urban use (25). Converting 127,000 ha of irrigated grass crops to native forests appears to be a win–win strategy, returning some of the area to forest land, providing habitat and connectivity for forest species, and easing irrigation demand. Because the afforested grass crop represents only 11% of the available grass cropland (1.18 million ha), it is not likely to result in leakage or indirect land use change. The two forest strategies combined are likely to be important contributors to water security.

Cobenefits with biodiversity were not assessed in our study. However, a recent study showed that in the mesic forests, cobenefits with biodiversity of forest species are largest on lands with harvest cycles longer than 80 y, and thus would be most pronounced on private lands (26). We selected 80 y for the harvest cycle mitigation strategy because productivity peaks at 80–125 y in this region, which coincides with the point at which cobenefits with wildlife habitat are substantial.

Habitat loss and climate change are the two greatest threats to biodiversity. Afforestation of areas that are currently grass crops would likely improve the habitat of forest species (27), as about 90% of the forests in these areas were replaced by agriculture. About 45 mammal species are at risk because of range contraction (28). Forests are more efficient at dissipating heat than grass and crop lands, and forest cover gains lead to net surface cooling in all regions south of about 45° latitude in North American and Europe (29). The cooler conditions can buffer climate-sensitive bird populations from approaching their thermal limits and provide more food and nest sites (30). Thus, the mitigation strategies of afforestation, protecting forests on public lands and lengthening harvest cycles to 80–125 y, would likely benefit forest-dependent species.

Oregon has a legislated mandate to reduce emissions, and is considering an offsets program that limits use of offsets to 8% of

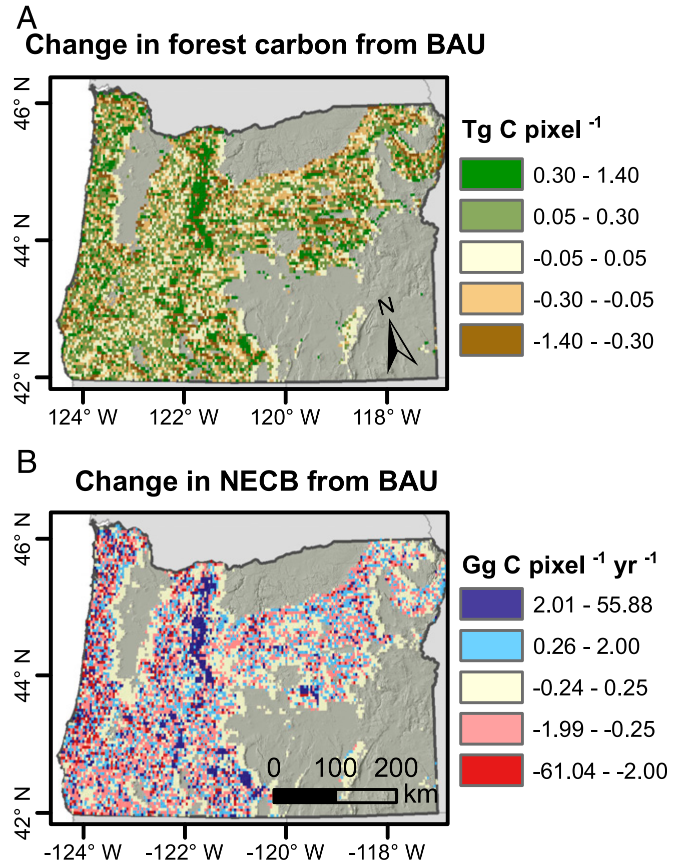

Fig. 4. Spatial patterns of forest carbon stocks and NECB by 2091–2100. The decadal average changes in forest carbon stocks (*A*) and NECB (*B*) due to afforestation, reforestation, protected areas, and lengthened harvest cycles relative to continued BAU forest management (red is increase in NECB) are shown.

the total emissions reduction to ensure that regulated entities substantially reduce their own emissions, similar to California's program (19). An offset becomes a net emissions reduction by increasing the forest carbon sink (NECB). If only 8% of the GHG reduction is allowed for forest offsets, the limits for forest offsets would be 2.1 and 8.4 million metric tCO_2e of total emissions by 2025 and 2050, respectively (Table S6). The combination of afforestation, reforestation, and reduced harvest would provide 13 million metric tCO_2e emissions reductions, and any one of the strategies or a portion of each could be applied. Thus, additionality beyond what would happen without the program is possible.

