June 15, 2018

Comments submitted online via:
https://www.arb.ca.gov/lispub/comm2/bcsubform.php?listname=nat-workinglands-ws&comm_period=1

Shelby Livingston
Air Resources Board
1001 I Street
Sacramento, CA 95814

RE: California's Natural and Working Lands Implementation Plan Concept Paper

Dear Ms. Livingston,

Thank you for the opportunity to comment on the Natural and Working Lands Implementation Plan Concept Paper ("NWL Concept Paper"). These comments are submitted on behalf of the Center for Biological Diversity ("Center"). The Center appreciates the responsiveness of the California Air Resources Board ("CARB") and California Natural Resources Agency ("CNRA") to some of our concerns regarding the NWL Implementation Plan and CALAND model. However, many concerns raised in our comment letters and workshop participation remain unaddressed.

As detailed below, we recommend the following: (1) Allow for adequate public review of the NWL Emissions Inventory, CALAND Model, and Draft NWL Implementation Plan and 2030 Goal; (2) Clarify and provide a sound scientific basis for the 2030 Emissions Reduction Goal; (3) Focus on forest management measures that support carbon storage while protecting forest ecosystem function and values; (4) Conduct CEQA review for the NWL Implementation Plan.

**I. Allow For Adequate Public Review of the NWL Emissions Inventory, CALAND Model, and Draft NWL Implementation Plan and 2030 Goal**

The current timeline for public release and review of the draft NWL Emissions Inventory, CALAND model version 3, and draft NWL Implementation Plan and 2030 Goal does not appear to allow for adequate public review and input. The NWL Concept Paper states that the final 2030 goal for sequestration and emissions will be informed by CARB's NWL Emission Inventory data and the CALAND and COMET-Planner model outputs on expected GHG emissions from management activities. In addition, monitoring progress in reducing GHG emissions will be based on the CALAND and COMET-Planner tools. Because of the central role of the CALAND model and Inventory in informing the NWL Implementation Plan and Goal, public stakeholders must be given adequate opportunity to assess and provide input on the Inventory and CALAND model, including technical documentation, inputs and outputs, in advance of the comment period for the draft NWL Implementation Plan and Goal.

At present, the NWL Concept Paper states that CARB will publish a draft updated NWL Inventory for public review in "fall 2018." The timeline for release of Version 3 of the CALAND model is unclear. The Concept Paper states that Version 3 will be completed in July 2018, but there is no indication of when the model, outputs, and technical documentation specifying the assumptions, data sources, and inputs are to be released for public review. The current timeline does not appear to provide sufficient time for review of the Inventory and CALAND model before the draft 2030 goal and NWL Implementation Plan is released, which is currently schedule for "summer 2018." CARB must also ensure that there is a transparent process on how public input will be incorporated into the final NWL 2030 Goal (to be released September 2018) and final Implementation Plan (to be released November 2018).

We are particularly concerned because a year and a half has passed since the public was able to see any results from the CALAND model, and those December 2016 results were preliminary. LBNL and CARB have still not released the detailed technical documentation needed to evaluate the model – or its inputs and assumptions – and whether this tool is adequate for fulfilling its intended purpose for informing the selection of management interventions under the NWL Implementation Plan. As detailed in our October 2017 comment letter, it does not appear that the CALAND model is capable of providing the robust carbon accounting that state agencies will be relying on it for – that is, providing robust estimates of GHG emissions associated with a set of management measures that will be used to justify recommending, funding, and pursuing those strategies. The model is too coarse, its input data too limited, and its outputs too broadly aggregated to provide a rational evidentiary basis for pursuing any particular management strategy.

**II. Clarify and Provide a Sound Scientific Basis for the 2030 Emissions Reduction Goal**

The NWL Concept Paper sets a preliminary goal to reduce GHG emissions "by at least 15 - 20 MMT CO2e by 2030." It is unclear how this number was selected and what baseline scenario it will be compared with. As we have previously recommended, we suggest that CARB and CNRA build a robust carbon accounting model that informs the reductions that are possible based on a comprehensive set of management measures, and derive the 2030 goal from that—not set an arbitrary goal, identify a preconceived set of management strategies, and build a model with the apparent purpose of making it appear that those strategies will achieve the goal.

**III. Focus on Forest Management Measures that Support Carbon Sequestration While Protecting Forest Ecosystem Function and Values**

The forest management measures presented in the NWL Concept Paper continue to focus on logging and thinning activities and forest bioenergy projects which the scientific literature indicates will increase carbon emissions and decrease ecosystem function. We recommend that the NWL Implementation Plan instead focus on management activities that support forest carbon storage while protecting biodiversity and ecosystem function.

The proposed forest management measures focus on cutting trees and clearing understory vegetation as "fuels reduction," paired with increased forest biomass utilization, which have been shown by scientific studies to increase carbon emissions for decades to centuries. Harvest of live trees from the forest not only reduces current standing carbon stocks, but also reduces the forest's future rate of carbon sequestration, and its future carbon storage capacity, by removing trees that otherwise would have continued to grow and remove $CO_2$ from the atmosphere. Numerous studies indicate that thinning forests to reduce fire activity decreases forest carbon stocks and results in increased carbon emissions to the atmosphere that can persist for decades. Forest biomass utilization for bioenergy is completely unwarranted: forest biomass combustion is extremely carbon-intensive and detrimental from a climate and carbon perspective. These practices are also harmful to biodiversity and wildlife habitat, air and water quality, public health, and forest connectivity. Rather than promoting logging, bioenergy, and further loss of carbon from forest ecosystems, forest management should prioritize the opportunities to keep forest biomass circulating within forest ecosystems.

We appreciate the addition of land management measures – land protection and reforestation -- that will promote carbon sequestration while protecting ecological function on forestlands, as well as less intensive forest management. The importance of these practices was highlighted by a recent study by Law et al. (2018) in Oregon forests which found that lengthened harvest cycles on private lands and restricting harvest on public lands were the most effective management measures for increasing net ecosystem carbon balance, followed by reforestation and afforestation.[1] In contrast, using forest harvest residue for bioenergy production increased cumulative net emissions compared to leaving residues in the forest to slowly decompose.[2] We recommend that the "less intensive forest management" category include clearly defined alternatives for reduced levels of harvest (including understory clearing and logging) with "no harvest" alternatives, longer harvest rotations, avoidance of clearcutting on private lands and other intensive forms of tree removal, and the retention of larger trees, all of which allow forests to accumulate more carbon. In the "understory clearing" category, the Plan should model scenarios where all understory biomass is scattered as dead debris.

The NWL Implementation Plan should also add important forest management measures that support forest carbon and ecosystem function, specifically managed wildland fire in which land managers make a decision to allow lightning-caused fires to burn in order to enhance natural heterogeneity, increase forest health and resilience, and benefit wildlife. Wildfire is a natural and necessary component of California's forest ecosystems, with many critical functions for supporting structural heterogeneity, biodiversity, nutrient cycling, and ecosystem resilience. Wildfire levels in most forest ecosystems are well below historical levels due to a long history of fire suppression, and it is widely recognized that restoring wildfire is important for forest health. Studies have demonstrated that "fuels reduction" is unnecessary, expensive, carbon-emitting, and is not needed as a precondition to the restoration of mixed-severity fire through managed wildland fire and prescribed mixed-severity fire. In short, the restoration of natural fire regimes

[1] Law, B.E. et al., Land use strategies to mitigate climate change in carbon dense temperate forests, 115 PNAS 3663-3668 (2018)
[2] *Id.*

to California's forests should be a core forest management measure in the NWL Implementation Plan.

Finally, many of the justifications provided for logging and thinning management measures are predicated on false premises and not grounded in the scientific literature. Statements in the Concept Paper that "disturbances such as severe wildfire… can cause these landscapes to emit more carbon dioxide than they store" and that forests need "fuels reduction" for health and carbon benefits are inaccurate and misleading. For example, a recent study by Berner et al. (2017) highlighted that logging is the highest source of tree mortality in the western United States, including California -- much more than fires, beetles, or drought.[3] The statement that "biomass" can advance statewide objectives for renewable energy is inconsistent with evidence that burning woody biomass is more carbon-intensive at the smokestack than coal, and emits pollutants harmful to public health. In the "understory clearing" category, the statement that removing the forest understory "supports forest health" is not consistent with the scientific literature. Likewise, in the "understory clearing" and "partial cut" categories, the claim that these practices "enhance net carbon accumulation and reduces the fraction of high-severity wildfire for 20 years without additional treatment" is not consistent with the scientific literature, and many studies indicate that fuel treatment effects last for much less than 20 years. These statements should be omitted or corrected.

## IV. CEQA Compliance is Required for the NWL Implementation Plan

There has been no indication thus far that CARB intends to comply with the California Environmental Quality Act ("CEQA") in connection with NWL Implementation Plan. The Plan is clearly a discretionary action that will cause (and indeed is *intended* to cause) both direct and reasonably foreseeable indirect changes in the environment. (Pub. Resources Code § 21065.) And to the extent any aspect of the Plan will be implemented by CARB or any other agency (or used to direct funding to any particular types of projects) in a manner that, as a practical matter, commits any agency to a particular course of management action, the Plan is a project that will be approved, supported, and/or carried out by public agencies. (*Ibid.*; see also CEQA Guidelines § 15352.). CEQA compliance is therefore mandatory.

CEQA compliance will be critical in helping stakeholders, the public, and decision makers across state government understand the full implications of the Plan. Management interventions under the Implementation Plan could have profound effects not only on climate and emissions, but on California's landscape and environment. Managing natural and working lands solely for carbon can cause other impacts, and making programmatic decisions solely on the basis of meeting a carbon goal can undermine assessment of alternatives and efforts to mitigate environmental damage. In concrete terms, all of the interventions under discussion in the Plan (including agriculture, forest management, and bioenergy) have a range of environmental consequences far beyond carbon emissions and sequestration. Those consequences must be disclosed and considered in evaluating alternatives for possible funding.

[3] Berner, L.T. et al., Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003-2012), 12 Environmental Research Letters 065005 (2017)

Thank you for your consideration of these comments. We look forward to reviewing the draft NWL Implementation Plan and 2030 Goal, CALAND technical documentation and outputs, and NWL Emissions Inventory. We are submitting pdfs of the cited references with these comments. Please contact me if you have any questions or would like to discuss these comments.

Sincerely,

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# Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003–2012)

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LETTER

# Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003–2012)

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Supplementary material for this article is available online

## Abstract

High temperatures and severe drought contributed to extensive tree mortality from fires and bark beetles during the 2000s in parts of the western continental United States. Several states in this region have greenhouse gas (GHG) emission targets and would benefit from information on the amount of carbon stored in tree biomass killed by disturbance. We quantified mean annual tree mortality from fires, bark beetles, and timber harvest from 2003–2012 for each state in this region. We estimated tree mortality from fires and beetles using tree aboveground carbon (AGC) stock and disturbance data sets derived largely from remote sensing. We quantified tree mortality from harvest using data from US Forest Service reports. In both cases, we used Monte Carlo analyses to track uncertainty associated with parameter error and temporal variability. Regional tree mortality from harvest, beetles, and fires ($MORT_{H+B+F}$) together averaged 45.8 ± 16.0 Tg AGC $yr^{-1}$ (±95% confidence interval), indicating a mortality rate of 1.10 ± 0.38% $yr^{-1}$. Harvest accounted for the largest percentage of $MORT_{H+B+F}$ (∼50%), followed by beetles (∼32%), and fires (∼18%). Tree mortality from harvest was concentrated in Washington and Oregon, where harvest accounted for ∼80% of $MORT_{H+B+F}$ in each state. Tree mortality from beetles occurred widely at low levels across the region, yet beetles had pronounced impacts in Colorado and Montana, where they accounted for ∼80% of $MORT_{H+B+F}$. Tree mortality from fires was highest in California, though fires accounted for the largest percentage of $MORT_{H+B+F}$ in Arizona and New Mexico (∼50%). Drought and human activities shaped regional variation in tree mortality, highlighting opportunities and challenges to managing GHG emissions from forests. Rising temperatures and greater risk of drought will likely increase tree mortality from fires and bark beetles during coming decades in this region. Thus, sustained monitoring and mapping of tree mortality is necessary to inform forest and GHG management.