State-level reporting of GHG emissions includes the agriculture sector, but does not appear to include forest sector emissions, except for industrial fuel (i.e., utility fuel in Table S3) and, potentially, fire emissions. Harvest-related emissions should be quantified, as they are much larger than fire emissions in the western United States. Full accounting of forest sector emissions is necessary to meet climate mitigation goals.

Increased long-term storage in buildings and via product substitution has been suggested as a potential climate mitigation option. Pacific temperate forests can store carbon for many hundreds of years, which is much longer than is expected for buildings that are generally assumed to outlive their usefulness or be replaced within several decades (7). By 2035, about 75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends (31, 32). Recent analysis suggests substitution benefits of using wood versus more fossil fuel-intensive materials have been overestimated by at least an order of magnitude (33). Our LCA accounts for losses in product substitution stores (PSSs) associated with building life span, and thus are considerably lower than when no losses are assumed (4, 34). While product substitution reduces the overall forest sector emissions, it cannot offset the losses incurred by frequent harvest and losses associated with product transportation, manufacturing, use, disposal, and decay. Methods for calculating substitution benefits should be improved in other regional assessments.

Wood bioenergy production is interpreted as being carbon-neutral by assuming that trees regrow to replace those that burned. However, this does not account for reduced forest carbon stocks that took decades to centuries to sequester, degraded productive capacity, emissions from transportation and the production process, and biogenic/direct emissions at the facility (35). Increased harvest through proposed thinning practices in the region has been shown to elevate emissions for decades to centuries regardless of product end use (36). It is therefore unlikely that increased wood bioenergy production in this region would decrease overall forest sector emissions.

Conclusions

GHG reduction must happen quickly to avoid surpassing a 2 °C increase in temperature since preindustrial times. Alterations in forest management can contribute to increasing the land sink and decreasing emissions by keeping carbon in high biomass forests, extending harvest cycles, reforestation, and afforestation. Forests are carbon-ready and do not require new technologies or infrastructure for immediate mitigation of climate change. Growing forests for bioenergy production competes with forest carbon sequestration and does not reduce emissions in the next decades (10). BECCS requires new technology, and few locations have sufficient geological storage for CO_2 at power facilities with high-productivity forests nearby. Accurate accounting of forest carbon in trees and soils, NECB, and historic harvest rates, combined with transparent quantification of emissions from the wood product process, can ensure realistic reductions in forest sector emissions.

As states and regions take a larger role in implementing climate mitigation steps, robust forest sector assessments are urgently needed. Our integrated approach of combining observations, an LCA, and high-resolution process modeling (4-km grid vs. typical 200-km grid) of a suite of potential mitigation actions and their effects on forest carbon sequestration and emissions under changing climate and CO_2 provides an analysis framework that can be applied in other temperate regions.

Materials and Methods

Current Stocks and Fluxes. We quantified recent forest carbon stocks and fluxes using a combination of observations from FIA; Landsat products on forest type, land cover, and fire risk; 200 intensive plots in Oregon (37); and a wood decomposition database. Tree biomass was calculated from species-specific allometric equations and ecoregion-specific wood density. We estimated ecosystem carbon stocks, NEP (photosynthesis minus respiration), and NECB (NEP minus losses due to fire or harvest) using a mass-balance approach (36, 38) (Table 1 and *SI Materials and Methods*). Fire emissions were computed from the Monitoring Trends in Burn Severity database, biomass data, and region-specific combustion factors (15, 39) (*SI Materials and Methods*).