## 1. Introduction

Forests help regulate Earth's climate in part by sequestering carbon from the atmosphere (Bonan 2008, Pan *et al* 2011a, Anderson-Teixeira *et al* 2012), yet tree mortality from disturbance accelerates carbon transfer from these ecosystems back to the atmosphere (Kurz *et al* 2008, Baccini *et al* 2012, Brinck *et al* 2017). Forests globally store a similar amount of carbon as the atmosphere (Pan *et al* 2011a, Houghton 2013), with much of the carbon (∼42%) held in the biomass of living trees (Pan *et al* 2011a). Disturbances such as forest fires, insect outbreaks, and timber harvest can kill trees over large areas each year (Goetz *et al* 2012, Meddens *et al* 2012, Kautz *et al* 2016, Williams *et al* 2016) and thus contribute to increased regional carbon emissions as tree biomass subsequently decomposes or is rendered into wood products that have finite longevity (Harmon *et al* 1990, Harmon *et al* 2011, Ghimire *et al* 2015). Carbon emissions from forest disturbance can challenge efforts to meet greenhouse gas (GHG) emission targets (Gonzalez *et al* 2015), but are also highly uncertain in many parts of the world (Pacala *et al* 2010). Both the growing global demand for wood products (FAO 2017) and the increase in forest disturbance due to ongoing climatic change (Allen *et al* 2010, Williams *et al* 2012, Kautz *et al* 2016) underscore the need to better understand carbon implications of tree mortality from disturbance.

Forest disturbance by fires and insects increased during recent decades in the western contiguous United States as the regional climate became warmer and more arid (Westerling *et al* 2006, Williams *et al* 2012, Dennison *et al* 2014, Hicke *et al* 2015, Abatzoglou and Williams 2016). Regional mean annual air temperature increased 0.8 °C–1.1 °C from 1895 to 2011, with most of the warming having occurred in recent decades (Mote *et al* 2014, Walsh *et al* 2014). Higher temperatures contributed to higher atmospheric vapor pressure deficits (Abatzoglou and Williams 2016), reduced mountain snowpack (Mote *et al* 2005), and more frequent and severe drought (McCabe *et al* 2004, Diffenbaugh *et al* 2015). For instance, the western US recently experienced its most severe drought (2000–2004) in the past 800 years (Schwalm *et al* 2012), with hot and dry conditions then prevailing through the 2000s (Diffenbaugh *et al* 2015, Abatzoglou and Williams 2016). These conditions contributed to extensive forest disturbance by fires and bark beetles relative to recent decades (Williams *et al* 2012, Creeden *et al* 2014, Abatzoglou and Williams 2016). High temperatures and drought increase regional forest fire occurrence (Littell *et al* 2016) and the likelihood of post-fire tree mortality (i.e. increased fire severity; van Mantgem *et al* 2013), while also increasing beetle populations and the vulnerability of drought-stricken trees to beetle attack (Raffa *et al* 2008, Creeden *et al* 2014, Hart *et al* 2014). Projections indicate that regional temperatures could rise another ~3.8 °C–5.5 °C by the end of the 21st century and that much of the region, particularly the Southwest, could become increasingly arid and prone to drought under a high GHG scenario (RCP 8.5; Kunkel *et al* 2013, Walsh *et al* 2014, Cook *et al* 2015). These changes in regional climate could further accelerate tree mortality (Adams *et al* 2009, Allen *et al* 2015) and increase carbon emissions from forest ecosystems (Spracklen *et al* 2009, Jiang *et al* 2013, McDowell *et al* 2015).

Several states in the western US have GHG reduction targets (e.g. Oregon, California) and would thus benefit from information on the magnitude and primary causes of recent tree mortality. Prior studies have shown that fires, bark beetles, and timber harvest are important causes of tree mortality in this region (Masek *et al* 2011, Meddens *et al* 2012, Hicke *et al* 2015). In this study, we asked 'What was the magnitude and relative contribution of mean annual tree mortality from fires, bark beetles, and timber harvest from 2003–2012 both regionally and among the 11 western states?' Tree mortality can be quantified over large areas in terms of carbon using remote sensing estimates of tree aboveground biomass (AGB) together with information on the carbon content of AGB, as well as disturbance extent and severity (Baccini *et al* 2012, Hicke *et al* 2013). Here we quantified tree mortality as the amount of *carbon stored in tree AGB* (AGC) killed by disturbance (e.g. Mg AGC ha $yr^{-1}$ or Tg AGC state $yr^{-1}$). Specifically, we developed spatially explicit estimates of annual tree mortality from fires and bark beetles across regional forestland building off of a remote sensing framework from an earlier study (Hicke *et al* 2013). In addition to the remote sensing analysis, we estimated mean annual tree mortality from timber harvest for each state using harvest statistics from the US Forest Service (USFS; Smith *et al* 2009, Oswalt *et al* 2014). The USFS recently recommended that metrics related to fire and insect effects be used to track national climate change impacts (Heath *et al* 2015), further underscoring the importance of quantifying the magnitude and regional variation in tree mortality from these types of disturbance.

## 2. Methods

### 2.1. Geospatial data sets and preprocessing

#### *2.1.1. General information*

We quantified tree mortality from fires ($MORT_{fire}$, Mg AGC ha $yr^{-1}$) and bark beetles ($MORT_{beetle}$, Mg AGC ha $yr^{-1}$) from 2003–2012 across 802 575 km² of forestland located in the western US (figure 1, 31.3°N–49.0°N, 102.0°W–124.5°W). Forestlands were included if consistently mapped as forest by three separate tree AGB data sets (described below). We conducted the analysis on a 1 km resolution grid in an Albers Equal Area projection, with the resolution chosen to match the bark beetle data set (Meddens *et al* 2012). We analyzed and visualized the data using ArcGIS 10 (ESRI, Redlands, CA) and *R* statistical software (version 3.2; R Core Team 2015), along with the *R* packages *raster* (Hijmans and van Etten 2013) and *dplyr* (Wickham and Francois 2015). The data preprocessing work flow described in the following section is illustrated in figure 2.

#### *2.1.2. Tree carbon stocks*

We estimated tree AGC using three tree AGB geospatial data sets together with information on the fraction of dry biomass that is carbon ($f_{carbon}$). The tree AGB data sets were produced by spatially interpolating USFS inventory measurements (FIA)

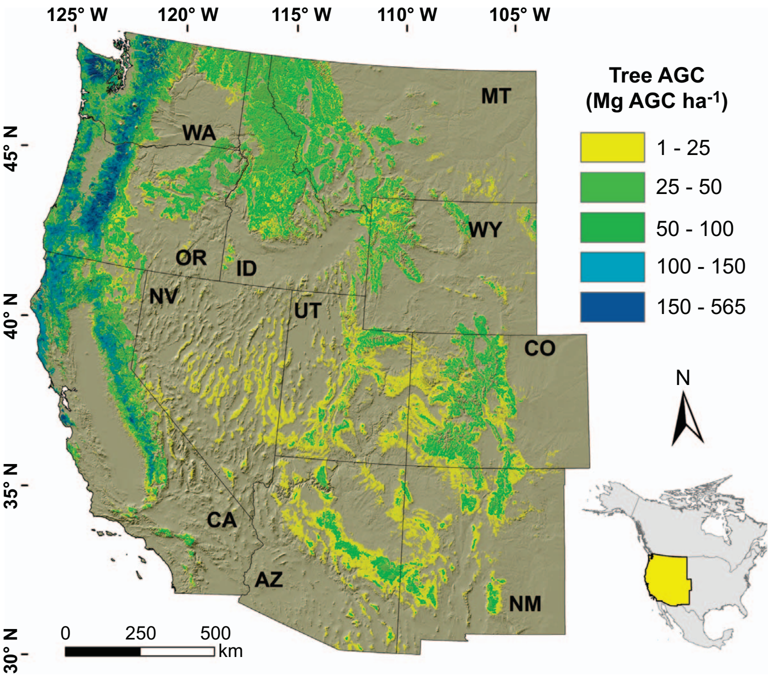


**Figure 1.** Study area extent, state boundaries, and live tree aboveground carbon (AGC) stocks on forestland in the western United States. Tree AGC stocks reflect conditions during the 2000s and are based on the average of three recent tree aboveground biomass data sets (Blackard *et al* 2008, Kellndorfer *et al* 2012, Wilson *et al* 2013) together with information on carbon content of dry biomass (Lamlom and Savidge 2003).

using satellite and geophysical data sets in conjunction with machine learning algorithms (Blackard *et al* 2008, Kellndorfer *et al* 2012, Wilson *et al* 2013). Wilson *et al* (2013) provided estimates of tree AGC assuming that $f_{\text{carbon}}$ was 50%; we converted these estimates back to tree AGB by multiplying by 2. We acquired each data set at 250 m spatial resolution, reprojected it to an Albers Equal Area projection, and then applied a common mask that identified pixels consistently classified as forest among all data sets. Next, we aggregated from 250 m to 1 km resolution by computing total tree AGB in each 1 km pixel (i.e. Mg AGB pixel$^{-1}$).

We quantify tree mortality from 2003–2012 and required estimates of tree AGB prior to disturbance. Two of these data sets represented tree AGB circa 2000 (Blackard *et al* 2008, Kellndorfer *et al* 2012), whereas the third depicted tree AGB circa 2003–2008 (Wilson *et al* 2013). Therefore we needed to estimate pre-disturbance tree AGB in pixels that were disturbed during these six years in the Wilson *et al* (2013) data set. We followed an existing approach (Hicke *et al* 2013) that involved comparing tree AGB in fire and beetles disturbed pixels against the average tree AGB of undisturbed pixels from the same forest type (Ruefenacht *et al* 2008) and ecoregion (Omernik 1987). If tree AGB in the disturbed pixel was less than the average tree AGB of undisturbed pixels, then we set tree AGB in the disturbed pixel to this average, but otherwise left the disturbed pixel unaltered. After implementing this correction, we then computed the average and standard error (SE) of tree AGB for each pixel over the three data sets ($\overline{\text{AGB}}$, $\text{SE}_{\text{AGB}}$). Lastly, we estimated the average and SE of $f_{\text{carbon}}$ ($\overline{f_{\text{carbon}}}$, $\text{SE}_{\text{carbon}}$) for hardwood (angiosperm, 48.49 ± 0.42% C, $n = 8$) and softwood (gymnosperm, 50.87 ± 0.63% C, $n = 11$) tree species found in this region (Lamlom and Savidge 2003).

### *2.1.3. Forest fires*

We used annual maps of fire severity from 2003–2012 produced by the Monitoring Trends in Burn Severity (MTBS) project (Eidenshink *et al* 2007). These maps were derived from 30 m resolution Landsat images acquired before and after fires larger than ~405 ha in the western US. Fire severity was mapped using the differenced Normalized Burn Ratio (dNBR) and then each pixel was classified as one of five thematic severity classes. We focused on pixels that burned at low-, moderate-, or high-severity and excluded those that were unburned or showed post-fire greening. We identified fire in forests by generating a 30 m resolution forest mask based on forestland in either of two national land cover maps (Homer *et al* 2007, Rollins 2009). We then computed the annual fraction of forest area that burned at each severity ($A_s$) in each 1 km pixel over the decade. Lastly, we incorporated field measurements of the fraction of tree biomass killed at each severity ($f_{\text{fire}}$). We computed the mean and SE of $f_{\text{fire}}$ ($\overline{f}_{\text{fire}}$, $\text{SE}_{\text{fire}}$) for forests dominated by hardwood and softwood tree species based on 116 estimates of $f_{\text{fire}}$ synthesized from 29 field studies conducted in the western United States (table S1 available at stacks.iop.org/ERL/12/065005/mmedia, Ghimire *et al* 2012).