Future Projections and Model Description. Carbon stocks and NEP were quantified to the years 2025, 2050, and 2100 using CLM4.5 with physiological parameters for 10 major forest species, initial forest biomass (36), and future climate and atmospheric carbon dioxide as input (Institut Pierre Simon Laplace climate system model downscaled to 4 km × 4 km, representative concentration pathway 8.5). CLM4.5 uses 3-h climate data, ecophysiological characteristics, site physical characteristics, and site history to estimate the daily fluxes of carbon, nitrogen, and water between the atmosphere, plant state variables, and litter and soil state variables. Model components are biogeophysics, hydrological cycle, and biogeochemistry. This model version does not include a dynamic vegetation model to simulate resilience and establishment following disturbance. However, the effect of regeneration lags on forest carbon is not particularly strong for the long disturbance intervals in this study (40). Our plant functional type (PFT) parameterization for 10 major forest species rather than one significantly improves carbon modeling in the region (41).

Forest Management and Land Use Change Scenarios. Harvest cycles, reforestation, and afforestation were simulated to the year 2100. Carbon stocks and NEP were predicted for the current harvest cycle of 45 y compared with simulations extending it to 80 y. Reforestation potential was simulated over areas that recently suffered mortality from harvest, fire, and 12 species of beetles (13). We assumed the same vegetation regrew to the maximum potential, which is expected with the combination of natural regeneration and planting that commonly occurs after these events. Future BAU harvest files were constructed using current harvest rates, where county-specific average harvest and the actual amounts per ownership were used to guide grid cell selection. This resulted in the majority of harvest occurring on private land (70%) and in the mesic ecoregions. Beetle outbreaks were implemented using a modified mortality rate of the lodgepole pine PFT with 0.1% y^{-1} biomass mortality by 2100.

For afforestation potential, we identified areas that are within forest boundaries that are not currently forest and areas that are currently grass crops. We assumed no competition with conversion of irrigated grass crops to urban growth, given Oregon's land use laws for developing within urban growth boundaries. A separate study suggested that, on average, about 17% of all irrigated agricultural crops in the Willamette Valley could be converted to urban area under future climate; however, because 20% of total cropland is grass seed, it suggests little competition with urban growth (25).

Landsat observations (12,500 scenes) were processed to map changes in land cover from 1984 to 2012. Land cover types were separated with an unsupervised K-means clustering approach. Land cover classes were assigned to an existing forest type map (42). The CropScape Cropland Data Layer (CDL 2015, https://nassgeodata.gmu.edu/CropScape/) was used to distinguish nonforage grass crops from other grasses. For afforestation, we selected grass cropland with a minimum soil water-holding capacity of 150 mm and minimum precipitation of 500 mm that can support trees (43).

Afforestation Cobenefits. Modeled irrigation demand of grass seed crops under future climate conditions was previously conducted with hydrology and agricultural models, where ET is a function of climate, crop type, crop growth state, and soil-holding capacity (20) (Table S7). The simulations produced total land area, ET, and irrigation demand for each cover type. Current grass seed crop irrigation in the Willamette Valley is 413 billion $m^3 \cdot y^{-1}$ for 238,679 ha and is projected to be 412 and 405 billion m^3 in 2050 and 2100 (20) (Table S7). We used annual output from the simulations to estimate irrigation demand per unit area of grass seed crops (1.73, 1.75, and 1.84 million $m^3 \cdot ha^{-1}$ in 2015, 2050, and 2100, respectively), and applied it to the mapped irrigated crop area that met conditions necessary to support forests (Table S7).

LCA. Decomposition of wood through the product cycle was computed using an LCA (8, 10). Carbon emissions to the atmosphere from harvest were calculated annually over the time frame of the analysis (2001–2015). The net carbon emissions equal NECB plus total harvest minus wood lost during manufacturing and wood decomposed over time from product use. Wood industry fossil fuel emissions were computed for harvest, transportation, and manufacturing processes. Carbon credit was calculated for wood product storage, substitution, and internal mill recycling of wood losses for bioenergy.

Products were divided into sawtimber, pulpwood, and wood and paper products using published coefficients (44). Long-term and short-term products were assumed to decay at 2% and 10% per year, respectively (45). For product substitution, we focused on manufacturing for long-term structures (building life span >30 y). Because it is not clear when product substitution started in the Pacific Northwest, we evaluated it starting in 1970 since use of concrete and steel for housing was uncommon before 1965. The displacement value for product substitution was assumed to be 2.1 Mg fossil C/Mg C wood use in long-term structures (46), and although it likely fluctuates over time, we assumed it was constant. We accounted for losses in product substitution associated with building replacement (33) using a loss rate of 2% per year (33), but ignored leakage related to fossil C use by other sectors, which may result in more substitution benefit than will actually occur.