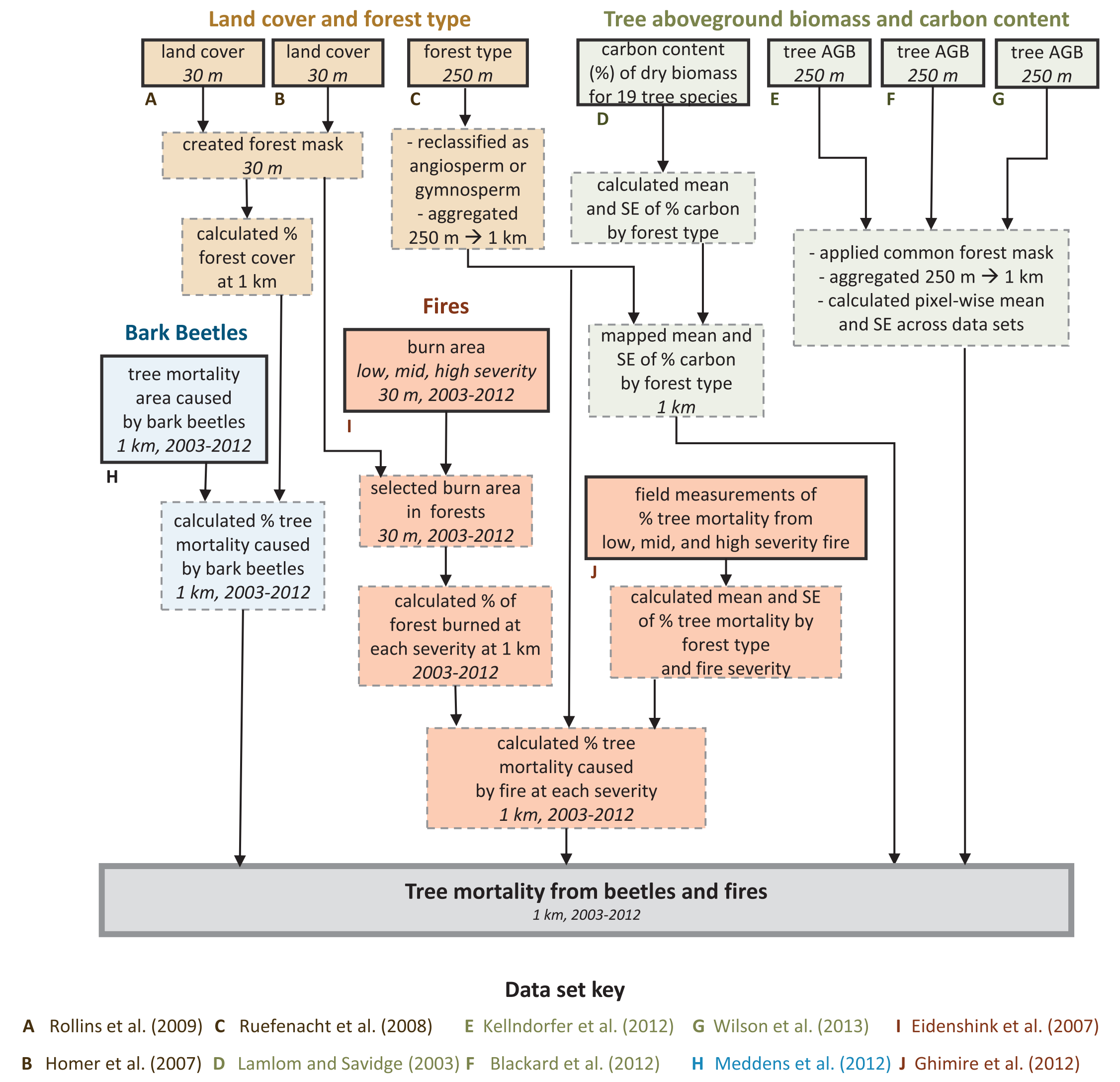


**Figure 2.** Workflow illustrating data sets (solid boxes) and processing steps (dashed boxes) for estimating tree mortality due to bark beetles and fires from 2003–2012 across the western United States (thick solid box). Data citations, spatial resolutions, and temporal ranges are shown where applicable.

### *2.1.4. Bark beetles*

We used annual maps depicting the areal extent of tree canopy mortality due to 12 bark beetle species that were produced from aerial surveys, forest inventory measurements, and high-resolution satellite imagery (Meddens *et al* 2012). The USFS National Forest Health Monitoring program conducts aerial detection surveys (ADS) that provide a coarse snapshot of insect and other forest disturbance at a landscape scale (Johnson and Wittwer 2008). Trained observers conduct surveys from fixed-wing aircraft that involve sketch-mapping the extent of areas affected by insect outbreaks and then visually estimating the number of dead trees within affected areas. Meddens *et al* (2012) used the ADS observations to estimate annual canopy mortality area due to bark beetles across the western US from 1997–2010, with estimates then extended through 2012 (Hicke *et al* 2015).

Canopy mortality area was estimated at 1 km resolution based on the number of recently killed trees and the average canopy area of each tree species. The ADS observations underestimated the number of trees killed by bark beetles in comparison with field observations and QuickBird (2.4 m resolution) satellite imagery from Colorado, Idaho, and New Mexico. This led to the development of adjustment factors for several forest types based on the ratio of dead trees mapped from satellite to ADS tree kill counts. These adjustment factors were then used to modify ADS tree kill counts, resulting in high and middle (most realistic) estimates as well as low estimates based on unaltered ADS numbers (more details in: Meddens *et al* 2012, Hicke *et al* 2015). The data set includes bark beetle species that cumulatively killed at least 100 000 trees across the domain from 1997–2010. Specifically, it includes (from greatest to least mortality area) mountain pine beetle (*Dendroctonus ponderosae* Hopkins), piñon ips (*Ips confusus* (LeConte)), Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopkins), western balsam bark beetle (*Dryocoetes confuses* (Swaine)), and fir engraver (*Scolytus ventralis* LeConte), as well as seven additional beetle species that killed fewer trees. We masked these data to our domain and then summed canopy mortality area across bark beetle species for each 1 km pixel in a given year (i.e. a voxel) from 2003–2012. We then calculated annual tree canopy mortality fraction ($f_{\text{beetle}}$) for each voxel by dividing canopy mortality area by forest cover fraction. We incorporated the lower, middle, and upper estimates of $f_{\text{beetle}}$ into our analysis.

### 2.2. Tree mortality from bark beetles and fires

We developed spatially explicit estimates of annual $\text{MORT}_{\text{beetle}}$ and $\text{MORT}_{\text{fire}}$ from 2003–2012 and quantified uncertainty in these estimates using a Monte Carlo approach (e.g. Harmon *et al* 2007, Gonzalez *et al* 2015). The Monte Carlo involved generating 100 realizations of annual tree mortality in each voxel, where each realization iteratively varied tree $\overline{\text{AGC}}$, $\bar{f}_{\text{carbon}}$, $f_{\text{beetle}}$, and $\bar{f}_{\text{fire}}$ based on uncertainty in each term. We assumed that tree AGC within a pixel only changed due to disturbance (i.e. no tree growth or recruitment), which potentially caused us to slightly underestimate tree mortality. For instance, stand age averaged 97 ± 73 years (± 1SD) in our study area (Pan *et al* 2011b). Field measurements from the Western Cascades (Hudiburg *et al* 2009) and the Rocky Mountains showed that tree AGC could increase 6%–7% between stands that are 97 and 106 years old. This simplifying assumption made it so that cumulative tree mortality within a pixel could not exceed the initial tree AGC. For each realization $r$ we first computed tree AGC for pixel $p$ of year $t$ = 2003 as

$$\overline{\text{AGC}}_{r,p,t} = (\overline{\text{AGB}}_{p,t} + a_{\text{AGB},r} \times \text{SE}_{\text{AGB},p}) \times (\bar{f}_{\text{carbon}} + a_{\text{carbon},r} \times \text{SE}_{\text{carbon}}) \quad (1)$$

where $\bar{f}_{\text{carbon}}$ and $\text{SE}_{\text{carbon}}$ varied by forest type and $a_x$ was a random number from a normal distribution (mean = 0, SD = 1) that differed for each variable with each realization. The other variables are defined above. We then calculated $\text{MORT}_{\text{beetle}}$ as

$$\text{MORT}_{\text{beetle},r,p,t} = \overline{\text{AGC}}_{r,p,t} \times \hat{f}_{\text{beetle},p,t} \quad (2)$$

where $\hat{f}_{\text{beetle},p,t}$ was an estimate of $f_{\text{beetle}}$ drawn randomly from a triangular distribution defined uniquely for each voxel using the low, middle, and high estimates of $f_{\text{beetle}}$. Each distribution was fit using the *triangle* package in R (Carnell 2016). Next, we computed $\text{MORT}_{\text{fire}}$ as

$$\text{MORT}_{\text{fire},r,p,t} = (\overline{\text{AGC}}_{r,p,t} - \text{MORT}_{\text{beetle},r,p,t}) \times \sum_{s}^{\substack{\text{severity}\\\text{classes}}} [A_{p,t,s} \times (\bar{f}_{\text{fire},s} + a_{\text{fire},s,r} \times \text{SE}_{\text{fire},s})] \quad (3)$$

We then reduced tree AGC stocks at the start of year $t$ + 1 to account for mortality in year $t$, such that

$$\overline{\text{AGC}}_{r,p,t+1} = \overline{\text{AGC}}_{r,p,t} - (\text{MORT}_{\text{beetle},r,p,t} + \text{MORT}_{\text{fire},r,p,t}) \quad (4)$$

We repeated equations 2–4 annually through 2012 and then repeated the entire process 100 times.

We then computed the mean and standard deviation (SD) of these multiple estimates of tree mortality for each voxel, where the SD represented uncertainty due to parameter error (e.g. $\text{SE}_{\text{AGB}}$, $\text{SE}_{\text{fire}}$). Altogether, this process yielded spatially explicit estimates of annual tree mortality (and uncertainty) caused by fires and bark beetles from 2003–2012 across forestland in the western United States.

Following the geospatial analysis, we estimated mean annual tree mortality from fires ($\overline{\text{MORT}}_{\text{fire}}$) and bark beetles ($\overline{\text{MORT}}_{\text{beetle}}$) for each state from 2003–2012 and used a Monte Carlo analysis to propagate uncertainty in these estimates associated with parameter error and temporal variability. As illustrated for fires (identical for beetles), we first derived 1000 realizations $r$ of annual tree mortality in each state from $t$ = 2003 to 2012, as per

$$\text{MORT}_{\text{fire},\,r,\text{state},t} = \sum_{p=1}^{\text{N pixels}} (\text{MORT}_{\text{fire},p,t} + a_r \times \text{SD}_{\text{MORT fire},p,t}) \quad (5)$$

and then summarized (mean and SD) each set of realizations. We used these annual statewide summaries to then repeatedly ($N_r$ = 1000) estimate both the mean and SE of annual mortality from 2003–2012 for each state, where each realization randomly varied annual statewide mortality by its parameter error (i.e. SD computed above). We again summarized (mean and SD) each set of realizations for both statistics. This yielded an estimate of $\overline{\text{MORT}}_{\text{fire}}$ for each state and produced estimates of uncertainty in $\overline{\text{MORT}}_{\text{fire}}$ due to parameter error (i.e. SD of estimates of $\overline{\text{MORT}}_{\text{fire}}$) and temporal variability (i.e. mean of estimates of SE of annual mortality).

In the final step, we derived a 95% confidence interval (CI) around each estimate of $\overline{\text{MORT}}_{\text{fire}}$ that accounted for both parameter error and temporal variability. This involved repeatedly estimating $\overline{\text{MORT}}_{\text{fire}}$ for each state, where each realization randomly varied $\overline{\text{MORT}}_{\text{fire}}$ by uncertainty associated with both parameter error and temporal variability. We then computed the 95% CI (Gonzalez *et al* 2015) for each state, as per

$$95\% \quad \text{CI} = \frac{\overline{\text{MORT}}^{97.5}_{\text{fire}} - \overline{\text{MORT}}^{2.5}_{\text{fire}}}{2} \quad (6)$$

where $\overline{\text{MORT}}^{97.5}_{\text{fire}}$ and $\overline{\text{MORT}}^{2.5}_{\text{fire}}$ were the 97.5th and 2.5th percentiles of 1000 realizations of $\overline{\text{MORT}}_{\text{fire}}$. Overall, this approach yielded estimates of mean annual tree mortality (±95% CI) due to fires and bark beetles from 2003–2012 for each state in the western US. We present these estimates both in terms of absolute tree mortality (Tg AGC yr$^{-1}$) and mortality rate (i.e. % of statewide $\overline{\mathrm{AGC}}$ in tree biomass killed each year).