The general assumption for modern buildings, including cross-laminate timber, is they will outlive their usefulness and be replaced in about 30 y (7). By 2035, ~75% of buildings in the United States will be replaced or renovated, based on new construction, demolition, and renovation trends, resulting in threefold as many buildings as there are now [2005 baseline (31, 32)]. The loss of

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the PSS is therefore PSS multiplied by the proportion of buildings lost per year (2% per year).

To compare the NECB equivalence to emissions, we calculated forest sector and energy sector emissions separately. Energy sector emissions ["in-boundary" state-quantified emissions by the Oregon Global Warming Commission (14)] include those from transportation, residential and commercial buildings, industry, and agriculture. The forest sector emissions are cradle-to-grave annual carbon emissions from harvest and product emissions, transportation, and utility fuels (Table S3). Forest sector utility fuels were subtracted from energy sector emissions to avoid double counting.

Uncertainty Estimates. For the observation-based analysis, Monte Carlo simulations were used to conduct an uncertainty analysis with the mean and SDs for NPP and Rh calculated using several approaches (36) (*SI Materials and Methods*). Uncertainty in NECB was calculated as the combined uncertainty of NEP, fire emissions (10%), harvest emissions (7%), and land cover estimates (10%) using the propagation of error approach. Uncertainty in CLM4.5 model simulations and LCA were quantified by combining the uncertainty in the observations used to evaluate the model, the uncertainty in input datasets (e.g., remote sensing), and the uncertainty in the LCA coefficients (41).

Model input data for physiological parameters and model evaluation data on stocks and fluxes are available online (37).

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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

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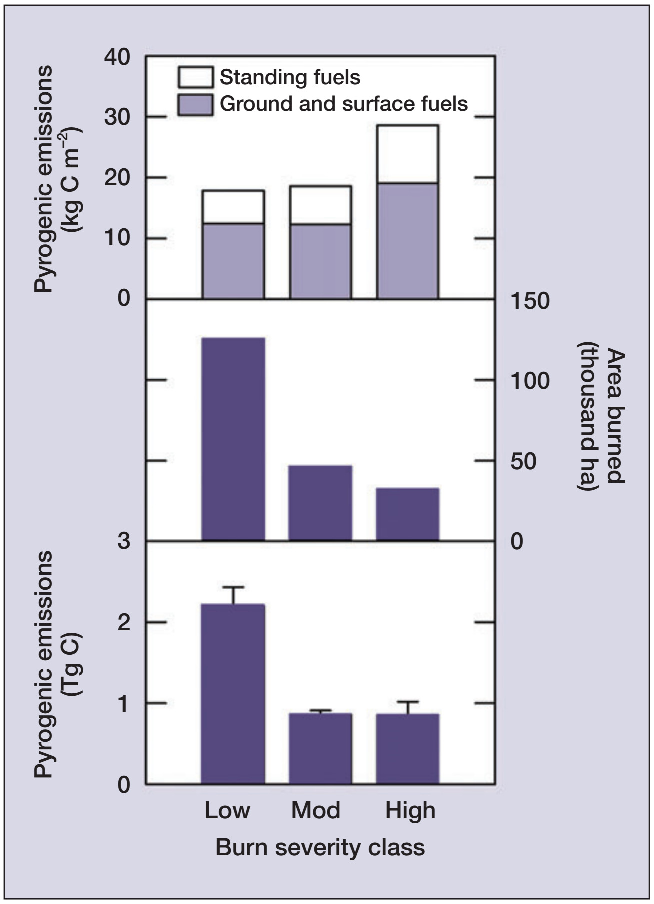

Figure 1. *Sources of pyrogenic emissions across the 2002 Biscuit Fire in southwestern Oregon and northern California. Because most emissions arise from the combustion of ground and surface fuels, pyrogenic emissions from high-severity fires were only one-third higher than those in low-severity fires. 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 remov-

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ing 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.