### 2.3. Tree mortality from timber harvest

We estimated mean annual tree mortality caused by timber harvest from 2003–2012 for each state ($\overline{\mathrm{MORT}}_{\mathrm{harvest}}$, Tg AGC yr$^{-1}$) using timber product output surveys from several USFS reports along with information on tree characteristics. We again used a Monte Carlo analysis to propagate uncertainty in $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$ that was associated with parameter error and temporal variability. The USFS reported harvest for each state in terms of mean annual growing stock volume (GSV, m$^3$ yr$^{-1}$) removed from c. 2003 to 2007 (Smith *et al* 2009) and c. 2008 to 2012 (Oswalt *et al* 2014). Harvest was reported separately for hardwood and softwood tree species. We calculated mean annual GSV removed across these two periods for both species groups in each state ($\overline{\mathrm{GSV}}$). In each case, we then estimated $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$ from $\overline{\mathrm{GSV}}$ using information on the ratio of total aboveground tree volume to GSV($R$), wood density (WD, Mg m$^{-3}$), and the fraction of dry woody biomass that is carbon ($f_{\mathrm{carbon}}$), as per

$$\overline{\mathrm{MORT}}_{\mathrm{harvest}} = \overline{\mathrm{GSV}} \times \overline{R} \times \overline{\mathrm{WD}} \times \overline{f} \qquad (7)$$

We repeatedly ($N_r = 1000$) estimated $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$ for both species groups in each state as part of the Monte Carlo analysis, which again involved randomly varying each parameter by its uncertainty (e.g. as shown in equation (1)). It was not possible to directly compute the uncertainty (SE) associated with temporal variability in the amount of GSV annually removed since annual harvest data were not available for each state. We therefore assumed that the SE was 4.6% of $\overline{\mathrm{GSV}}$ based concurrent annual harvest data from Oregon, which accounted for 35% regional harvest (Oregon Department of Forestry 2017). We used estimates of average $R$ ($\overline{R}$) that varied for hardwoods and softwoods, as well as among states in different USFS regions (Birdsey 1992), assuming in each case that the SE was 2% of $\overline{R}$ (Levy *et al* 2004). We computed average *WD* ($\overline{WD}$) and its uncertainty ($\mathrm{SE_{WD}}$) for both species groups in each state by (1) identifying the four tree species in each group that accounted for the largest percentages of total tree AGB on USFS inventory plots surveyed 2003–2012 (Smith 2002) and then (2) summarizing WD measurements among these species (Miles and Smith 2009). After repeatedly estimating $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$, we then derived the average and associated 95% CI from each set of 1000 realizations (e.g. equation (6)). We report $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$ ($\pm$95% CI) for each state both in terms of absolute mortality (Tg AGC yr$^{-1}$) and mortality rate (% yr$^{-1}$).

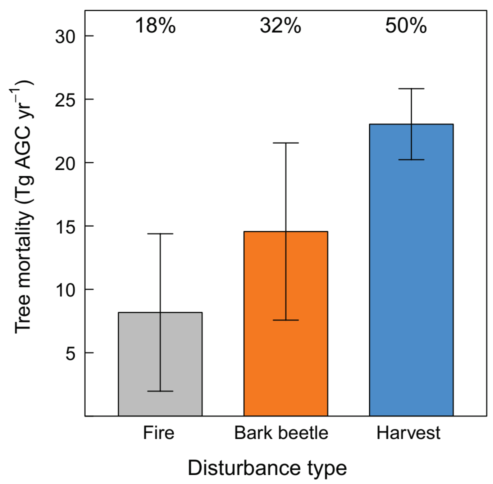


**Figure 3.** Regional mean annual tree mortality from fires, bark beetles, and timber harvest from 2003–2012 on forestland in the western United States. Tree mortality was quantified as the amount of aboveground carbon (AGC) stored in tree biomass killed by disturbance (Tg AGC yr$^{-1}$). Tree mortality from bark beetles and fires was derived using remote sensing, whereas harvest was derived from USFS forest resource assessments (Smith *et al* 2009, Oswalt *et al* 2014). Error bars depict 95% confidence intervals computed with a Monte Carlo analysis and incorporate temporal variability and parameter error. We estimated that regional mean annual tree mortality from fires, beetles, and harvest was 45.8 ± 16.0 Tg AGC yr$^{-1}$ from 2003–2012, with fires, beetles, and harvest accounting for 18%, 32%, and 50% of annual mortality, respectively.

## 3. Results

### 3.1. Regional tree mortality from disturbance

Taken together, mean annual tree mortality from timber harvest, bark beetles, and fires ($\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$) was 45.8 ± 16.0 Tg AGC yr$^{-1}$ from 2003–2012 across the western US. Regional tree $\overline{\mathrm{AGC}}$ stocks totaled 4.16 ± 0.12 Pg, suggesting that the tree mortality rate was 1.10 ± 0.38% yr$^{-1}$. Timber harvest accounted for the largest percentage of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$ (50%), followed by bark beetles (32%) and then fires (18%, figure 3, table S2).

### 3.2. Tree mortality from fire

Mean annual tree mortality from fires ($\overline{\mathrm{MORT}}_{\mathrm{fire}}$) was 8.2 ± 6.2 Tg AGC yr$^{-1}$ and the mortality rate was 0.20 ± 0.15% yr$^{-1}$ from 2003–2012 in the western US (figures 3, 4(*a*), 5, table S2). Absolute $\overline{\mathrm{MORT}}_{\mathrm{fire}}$ was highest in northern California, central Idaho, and western Montana, with these states accounting for 64% of regional $\overline{\mathrm{MORT}}_{\mathrm{fire}}$. Forests in Arizona and New Mexico experienced the highest annual rates of $\overline{\mathrm{MORT}}_{\mathrm{fire}}$ (0.36%–0.57% yr$^{-1}$), as well as the highest percentage of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$ caused by fire (51%–55%). Conversely, forests in Colorado, Oregon, and Washington had the lowest rates of $\overline{\mathrm{MORT}}_{\mathrm{fire}}$ (0.03%–0.08% yr$^{-1}$) and the lowest percentage of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$ caused by fire (2%–8%). Fires that occurred in Oregon and Washington were largely concentrated along the eastern slopes of the Cascade Range.

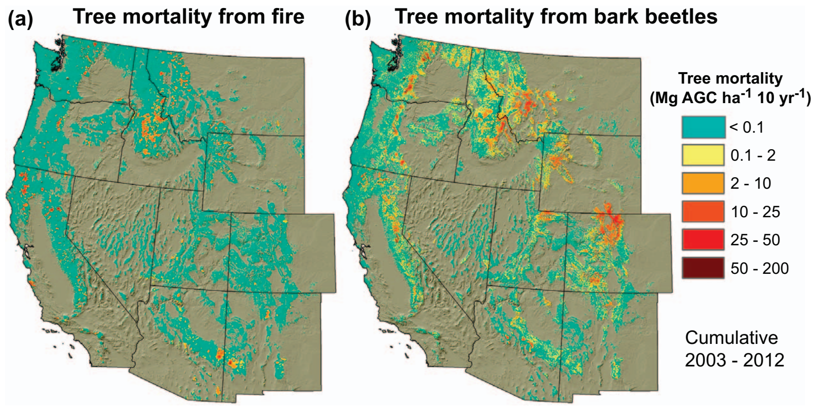


**Figure 4.** Cumulative tree mortality due to (*a*) fires and (*b*) bark beetles from 2003–2012 on forestland in the western United States. Tree mortality was quantified as the amount of aboveground carbon (AGC) stored in tree biomass killed by disturbance (e.g. Mg AGC $ha^{-1}$). Note that (*a*) tree mortality from fire was concentrated in northern California, central Idaho, and southern Arizona/New Mexico, whereas (*b*) tree mortality from bark beetles occurred widely at low levels, but was highest in northern Colorado and western Montana.

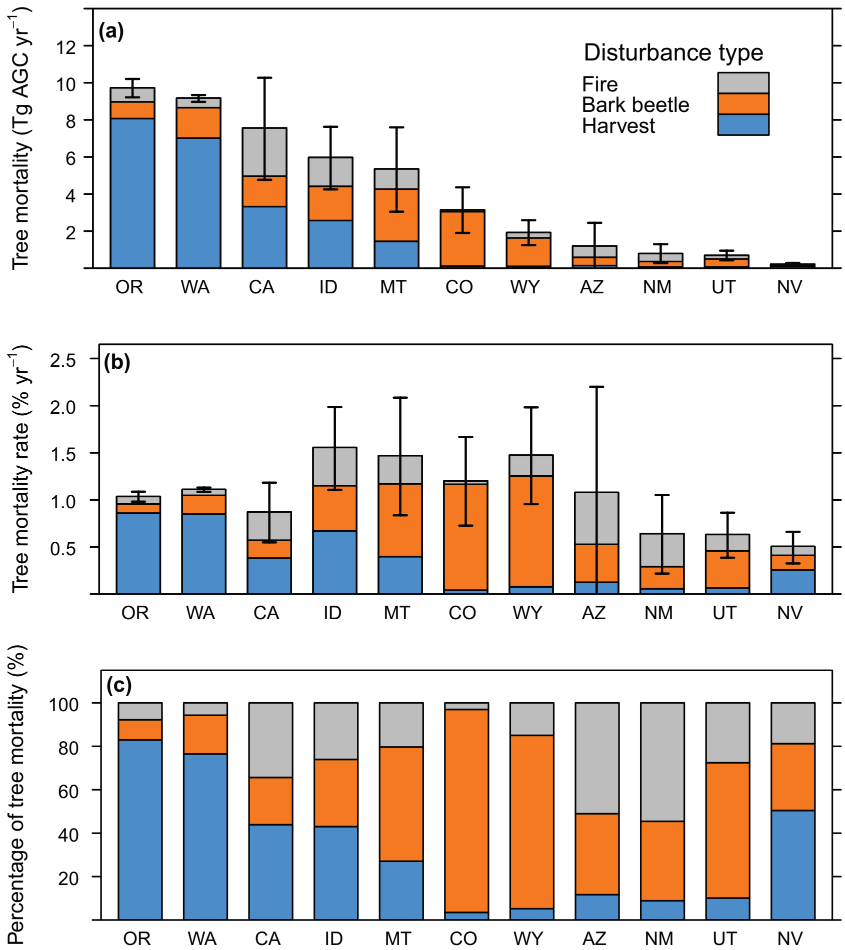


**Figure 5.** Mean annual tree mortality from fires, bark beetles, and timber harvest on forestland from 2003–2012 for each state in the western US. Tree mortality was quantified as the amount of aboveground carbon (AGC) stored in tree biomass killed by disturbance. Mean annual tree mortality is summarized in terms of (*a*) absolute mortality (Tg AGC $yr^{-1}$), (*b*) mortality rate (percentage of statewide AGC in tree biomass killed per year, % $yr^{-1}$), and (*c*) percentage of mortality caused by each type of disturbance (%). Tree mortality from bark beetles and fires was derived using remote sensing, whereas harvest was derived from USFS forest resource assessments. Error bars depict 95% confidence intervals that were computed using a Monte Carlo analysis and incorporate temporal variability and parameter error from all three types of disturbances.

### 3.3. Tree mortality from bark beetles

Mean annual tree mortality from bark beetles ($\overline{\mathrm{MORT}}_{\mathrm{bettle}}$) was 14.6 ± 7.0 Tg AGC $\mathrm{yr}^{-1}$ and the mortality rate was 0.35 ± 0.17% $\mathrm{yr}^{-1}$ from 2003–2012 in the western US (figures 3, 4(*b*), 5, table S2). Absolute $\overline{\mathrm{MORT}}_{\mathrm{bettle}}$ was highest in northern Colorado, western Montana, and central Idaho, with these states accounting for 52% of regional $\overline{\mathrm{MORT}}_{\mathrm{bettle}}$ (table S2). Forests in Colorado and Wyoming had the highest annual rates of $\overline{\mathrm{MORT}}_{\mathrm{bettle}}$ (1.12%–1.22% $\mathrm{yr}^{-1}$) and the highest percentage of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$ caused by bark beetles (80%–93%). Conversely, Oregon and Washington had not only two of the lowest rates of $\overline{\mathrm{MORT}}_{\mathrm{fire}}$, but also two of the lowest rates of $\overline{\mathrm{MORT}}_{\mathrm{bettle}}$ (0.10%–0.20% $\mathrm{yr}^{-1}$).

### 3.4. Tree mortality from timber harvest

Mean annual tree mortality from timber harvest ($\overline{\mathrm{MORT}}_{\mathrm{harvest}}$) was 23.0±2.8 Tg AGC $\mathrm{yr}^{-1}$ and the mortality rate was 0.55±0.07% $\mathrm{yr}^{-1}$ from 2003–2012 in the western US (figures 3, 5, table S2). Timber harvest in Oregon and Washington accounted for 67% of regional $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$. These two states had the highest rates of $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$ (0.85%–0.86% $\mathrm{yr}^{-1}$) and the highest percentage of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$ caused by harvest (76%–83%). Conversely, forest in Colorado, Utah, and New Mexico had the lowest rates of $\overline{\mathrm{MORT}}_{\mathrm{harvest}}$ (0.04%–0.06% $\mathrm{yr}^{-1}$) and among the lowest percentage of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$ caused by harvest (3%–10%).

## 4. Discussion

### 4.1. Regional tree mortality from disturbance

In this study, we quantified tree mortality caused by timber harvest, bark beetles, and fire from 2003–2012 across the western US. We found that regional mean annual tree mortality from timber harvest (~23 Tg AGC $\mathrm{yr}^{-1}$) was quite similar in magnitude to the mortality from fires and bark beetles combined (~22 Tg AGC $\mathrm{yr}^{-1}$). Regional timber harvest declined about 40% since the 1980s (Oswalt *et al* 2014), yet harvest still caused significantly more tree mortality than bark beetles or fires during the 2000s, a period during which hot and dry conditions contributed to extensive beetle and fire activity relative to the last several decades (Williams *et al* 2012, Hicke *et al* 2015, Meddens *et al* 2015, Abatzoglou and Williams 2016, Cohen *et al* 2016).

Our regional tree mortality estimates were comparable with several other studies from the western US. For instance, forest inventories indicated that the regional tree mortality rate was ~0.72%–0.92% $\mathrm{yr}^{-1}$ during the 2000s in areas unaffected by harvest or land clearing (Oswalt *et al* 2014). By comparison, we estimated that the tree mortality rate from fires and bark beetles combined was ~0.56% $\mathrm{yr}^{-1}$, suggesting that these disturbances together might have accounted for ~60%–80% of regional tree mortality not associated timber harvest or land clearing. Our estimates of tree mortality from fires and bark beetles were also of similar magnitude to those predicted using the CASA (Carnegie–Ames–Stanford Approach) carbon cycle model with fire (Ghimire *et al* 2012) and beetle (Ghimire *et al* 2015) disturbances prescribed using MTBS and ADS data sets, respectively. Lastly, our estimates of regional $\mathrm{MORT}_{\mathrm{H+B+F}}$ in 2003 and 2004 differed by ±5% from estimates of tree mortality due to all forms (undifferentiated) of disturbance that were derived using a regional sample of Landsat scenes from 1986–2004 (Powell *et al* 2014). Similarity among these field, modeling, and remote sensing estimates of regional tree mortality is encouraging. Together these comparisons suggest that timber harvest, bark beetles, and fires were the primary causes of tree mortality from disturbance during the 2000s in the western US.

Our current estimates of cumulative (2003–2012) regional tree mortality from fires and bark beetles were both ~40% lower than earlier best-estimates reported by Hicke *et al* (2013). This downward revision stemmed from refinements in representation of tree AGB, more realistic parameterization of tree mortality from fire, and a routine that tracked prior tree mortality within each grid cell. Estimating tree mortality in terms of carbon necessitated accurate representation of tree AGB; however, estimates of tree AGB differ among available data sets, particularly at the pixel scale (figure S1; Huang *et al* 2015, Neeti and Kennedy 2016). We therefore chose to incorporate an ensemble of available data sets (Blackard *et al* 2008, Kellndorfer *et al* 2012, Wilson *et al* 2013) rather than rely on a single data set as was done in the prior study (Hicke *et al* 2013). This improvement accounted for ~10% of the reduction in tree mortality between studies and allowed us to propagate uncertainty in tree AGB into our estimates of mortality. Our current analysis also used a regional synthesis of field measurements (Ghimire *et al* 2012) to vary tree mortality fraction by forest type and fire severity rather than assume complete tree mortality in areas that burned at moderate or high severity (Hicke *et al* 2013). This further reduced our estimates of tree mortality from fire. Lastly, we found that 39% of the forested grid cells (1 $\mathrm{km}^2$) registered multiple disturbances during this decade. We therefore implemented a routine that reduced tree AGB each time a disturbance occurred, which reduced the amount of tree AGB killed by subsequent disturbance. These improvements enabled more robust estimation of how much carbon was stored in trees recently killed by fires and beetles across the western US.

The carbon in trees killed by disturbance will be emitted to the atmosphere as dead trees decompose in the forest and as harvested trees are processed into wood products with varying efficiency and longevity (Harmon *et al* 1990, Harmon *et al* 2011, Hudiburg *et al* 2011). Ignoring pyrogenic emissions, mean annual tree mortality from fire and beetles could together lead to emission of ~0.8 Tg C yr$^{-1}$ during the following decade assuming exponential decay and an intermediate rate constant ($k = 0.04$; Harmon *et al* 1986). Actual decomposition rates will depend on the interplay among snag fall rate, microclimate, wood chemistry, tree anatomy, and other factors (Weedon *et al* 2009, Harmon *et al* 2011). Similarly, mean annual tree mortality from harvest could lead to emission of around 1.3 Tg C yr$^{-1}$ during the following decade assuming that (1) 60% of harvested material went to fuels, paper, and residues that lasted < 5 years (Harmon *et al* 1990) and that (2) the remaining 40% of material went to longer term storage that remained unchanged during the first decade. In other words, tree mortality from harvest, bark beetles, and fires averaged 45.8 ± 16.0 Tg AGC yr$^{-1}$, yet annual emissions are potentially closer to 2 Tg C yr$^{-1}$ during the first decade following disturbance and will persist for decades to centuries. By comparison, regional fossil fuel emissions averaged ~260 Tg C yr$^{-1}$ during this period (US Energy Information Administration 2015). More accurate assessment of the timing and magnitude of carbon emissions requires using robust ecosystem models and life cycle assessments (e.g. Hudiburg *et al* 2011, Ghimire *et al* 2015).

### 4.2. Tree mortality from fire

Tree mortality from fire was highest in the Southwest, with regional variation in mortality associated with differences in drought severity and human activities. States with more consistently severe summer drought tended to have the highest average tree mortality rate from fires (e.g. Arizona), whereas states that experienced less consistently severe drought tended to have lower mortality from fires (e.g. Oregon, figures S2, S3). Hot and dry conditions increase fuel flammability and fire occurrence among forests in the western US (Littell *et al* 2009, Morton *et al* 2013, Abatzoglou and Williams 2016). Furthermore, drought-stricken trees are more likely to die if a fire does occur (van Mantgem *et al* 2013), potentially because of more extensive xylem cavitation and hydraulic impairment during a fire (Michaletz *et al* 2012, van Mantgem *et al* 2013).

Human activities, including fire ignition and suppression, also affected regional variation in fire activity. Human fire ignitions were responsible for ~32% of the total area burned in the western US from 1992–2012 and were concentrated among the more densely populated coastal states (Balch *et al* 2017). Humans were the primary cause of fire ignitions in California (Balch *et al* 2017), where drought-affected forests experienced the highest tree mortality from fire of any western state (~2.6 Tg AGC yr$^{-1}$). The federal government recently spent ~$400 million per year on fire suppression in California, which accounted for about half of the federal expenditure on fire suppression in the western US (Kenward and Raja 2013). Nevertheless, plant mortality from fire averaged ~4.3 Tg AGC yr$^{-1}$ across all of California's ecosystems from 2001–2010, which led to an estimated 8% net reduction in statewide plant AGC and complicated efforts to reduce GHG emissions (Gonzalez *et al* 2015).

It is widely perceived in California and other parts of the region that beetle-killed trees increase subsequent risk of fire (Heller 2017), yet recent studies indicate that beetle outbreaks have not increased burn area (Hart *et al* 2015, Meigs *et al* 2015a) or fire severity in the region (Bond *et al* 2009, Meigs *et al* 2016, Reilly and Spies 2016). Similarly, we found no association between statewide tree mortality rates from fires and beetles ($P = 0.88$), though lagged effects and the spatial scale of this comparison could mask potential interactions.

The MTBS data set provides a valuable record of fire extent and severity across the nation (Eidenshink *et al* 2007), though additional efforts are needed to better quantify fire severity (Kolden *et al* 2015). It is also important to better understand how mortality differs among co-occurring trees species with different functional traits (e.g. bark thickness; Pausas 2015). Projected shifts towards a hotter, drier climate could potentially increase fire activity in the Southwest and other dry parts of the region over the coming century (Spracklen *et al* 2009, Williams *et al* 2014), contributing to increases in tree mortality, cost of wildfire management, and GHG emissions.

### 4.3. Tree mortality from bark beetles

Tree mortality from bark beetles was pervasive at low levels across the region, though was pronounced in the Rocky Mountains (e.g. Colorado) where beetles accounted for ~80% of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$. Mountain pine beetles were responsible for ~62% of tree mortality area (5.37 Mha) caused by bark beetles from 1997–2010 in the western US and largely affected lodgepole pine (Meddens *et al* 2012, Hicke *et al* 2015). In this region, lodgepole pine occur most extensively in the Rocky Mountains, where stands were generally of an age (80–120 years) and stem density (>500 stems ha$^{-1}$) that rendered them more susceptible to beetle attack than lodgepole pine in the coastal states, which tended to be younger and less widespread (Hicke and Jenkins 2008). Forests in the Rocky Mountains also experienced long-term warming that increased beetle populations by reducing beetle mortality during winter and accelerating beetle development (Logan and Powell 2001, Creeden *et al* 2014). Furthermore, states with the most severe singe-year summer drought tended to have the highest average tree mortality rate from beetles (e.g. Colorado, Wyoming), whereas states that experienced less acute drought tended to have lower mortality from beetles (e.g. Oregon, figure S4). High tree mortality from beetles in the Rocky Mountains has been linked to acute drought that weakened tree defense against beetle attack (Creeden *et al* 2014, Hart *et al* 2014) in stands already subject to high structural susceptibility and larger beetle populations (Logan and Powell 2001, Hicke and Jenkins 2008).

Lower tree mortality from beetles in other parts of the region could reflect less severe drought (Abatzoglou and Redmond 2017), lower stand susceptibility (Hicke and Jenkins 2008), and different host and beetle species. Low insect-caused mortality among wet coastal forests (Meddens *et al* 2012, Meigs *et al* 2015b, Reilly and Spies 2016) might be linked to higher tree species diversity (Waring *et al* 2006) limiting host availability and population size of host-specific beetle species (Anderegg *et al* 2015). The coastal forests also tend to be younger than forests in the Rocky Mountains (Pan *et al* 2011b) due to more extensive timber harvest (Masek *et al* 2011), which might reduce the likelihood of beetle attack since beetles tend to prefer large old trees (Raffa *et al* 2008). On the other hand, land-use activities that homogenize forest structure and composition can increase the likelihood of beetle outbreaks (Raffa *et al* 2008). Future efforts are needed to reduced uncertainty in the causes, extent, and severity of bark beetle outbreaks (Meddens *et al* 2012, Gartner *et al* 2015) given that rising temperatures and local land use activities could amplify tree mortality from bark beetles over the coming century in parts of this region (Hicke *et al* 2006, Raffa *et al* 2008).

### 4.4. Tree mortality from harvest

Tree mortality from timber harvest was highest in Oregon and Washington and accounted for ~80% of $\overline{\mathrm{MORT}}_{\mathrm{H+B+F}}$ in these states. Much of the timber harvest in these states occurs in highly productive coastal forests, where rates of timber extraction per unit of forest area are the highest in the country (Masek *et al* 2011, Law and Waring 2015). These coastal forests were a net source of carbon to the atmosphere during the 1980s due to high rates of timber harvest, though declines in harvest following implementation of the Northwest Forest Plan in 1994 led forests on public lands to became a net carbon sink during the 2000s (Turner *et al* 2011). Forests in Oregon and northern California currently store ~3.2 Pg C, yet could theoretically store ~5.9 Pg C if stand-replacing disturbances did not occur on the landscape (Hudiburg *et al* 2009). Recent tree mortality from timber harvest far exceeded tree mortality caused by both bark beetles and fires in the Pacific Northwest, highlighting that reductions in timber harvest could help these states meet GHG emission reduction targets. Continued warming and drying could reduce the amount of tree biomass available to be harvested in the Interior West over the coming century (Williams *et al* 2012, Jiang *et al* 2013, Berner *et al* 2017), yet it is currently unclear how the net effects of ongoing climate change will affect tree biomass and resource availability in the Pacific Northwest (Hudiburg *et al* 2013, Jiang *et al* 2013, Kang *et al* 2014, McDowell *et al* 2015).

## 5. Conclusions

Timber harvest, bark beetles, and fires were important causes of tree mortality from 2003–2012 across forests in the western US. This was a period during which high temperatures contributed to severe drought that increased both fire and bark beetle activity relative to recent decades. Regional tree mortality from these disturbances together averaged 45.8 ± 16.0 Tg AGC yr$^{-1}$, with harvest accounting for a significantly larger percentage (50%) than bark beetles (32%) or fires (18%). The amount of carbon in tree aboveground biomass killed each year by these disturbances was equivalent to ~18% of concurrent regional carbon emissions from fossil fuel consumption (US Energy Information Administration 2015). Tree mortality from timber harvest was concentrated in the high-biomass forests of the Northwest, where it accounted for ~80% of the mortality caused by these three types of disturbance. Shifts in management priorities in the Northwest could reduce tree mortality and subsequent GHG emissions as a means of mitigating climate change. Bark beetles caused tree mortality widely across the region, thought mortality was concentrated in the Rocky Mountains. Tree age and high stem density in these mountain forests made them susceptible to beetle attack, while rising air temperatures increased beetle populations and caused drought-induced reductions in tree defense against beetle attack. Lastly, tree mortality from fires peaked in California, where high temperatures increased fuel aridity and human activities increased fire ignitions. Tree mortality from bark beetles and fires will likely increase in parts of the regions over the coming decades as anthropogenic GHG emissions drive higher temperatures and increased risk of drought. Efforts to manage natural resources and meet GHG emission targets will all benefit from better understanding of the magnitude, location, and causes of tree mortality.