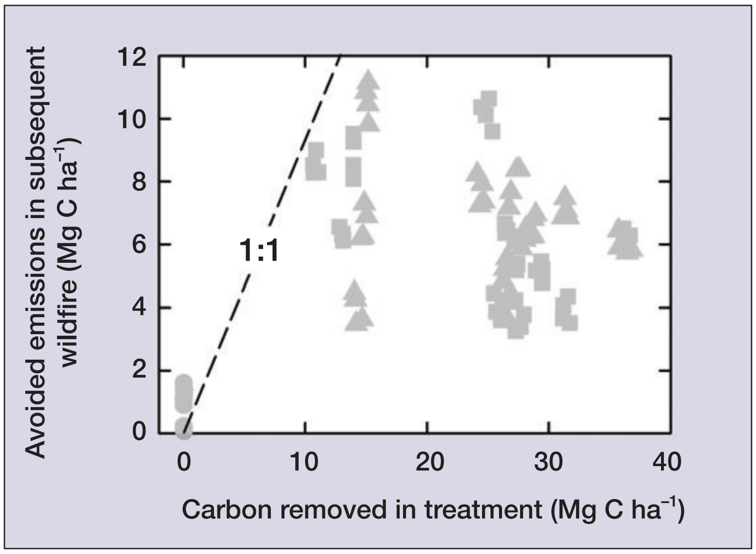

Figure 2. *Simulated effectiveness of various fuel-reduction treatments in reducing future wildfire combustion in a ponderosa pine (*Pinus ponderosa*) forest. In general, protecting one unit of C from wildfire combustion came at the cost of removing approximately three units of C in treatment. At the 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-

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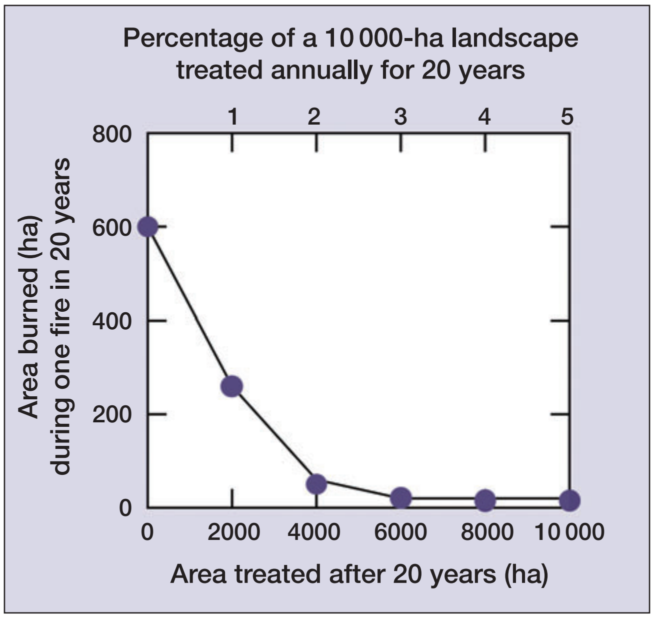

Figure 3. *Simulated effects of strategically placed fuel treatments on wildfire spread across a fire-prone ponderosa and lodgepole pine (*Pinus contorta*) landscape in western Montana. Treating only 1% of the forest annually for 20 years reduced the area impacted by a single large wildfire (assumed 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,