## Data Availability

Our geospatial estimates of tree mortality from fires and bark beetles will be publicly archived with Oak Ridge National Laboratory at the Distributed Active Archive Center for Biogeochemical Dynamics (https://daac.ornl.gov/). This data set includes annual estimates of tree mortality and uncertainty from 2003 to 2012 at 1 km spatial resolution for the western US.

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## References

Abatzoglou J T and Redmond K 2017 West Wide Drought Tracker, University of Idaho (www.wrcc.dri.edu/wwdt/time/)

Abatzoglou J T and Williams A P 2016 Impact of anthropogenic climate change on wildfire across western US forests *Proc. Natl Acad. Sci.* **113** 11770–5

Adams H D, Guardiola-Claramonte M, Barron-Gafford G A, Villegas J C, Breshears D D, Zou C B, Troch P A and Huxman T E 2009 Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought *Proc. Natl Acad. Sci. USA* **106** 7063–6

Allen C D, Breshears D D and McDowell N G 2015 On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the Anthropocene *Ecosphere* **6** 1–55

Allen C D, Macalady A K, Chenchouni H, Bachelet D, McDowell N, Vennetier M, Kitzberger T, Rigling A, Breshears D D and Hogg E H 2010 A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests *Forest Ecol. Manage.* **259** 660–84

Anderegg W R, Hicke J A, Fisher R A, Allen C D, Aukema J, Bentz B, Hood S, Lichstein J W, Macalady A K and McDowell N 2015 Tree mortality from drought, insects, and their interactions in a changing climate *New Phytol.* **208** 674–83

Anderson-Teixeira K J, Snyder P K, Twine T E, Cuadra S V, Costa M H and DeLucia E H 2012 Climate-regulation services of natural and agricultural ecoregions of the Americas *Nat. Clim. Change* **2** 177–81

Baccini A *et al* 2012 Estimated carbon dioxide emissions from tropical deforestation improved by carbon-density maps *Nat. Clim. Change* **2** 182–5

Balch J K, Bradley B A, Abatzoglou J T, Nagy R C, Fusco E J and Mahood A L 2017 Human-started wildfires expand the fire niche across the United States *Proc. Natl Acad. Sci.* **114** 2946–51

Berner L T, Law B E and Hudiburg T W 2017 Water availability limits tree productivity, carbon stocks, and carbon residence time in mature forests across the western US *Biogeosciences* **14** 365–78

Birdsey R A 1992 Carbon storage accumulation in United States forest ecosystems *General Technical Report WO-59* (Washington, DC: USDA Forest Service) 55

Blackard J, Finco M, Helmer E, Holden G, Hoppus M, Jacobs D, Lister A, Moisen G, Nelson M and Riemann R 2008 Mapping US forest biomass using nationwide forest inventory data and moderate resolution information *Remote Sens. Environ.* **112** 1658–77

Bonan G B 2008 Forests and climate change: forcings, feedbacks, and the climate benefits of forests *Science* **320** 1444–9

Bond M L, Lee D E, Bradley C M and Hanson C T 2009 Influence of pre-fire tree mortality on fire severity in conifer forests of the San Bernardino Mountains, California *Open Forest Sci. J.* **2** 41–7

Brinck K, Fischer R, Groeneveld J, Lehmann S, De Paula M D, Pütz S, Sexton J O, Song D and Huth A 2017 High resolution analysis of tropical forest fragmentation and its impact on the global carbon cycle *Nat. Commun.* **8** 14855

Carnell R 2016 triangle: Provides the standard distribution functions for the triangle distribution *R package version 0.10* (https://cran.r-project.org/package=triangle)

Cohen W B, Yang Z, Stehman S V, Schroeder T A, Bell D M, Masek J G, Huang C and Meigs G W 2016 Forest disturbance across the conterminous United States from 1985–2012: the emerging dominance of forest decline *Forest Ecol. Manage.* **360** 242–52

Cook B I, Ault T R and Smerdon J E 2015 Unprecedented 21st century drought risk in the American Southwest and Central Plains *Sci. Adv.* **1** e1400082

Creeden E P, Hicke J A and Buotte P C 2014 Climate, weather, and recent mountain pine beetle outbreaks in the western United States *Forest Ecol. Manage.* **312** 239–51

Dennison P E, Brewer S C, Arnold J D and Moritz M A 2014 Large wildfire trends in the western United States, 1984–2011 *Geophys. Res. Lett.* **41** 2928–33

Diffenbaugh N S, Swain D L and Touma D 2015 Anthropogenic warming has increased drought risk in California *Proc. Natl Acad. Sci.* **112** 3931–6

Eidenshink J, Schwind B, Brewer K, Zhu Z, Quayle B and Howard S 2007 A project for monitoring trends in burn severity *Fire Ecol.* **3** 3–21

FAO 2017 Forest production and trade (United Nations Food and Agriculture Organization)

Gartner M H, Veblen T T, Leyk S and Wessman C A 2015 Detection of mountain pine beetle-killed ponderosa pine in a heterogeneous landscape using high-resolution aerial imagery *Int. J. Remote Sens.* **36** 5353–72

Ghimire B, Williams C A, Collatz G J and Vanderhoof M 2012 Fire-induced carbon emissions and regrowth uptake in western US forests: documenting variation across forest types, fire severity, and climate regions *J. Geophys. Res. Biogeosci.* **117** G03036

Ghimire B, Williams C A, Collatz G J, Vanderhoof M, Rogan J, Kulakowski D and Masek J G 2015 Large carbon release legacy from bark beetle outbreaks across Western United States *Glob. Change Biol.* **21** 3087–101

Goetz S J *et al* 2012 Observations and assessment of forest carbon dynamics following disturbance in North America *J. Geophys. Res.* **117** G02022

Gonzalez P, Battles J J, Collins B M, Robards T and Saah D S 2015 Aboveground live carbon stock changes of California wildland ecosystems, 2001–2010 *Forest Ecol. Manage.* **348** 68–77

Harmon M E, Bond-Lamberty B, Tang J and Vargas R 2011 Heterotrophic respiration in disturbed forests: a review with examples from North America *J. Geophys. Res. Biogeosci.* **116** G00K04

Harmon M E, Ferrell W K and Franklin J F 1990 Effects on carbon storage of conversion of old-growth forests to young forests *Science* **247** 699–702

Harmon M E, Franklin J F, Swanson F J, Sollins P, Gregory S, Lattin J, Anderson N, Cline S, Aumen N G and Sedell J 1986 Ecology of coarse woody debris in temperate ecosystems *Adv. Ecol. Res.* **15** 133–302

Harmon M E, Phillips D L, Battles J, Rassweiler A, Hall R and Lauenroth W 2007 *Principles and Standards for Measuring Net Primary Production in Long-Term Ecological Studies* ed T Fahey and A Knapp (Oxford: Oxford University Press) pp 238–60

Hart S J, Schoennagel T, Veblen T T and Chapman T B 2015 Area burned in the western United States is unaffected by recent mountain pine beetle outbreaks *Proc. Natl Acad. Sci.* **112** 4375–80

Hart S J, Veblen T T, Eisenhart K S, Jarvis D and Kulakowski D 2014 Drought induces spruce beetle (*Dendroctonus rufipennis*) outbreaks across northwestern Colorado *Ecology* **95** 930–9

Heath L S, Anderson S M, Emery M R, Hicke J A, Littell J, Lucier A, Masek J G, Peterson D L, Pouyat R and Potter K M 2015 Indicators of climate impacts for forests: recommendations for the US National Climate Assessment indicators system *General Technical Report NRS-155* (Newtown Square, PA: USDA Forest Service) 156

Heller M 2017 Federal fire policy may be based on faulty science *E&E News Greenwire* (www.eenews.net/greenwire/stories/1060051219/print)

Hicke J A and Jenkins J C 2008 Mapping lodgepole pine stand structure susceptibility to mountain pine beetle attack across the western United States *Forest Ecol. Manage.* **255** 1536–47

Hicke J A, Logan J A, Powell J and Ojima D S 2006 Changing temperatures influence suitability for modeled mountain pine beetle (*Dendroctonus ponderosae*) outbreaks in the western United States *J. Geophys. Res. Biogeosci.* **111** G02019

Hicke J A, Meddens A J, Allen C D and Kolden C A 2013 Carbon stocks of trees killed by bark beetles and wildfire in the western United States *Environ. Res. Lett.* **8** 035032

Hicke J A, Meddens A J and Kolden C A 2015 Recent tree mortality in the Western United States from bark beetles and forest fires *Forest Sci.* **62** 141–53

Hijmans R J and van Etten J 2013 *Raster: Geographic Analysis and Modeling with Raster Data* (Vienna: R Foundation for Statistical Computing)

Homer C, Dewitz J, Fry J, Coan M, Hossain N, Larson C, Herold N, McKerrow A, VanDriel J N and Wickham J 2007 Completion of the 2001 national land cover database for the counterminous United States *Photogramm. Eng. Remote Sens.* **73** 337

Houghton R 2013 *Global Forest Monitoring from Earth Observation* ed F Achard and M C Hansen (Boca Raton, FL: CRC Press) pp 15–39

Huang W, Swatantran A, Johnson K, Duncanson L, Tang H, O'Neil Dunne J, Hurtt G and Dubayah R 2015 Local discrepancies in continental scale biomass maps: a case study over forested and non-forested landscapes in Maryland, USA *Carbon Balance Manage.* **10** 1–16

Hudiburg T, Law B, Turner D P, Campbell J, Donato D and Duane M 2009 Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage *Ecol. Appl.* **19** 163–80

Hudiburg T W, Law B E, Wirth C and Luyssaert S 2011 Regional carbon dioxide implications of forest bioenergy production *Nat. Clim. Change* **1** 419–23

Hudiburg T W, Luyssaert S, Thornton P E and Law B E 2013 Interactive effects of environmental change and management strategies on regional forest carbon emissions *Environ. Sci. Technol.* **47** 13132–40

Jiang X, Rauscher S A, Ringler T D, Lawrence D M, Williams A P, Allen C D, Steiner A L, Cai D M and McDowell N G 2013 Projected future changes in vegetation in Western North America in the twenty-first century *J. Clim.* **26** 3671–87

Johnson E and Wittwer D 2008 Aerial detection surveys in the United States *Australian Forestry* **71** 212–5

Kang S, Running S W, Kimball J S, Fagre D B, Michaelis A, Peterson D L, Halofsky J E and Hong S 2014 Effects of spatial and temporal climatic variability on terrestrial carbon and water fluxes in the Pacific Northwest, USA *Environ. Modell. Softw.* **51** 228–39

Kautz M, Meddens A J, Hall R J and Arneth A 2016 Biotic disturbances in Northern Hemisphere forests–a synthesis of recent data, uncertainties and implications for forest monitoring and modelling *Glob. Ecol. Biogeogr.* **26** 533–52

Kellndorfer J, Walker W, LaPoint E, Bishop J, Cormier T, Fiske G, Hoppus M, Kirsch K and Westfall J 2012 NACP Aboveground Biomass and Carbon Baseline Data (NBCD 2000), USA, 2000 Data set *ORNL DAAC, Oak Ridge, Tennessee, USA*

Kenward A and Raja U 2013 Nearly half of all western wildfire costs go to California *Climate Central* (www.climatecentral.org/news/nearly-half-of-all-wildfire-costs-go-to-california-16406)

Kolden C A, Smith A M and Abatzoglou J T 2015 Limitations and utilisation of monitoring trends in burn severity products for assessing wildfire severity in the USA *Int. J. Wildland Fire* **24** 1023–8

Kunkel K E, Stevens L E, Stevens S E, Sun L, Janssen E, Wuebbles D, Redmond K T and Dobson J G 2013 Regional climate trends and scenarios for the US National Climate Assessment: Part 5 *Climate of the Southwest US* (Washington, DC: US Department of Commerce, National Oceanic and Atmospheric Administration, National Environmental Satellite, Data, and Information Service) p 87

Kurz W A, Dymond C C, Stinson G, Rampley G J, Neilson E T, Carroll A L, Ebata T and Safranyik L 2008 Mountain pine beetle and forest carbon feedback to climate change *Nature* **452** 987–90

Lamlom S and Savidge R 2003 A reassessment of carbon content in wood: variation within and between 41 North American species *Biomass Bioenergy* **25** 381–8

Law B E and Waring R H 2015 Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests *Forest Ecol. Manage.* **355** 4–14

Levy P, Hale S and Nicoll B 2004 Biomass expansion factors and root: shoot ratios for coniferous tree species in Great Britain *Forestry* **77** 421–30

Littell J S, McKenzie D, Peterson D L and Westerling A L 2009 Climate and wildfire area burned in western US ecoprovinces, 1916–2003 *Ecol. Appl.* **19** 1003–21

Littell J S, Peterson D L, Riley K L, Liu Y and Luce C H 2016 A review of the relationships between drought and forest fire in the United States *Glob. Change Biol.* **22** 2353–69

Logan J A and Powell J A 2001 Ghost forests, global warming, and the mountain pine beetle (Coleoptera: Scolytidae) *Am. Entomol.* **47** 160

Masek J G, Cohen W B, Leckie D, Wulder M A, Vargas R, de Jong B, Healey S, Law B, Birdsey R and Houghton R 2011 Recent rates of forest harvest and conversion in North America *J. Geophys. Res. Biogeosci.* **116** G00K03

McCabe G J, Palecki M A and Betancourt J L 2004 Pacific and Atlantic Ocean influences on multidecadal drought frequency in the United States *Proc. Natl Acad. Sci.* **101** 4136–41

McDowell N G, Williams A, Xu C, Pockman W, Dickman L, Sevanto S, Pangle R, Limousin J, Plaut J and Mackay D 2015 Multi-scale predictions of massive conifer mortality due to chronic temperature rise *Nat. Clim. Change* **6** 295–300

Meddens A J, Hicke J A and Ferguson C A 2012 Spatiotemporal patterns of observed bark beetle-caused tree mortality in British Columbia and the western United States *Ecol. Appl.* **22** 1876–91

Meddens A J, Hicke J A, Macalady A K, Buotte P C, Cowles T R and Allen C D 2015 Patterns and causes of observed piñon pine mortality in the southwestern United States *New Phytol.* **206** 91–7

Meigs G W, Campbell J L, Zald H S, Bailey J D, Shaw D C and Kennedy R E 2015a Does wildfire likelihood increase following insect outbreaks in conifer forests? *Ecosphere* **6** 1–24

Meigs G W, Kennedy R E, Gray A N and Gregory M J 2015b Spatiotemporal dynamics of recent mountain pine beetle and western spruce budworm outbreaks across the Pacific Northwest Region, USA *Forest Ecol. Manage.* **339** 71–86

Meigs G W, Zald H S, Campbell J L, Keeton W S and Kennedy R E 2016 Do insect outbreaks reduce the severity of subsequent forest fires? *Environ. Res. Lett.* **11** 045008

Michaletz S T, Johnson E and Tyree M 2012 Moving beyond the cambium necrosis hypothesis of post-fire tree mortality: cavitation and deformation of xylem in forest fires *New Phytol.* **194** 254–63

Miles P K and Smith B W 2009 Specific gravity and other properties of wood and bark for 156 tree species found in north America *USDA Forest Service Northern Research Station, NSP-RP-38* 18

Morton D, Collatz G, Wang D, Randerson J, Giglio L and Chen Y 2013 Satellite-based assessment of climate controls on US burned area *Biogeosciences* **10** 247–60

Mote P A *et al* 2014 *Climate Change Impacts in the United States: The Third National Climate Assessment* ed J M Melillo (Washington, DC: US Global Change Research Program) pp 487–513

Mote P W, Hamlet A F, Clark M P and Lettenmaier D P 2005 Declining mountain snowpack in Western North America *Bull. Am. Meteorol. Soc.* **86** 39–49

Neeti N and Kennedy R 2016 Comparison of national level biomass maps for conterminous US: understanding pattern and causes of differences *Carbon Balance Manage.* **11** 1–19

Omernik J M 1987 Ecoregions of the conterminous United States *Ann. Assoc. Am. Geogr.* **77** 118–25

Oregon Department of Forestry 2017 Timber Harvest Data 1942–2015 *Available online* (https//data.oregon.gov/Natural-Resources/Timber-Harvest-Data-1942-2015/v7yh-3r7a) (Accessed: 14 July 2016)

Oswalt S N, Smith W B, Miles P D and Pugh S A 2014 Forest Resources of the United States, 2012: a technical document supporting the Forest Service 2010 update of the RPA Assessment *General Technical Report WO-91* (Washington, DC: USDA Forest Service) 228

Pacala S W, Breidenich C, Brewer P, Fung I, Gunson M, Heddle G, Law B, Marland G, Paustian K and Prather M 2010 Verifying greenhouse gas emissions: methods to support international climate agreements *Committee on Methods for Estimating Greenhouse Gas Emissions*

Pan Y, Birdsey R A, Fang J, Houghton R, Kauppi P E, Kurz W A, Phillips O L, Shvidenko A, Lewis S L and Canadell J G 2011a A large and persistent carbon sink in the world's forests *Science* **333** 988–93

Pan Y, Chen J M, Birdsey R, McCullough K, He L and Deng F 2011b Age structure and disturbance legacy of North American forests *Biogeosciences* **8** 715–32

Pausas J G 2015 Bark thickness and fire regime *Funct. Ecol.* **29** 315–27

Powell S L, Cohen W B, Kennedy R E, Healey S P and Huang C 2014 Observation of trends in biomass loss as a result of disturbance in the conterminous US: 1986–2004 *Ecosystems* **17** 142–57

R Core Team 2015 *R: A Language and Environment for Statistical Computing* (Vienna: R Foundation for Statistical Computing)

Raffa K F, Aukema B H, Bentz B J, Carroll A L, Hicke J A, Turner M G and Romme W H 2008 Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions *Bioscience* **58** 501–17

Reilly M J and Spies T A 2016 Disturbance, tree mortality, and implications for contemporary regional forest change in the Pacific Northwest *Forest Ecol. Manage.* **374** 102–10

Rollins M G 2009 LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment *Int. J. Wildland Fire* **18** 235–49

Ruefenacht B, Finco M, Nelson M, Czaplewski R, Helmer E, Blackard J, Holden G, Lister A, Salajanu D and Weyermann D 2008 Conterminous US and Alaska forest type mapping using forest inventory and analysis data *Photogramm. Eng. Remote Sens.* **74** 1379–88

Schwalm C R, Williams C A, Schaefer K, Baldocchi D, Black T A, Goldstein A H, Law B E, Oechel W C and Scott R L 2012 Reduction in carbon uptake during turn of the century drought in western North America *Nat. Geosci.* **5** 551–6

Smith W, Miles P, Perry C and Pugh S 2009 Forest resources of the United States 2007: A technical document supporting the Forest Service 2010 RPA assessment *General Technical Report WO-78* (Washington, DC: USDA Forest Service) 349

Smith W B 2002 Forest inventory and analysis: a national inventory and monitoring program *Environ. Pollut.* **116** S233–42

Spracklen D V, Mickley L J, Logan J A, Hudman R C, Yevich R, Flannigan M D and Westerling A L 2009 Impacts of climate change from 2000 to 2050 on wildfire activity and carbonaceous aerosol concentrations in the western United States *J. Geophys. Res. Atmos.* **114** D20301

Turner D P, Ritts W D, Yang Z, Kennedy R E, Cohen W B, Duane M V, Thornton P E and Law B E 2011 Decadal trends in net ecosystem production and net ecosystem carbon balance for a regional socioecological system *Forest Ecol. Manage.* **262** 1318–25

US Energy Information Administration 2015 *Energy-related Carbon Dioxide Emissions at the State Level, 2000–2013* (Washington, DC: US Department of Energy)

van Mantgem P J, Nesmith J C, Keifer M, Knapp E E, Flint A and Flint L 2013 Climatic stress increases forest fire severity across the western United States *Ecology letters* **16** 1151–6

Walsh J *et al* 2014 *Climate Change Impacts in the United States: The Third National Climate Assessment* ed J M Melillo *et al* (US Global Change Research Program) pp 19–67

Waring R, Coops N, Fan W and Nightingale J 2006 MODIS enhanced vegetation index predicts tree species richness across forested ecoregions in the contiguous USA *Remote Sens. Environ.* **103** 218–26

Weedon J T, Cornwell W K, Cornelissen J H, Zanne A E, Wirth C and Coomes D A 2009 Global meta-analysis of wood decomposition rates: a role for trait variation among tree species? *Ecol. Lett.* **12** 45–56

Westerling A L, Hidalgo H G, Cayan D R and Swetnam T W 2006 Warming and earlier spring increase western US forest wildfire activity *Science* **313** 940–3

Wickham H and Francois R 2015 dplyr: A Grammar of Data Manipulation *R package version 0.4.2* (http://CRAN.R-project.org/package=dplyr)

Williams A P, Allen C D, Macalady A K, Griffin D, Woodhouse C A, Meko D M, Swetnam T W, Rauscher S A, Seager R and Grissino-Mayer H D 2012 Temperature as a potent driver of regional forest drought stress and tree mortality *Nat. Clim. Change* **3** 292–7

Williams A P, Seager R, Macalady A K, Berkelhammer M, Crimmins M A, Swetnam T W, Trugman A T, Buenning N, Noone D and McDowell N G 2014 Correlations between components of the water balance and burned area reveal new insights for predicting forest fire area in the southwest United States *Int. J. Wildland Fire* **24** 14–26

Williams C A, Gu H, MacLean R, Masek J G and Collatz G J 2016 Disturbance and the carbon balance of US forests: a quantitative review of impacts from harvests, fires, insects, and droughts *Glob. Planet. Chang.* **143** 66–80

Wilson B T, Woodall C and Griffith D 2013 Imputing forest carbon stock estimates from inventory plots to a nationally continuous coverage *Carbon Balance Manage.* **8** 1–15

# 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.**

Author contributions: B.E.L. and T.W.H. designed research; B.E.L., T.W.H., and P.C.B. performed research; M.E.H. contributed new reagents/analytic tools; B.E.L., L.T.B., J.J.K., and P.C.B. analyzed data; B.E.L., T.W.H., L.T.B., and M.E.H. wrote the paper; and M.E.H. contributed the substitution model.

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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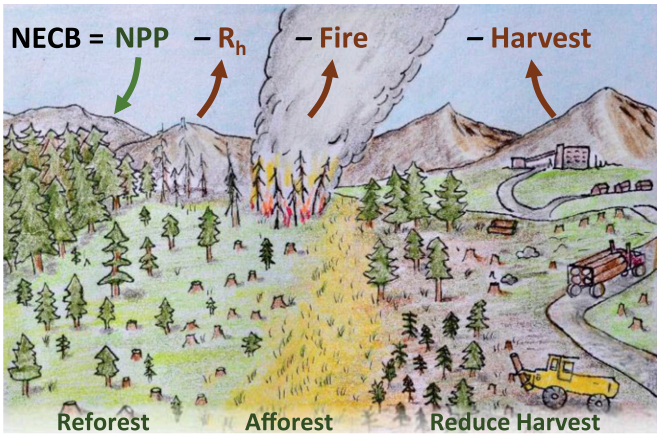


**Fig. 1.** Approach to assessing effects of mitigation strategies on forest carbon and forest sector emissions. NECB is productivity (NPP) minus Rh and losses from fire and harvest (red arrows). Harvest emissions include those associated with wood products and bioenergy.

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 $tCO_2e$).