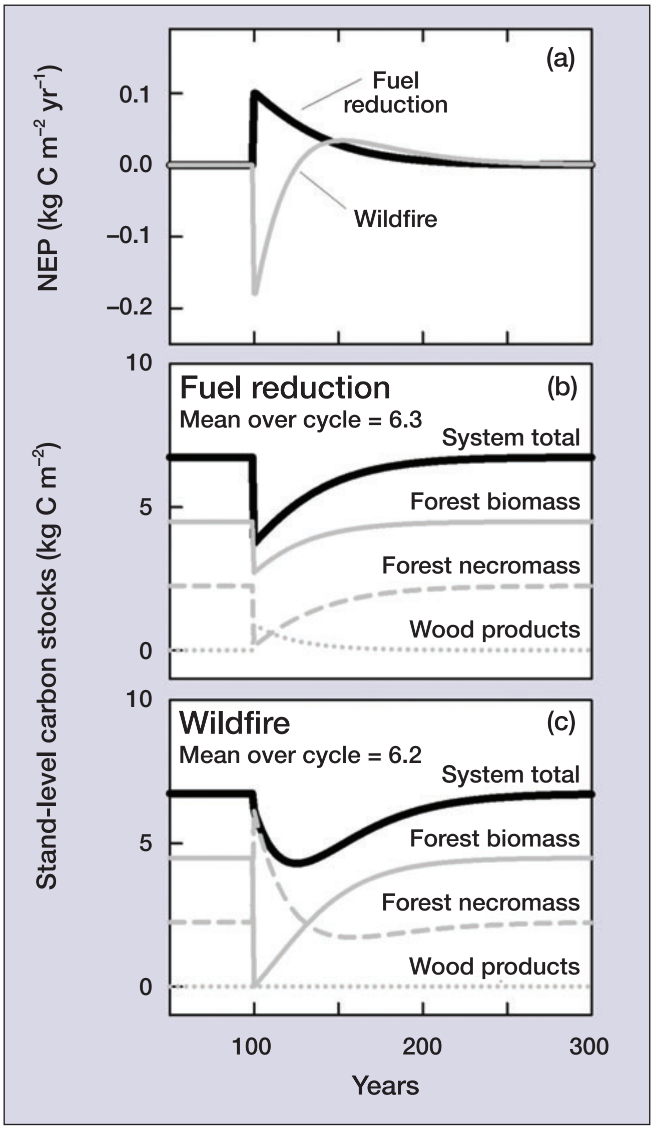

Figure 4. *(a) Simulated net ecosystem production and (b–c) C stocks throughout an entire disturbance interval, initiated by either wildfire or fuel-reduction treatment. Unlike the stand subject to fuel reduction via thinning, the combination of low biomass and high necromass after wildfire functions 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.*

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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.

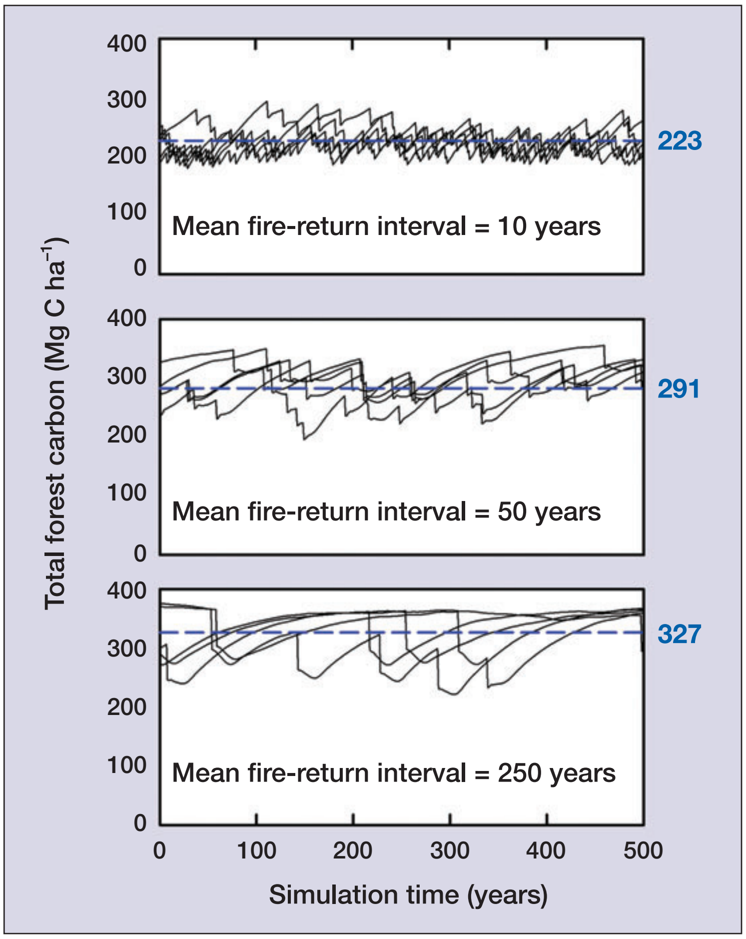

Figure 5. *Total forest C stores simulated for a ponderosa pine forest in the eastern Cascades of Oregon experiencing three different hypothetical fire regimes. Black lines depict the C stores of five individual stands subject to random fire events. Blue lines mark the 500-year average of all five stands. 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.