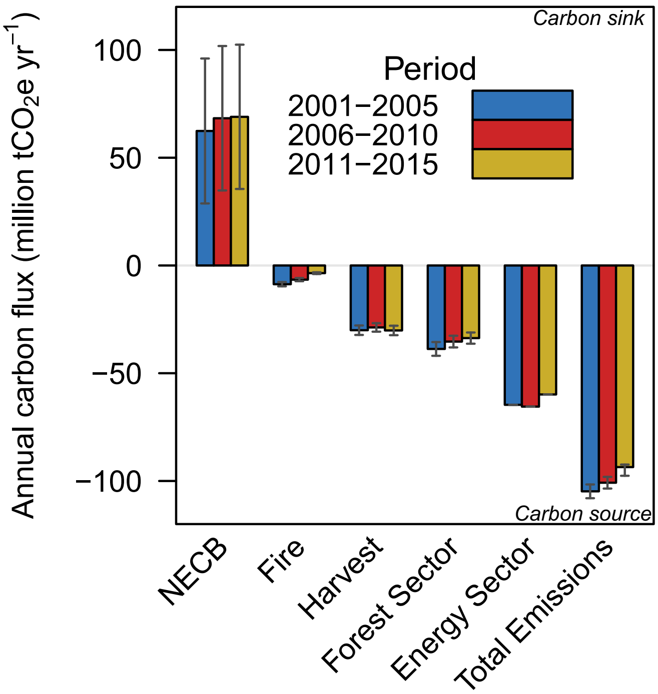


**Fig. 2.** Oregon's forest carbon sink and emissions from forest and energy sectors. Harvest emissions are computed by LCA. Fire and harvest emissions sum to forest sector emissions. Energy sector emissions are from the Oregon Global Warming Commission (14), minus forest-related emissions. Error bars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

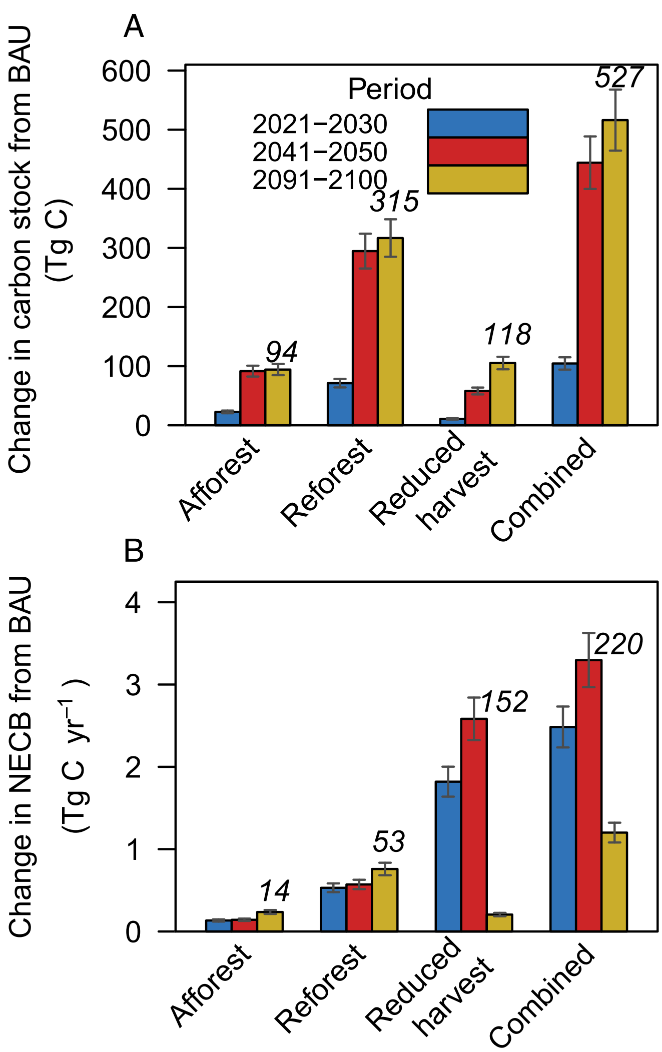


**Fig. 3.** Future change in carbon stocks and NECB with mitigation strategies relative to BAU management. The decadal average change in forest carbon stocks (*A*) and NECB relative to BAU (*B*) are shown. Italicized numbers over bars indicate mean forest carbon stocks in 2091–2100 (*A*) and cumulative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

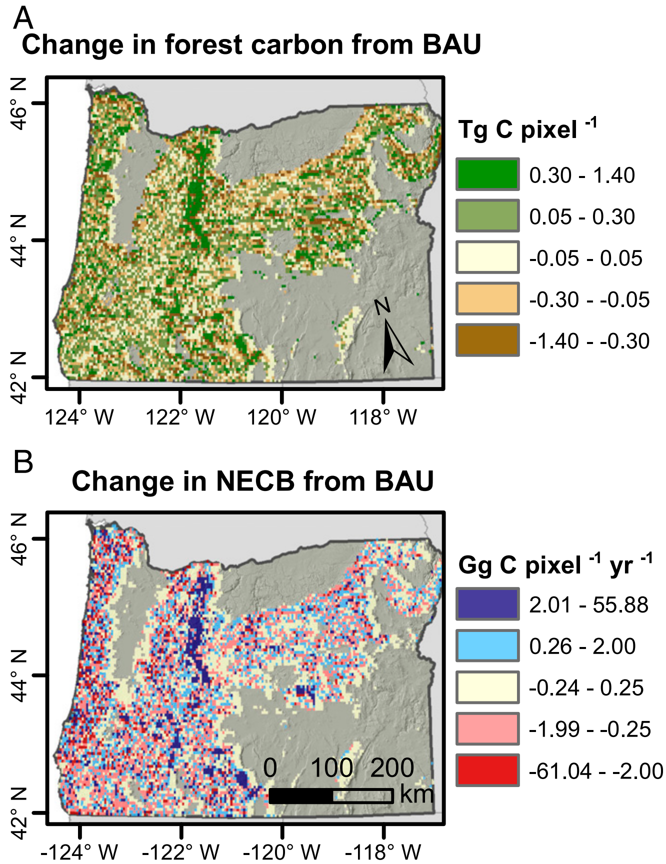


**Fig. 4.** Spatial patterns of forest carbon stocks and NECB by 2091–2100. The decadal average changes in forest carbon stocks (*A*) and NECB (*B*) due to afforestation, reforestation, protected areas, and lengthened harvest cycles relative to continued BAU forest management (red is increase in NECB)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

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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1. Canadell JG, Raupach MR (2008) Managing forests for climate change mitigation. *Science* 320:1456–1457.
2. Holtsmark B (2013) The outcome is in the assumptions: Analyzing the effects on atmospheric CO2 levels of increased use of bioenergy from forest biomass. *Glob Change Biol Bioenergy* 5:467–473.
3. Repo A, Tuomi M, Liski J (2011) Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. *Glob Change Biol Bioenergy* 3:107–115.
4. Schlamadinger B, Marland G (1996) The role of forest and bioenergy strategies in the global carbon cycle. *Biomass Bioenergy* 10:275–300.
5. Heck V, Gerten D, Lucht W, Popp A (2018) Biomass-based negative emissions difficult to reconcile with planetary boundaries. *Nat Clim Chang* 8:151–155.
6. Field CB, Mach KJ (2017) Rightsizing carbon dioxide removal. *Science* 356:706–707.
7. Tollefson J (2017) The wooden skyscrapers that could help to cool the planet. *Nature* 545:280–282.
8. Law BE, Harmon ME (2011) Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. *Carbon Manag* 2: 73–84.
9. Law BE (2014) Regional analysis of drought and heat impacts on forests: Current and future science directions. *Glob Change Biol* 20:3595–3599.
10. Hudiburg TW, Luyssaert S, Thornton PE, Law BE (2013) Interactive effects of environmental change and management strategies on regional forest carbon emissions. *Environ Sci Technol* 47:13132–13140.
11. Keith H, Mackey BG, Lindenmayer DB (2009) Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proc Natl Acad Sci USA* 106:11635–11640.
12. Law B, Waring R (2015) Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests. *For Ecol Manage* 355:4–14.
13. Berner LT, Law BE, Meddens AJ, Hicke JA (2017) Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003–2012). *Environ Res Lett* 12:065005.
14. Oregon Global Warming Commission (2017) Biennial Report to the Legislature (Oregon Global Warming Commission, Salem, OR).
15. Campbell J, Donato D, Azuma D, Law B (2007) Pyrogenic carbon emission from a large wildfire in Oregon, United States. *J Geophys Res Biogeosci* 112:G04014.
16. King AW, Hayes DJ, Huntzinger DN, West TO, Post WM (2012) North American carbon dioxide sources and sinks: Magnitude, attribution, and uncertainty. *Front Ecol Environ* 10:512–519.
17. Wilson BT, Woodall CW, Griffith DM (2013) Imputing forest carbon stock estimates from inventory plots to a nationally continuous coverage. *Carbon Balance Manag* 8:1.
18. Reilly MJ, et al. (2017) Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). *Ecosphere* 8:e01695.
19. Anderson CM, Field CB, Mach KJ (2017) Forest offsets partner climate-change mitigation with conservation. *Front Ecol Environ* 15:359–365.
20. Willamette Water 2100 Explorer (2017) Assessing water futures under alternative climate and management scenarios: Agricultural water demand, crop and irrigation decisions. Available at explorer.bee.oregonstate.edu/Topic/WW2100/AgSummaries.aspx. Accessed November 13, 2017.
21. Hudiburg T, et al. (2009) Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage. *Ecol Appl* 19:163–180.
22. Bonan GB, Doney SC (2018) Climate, ecosystems, and planetary futures: The challenge to predict life in Earth system models. *Science* 359:eaam8328.
23. Riitters KH, Wickham JD (2012) Decline of forest interior conditions in the conterminous United States. *Sci Rep* 2:653.
24. Noss RF, LaRoe ET, Scott JM (1995) *Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation* (US Department of the Interior, National Biological Service, Washington, DC).
25. Jaeger W, Plantinga A, Langpap C, Bigelow D, Moore K (2017) *Water, Economics and Climate Change in the Willamette Basin* (Oregon State University Extension Service, Corvallis, OR).
26. Kline JD, et al. (2016) Evaluating carbon storage, timber harvest, and habitat possibilities for a Western Cascades (USA) forest landscape. *Ecol Appl* 26:2044–2059.
27. Matthews S, O'Connor R, Plantinga AJ (2002) Quantifying the impacts on biodiversity of policies for carbon sequestration in forests. *Ecol Econ* 40:71–87.
28. Arnold S, Kagan J, Taylor B (2000) Summary of current status of Oregon's biodiversity. *Oregon State of the Environment Report* (Oregon Progress Board, Salem, OR), pp 121–126.
29. Bright RM, et al. (2017) Local temperature response to land cover and management change driven by non-radiative processes. *Nat Clim Chang* 7:296–302.
30. Betts M, Phalan B, Frey S, Rousseau J, Yang Z (2017) Old-growth forests buffer climate-sensitive bird populations from warming. *Diversity Distrib*, 10.1111/ddi.12688.
31. Architecture 2030(2017) The 2030 Challenge. Available at architecture2030.org. Accessed October 23, 2017.
32. Oliver CD, Nassar NT, Lippke BR, McCarter JB (2014) Carbon, fossil fuel, and biodiversity mitigation with wood and forests. *J Sustain For* 33:248–275.
33. Harmon ME, Moreno A, Domingo JB (2009) Effects of partial harvest on the carbon stores in Douglas-fir/western hemlock forests: A simulation study. *Ecosystems (N Y)* 12:777–791.
34. Lippke B, et al. (2011) Life cycle impacts of forest management and wood utilization on carbon mitigation: Knowns and unknowns. *Carbon Manag* 2:303–333.
35. Gunn JS, Ganz DJ, Keeton WS (2011) Biogenic vs. geologic carbon emissions and forest biomass energy production. *GCB Bioenergy* 4:239–242.
36. Hudiburg TW, Law BE, Wirth C, Luyssaert S (2011) Regional carbon dioxide implications of forest bioenergy production. *Nat Clim Chang* 1:419–423.
37. Law BE, Berner LT (2015) *NACP TERRA-PNW: Forest Plant Traits, NPP, Biomass, and Soil Properties, 1999-2014* (ORNL DAAC, Oak Ridge, TN).
38. Law BE, Hudiburg TW, Luyssaert S (2013) Thinning effects on forest productivity: Consequences of preserving old forests and mitigating impacts of fire and drought. *Plant Ecol Divers* 6:73–85.
39. Meigs G, Donato D, Campbell J, Martin J, Law B (2009) Forest fire impacts on carbon uptake, storage, and emission: The role of burn severity in the Eastern Cascades, Oregon. *Ecosystems (N Y)* 12:1246–1267.
40. Harmon ME, Marks B (2002) Effects of silvicultural practices on carbon stores in Douglas-fir western hemlock forests in the Pacific Northwest, USA: Results from a simulation model. *Can J For Res* 32:863–877.
41. Hudiburg TW, Law BE, Thornton PE (2013) Evaluation and improvement of the community land model (CLM4) in Oregon forests. *Biogeosciences* 10:453–470.
42. Ruefenacht B, et al. (2008) Conterminous U.S. and Alaska forest type mapping using forest inventory and analysis data. *Photogramm Eng Remote Sens* 74:1379–1388.
43. Peterman W, Bachelet D, Ferschweiler K, Sheehan T (2014) Soil depth affects simulated carbon and water in the MC2 dynamic global vegetation model. *Ecol Modell* 294:84–93.
44. Smith JE, Heath L, Skog KE, Birdsey R (2006) Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States (US Department of Agriculture, Forest Service, Northeastern Research Station, Newtown Square, PA), General Technical Report NE-343.
45. Harmon ME, Harmon JM, Ferrell WK, Brooks D (1996) Modeling carbon stores in Oregon and Washington forest products: 1900–1992. *Clim Change* 33:521–550.
46. Sathre R, O'Connor J (2010) Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environ Sci Policy* 13:104–114.