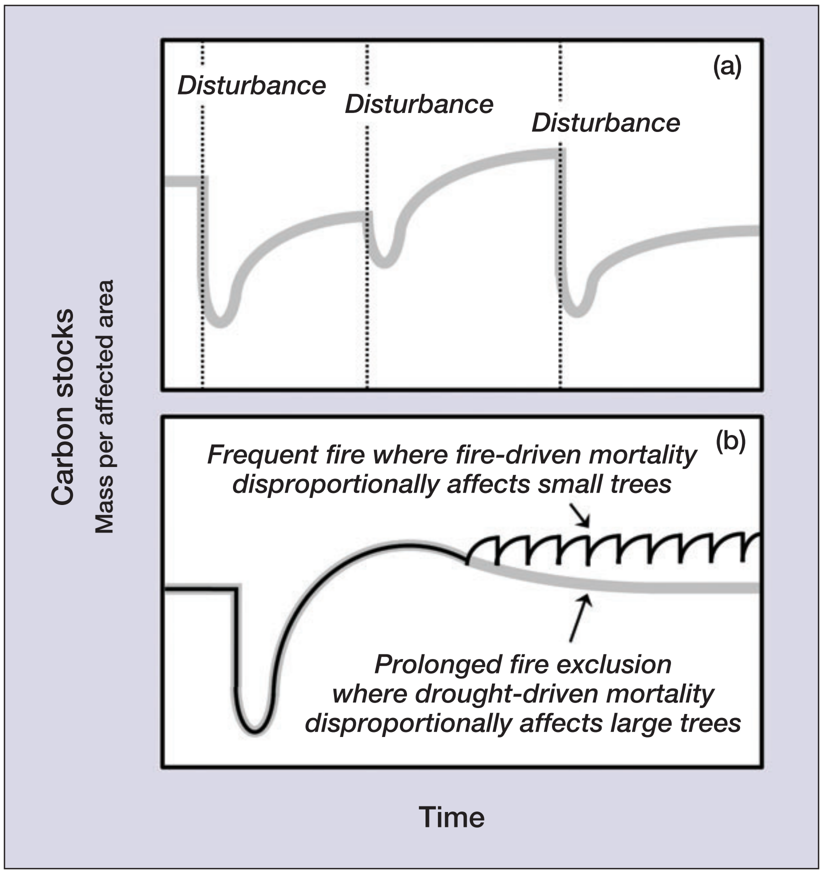

Figure 6. *Hypothetical examples of how disturbances could initiate alternate steady-state C stocks. (a) Illustration of what C stocks might look like if long-term successional trajectories were contingent more on seed availability at the time of fire than they were on fixed site conditions, as suggested 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 conduct-

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ing 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-sup-

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pressed 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).

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

	Fraction of forest area burned annually			
Forest type (ecoregion)	*Any severity*	*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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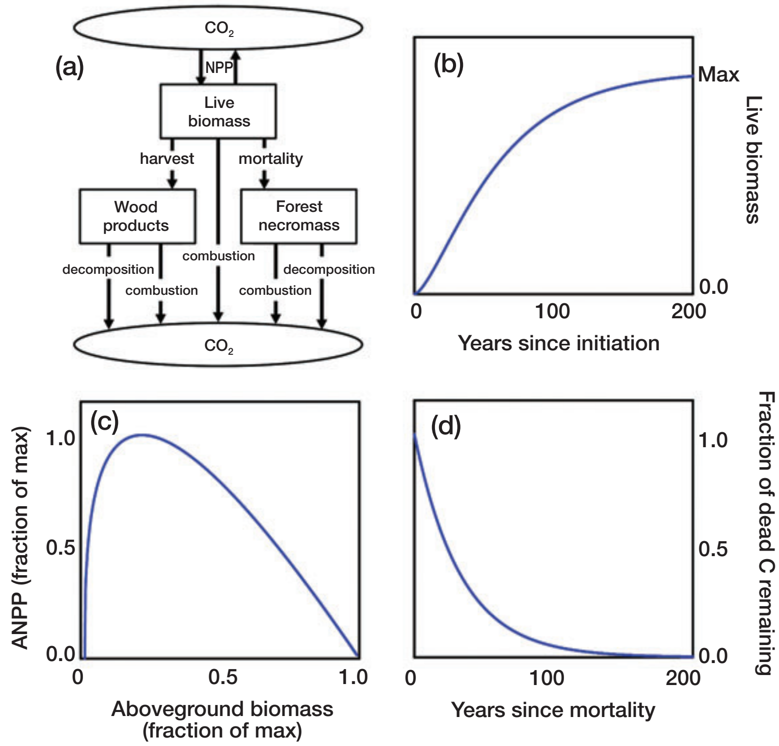

WebFigure 1. *(a) Structure and (c and d) dynamic functions behind the forest carbon model used to produce Figure 4. (b) Live biomass is assumed to aggrade over time according to a Chapman-Richards function* $y_1 = a*(1 - \exp[-b_1x_1])^c$*, the derivative of which,* $y_2 = c*b_1*y_1*(1 - \exp[\ln\{y_1/a\}/c])/\exp(\ln[y_1/a]/c)$*, 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.*

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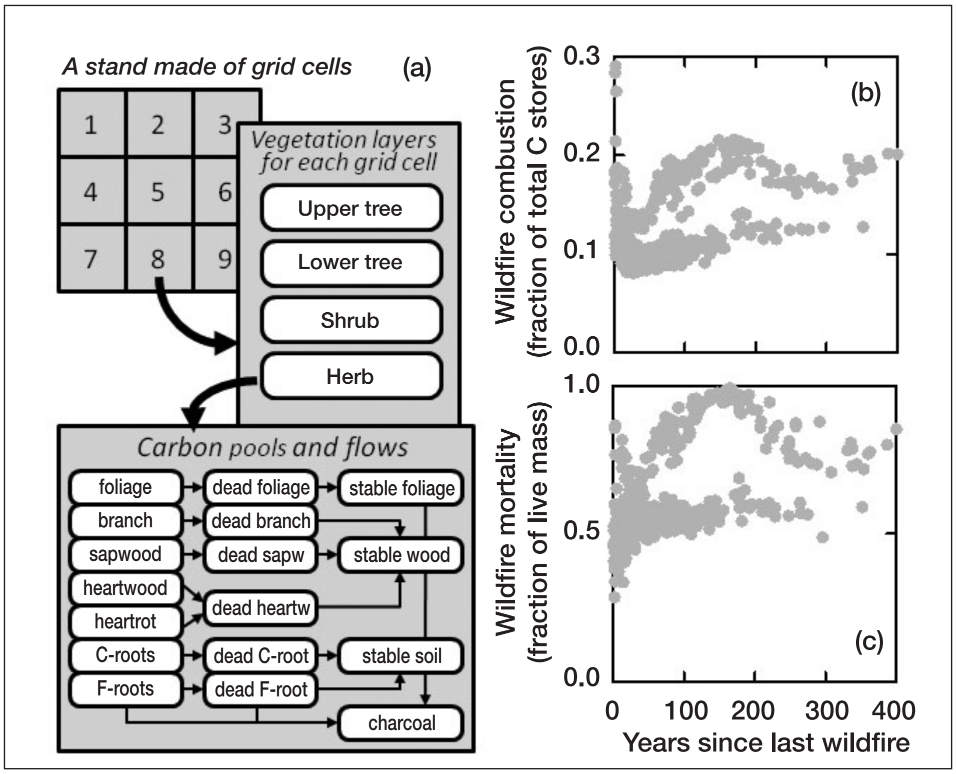

WebFigure 2. *(a) Structure and (b and c) disturbance responses behind STANDCARB, the forest carbon model used to produce Figure 5. STANDCARB simulates the accumulation of C over succession in mixed-species and mixed-age forest stands at annual time steps. The growth of vegetation and subsequent transfer 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.*

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