Global Forest Watch Report

Clearcutting our Carbon Accounts[1]

How State and private forest practices are subverting Oregon's climate agenda

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Key points

- The timber industry shields its global warming pollution from public scrutiny by using a greenhouse gas (GHG) accounting trick other sectors cannot make use of – taking credit for the emissions reductions achieved by others. In particular, the timber industry claims that the carbon dioxide absorbed by forests conserved by non-profits, small landowners, and government exceed what it emits and therefore net emissions from what they call the "forest sector" are zero.
- Because of this accounting trick – used worldwide – GHG emissions from the timber industry in Oregon have not been tracked and evaluated since 2002 and are ignored by Oregon's climate agenda. But rapid clearcutting on these lands over the past 14 years has generated significant greenhouse gas emissions from industrial logging and a loss of carbon sequestration capacity.
- These emissions have averaged between 9.75 and 19.35 million metric tons carbon dioxide equivalent (MMT CO2-e) per year since 2000 on State and private forestlands in western Oregon. This represents between 16% and 32% of the 60.8 million MMT CO2-e "in-boundary" emissions estimated for the State by the latest (2012) GHG inventory.
- These emissions are four to seven times higher those associated with coal combustion by the Boardman coal-fired plant in 2012, are equivalent to 2-4 million new cars on the road, and make logging on State and private lands one of Oregon's biggest GHG polluters and a major impediment to Oregon's ambitious GHG reduction targets.
- Industrial forest practices are also undermining goals for climate adaptation by keeping millions of acres of forestland in a high-risk condition for wildfire, landslides, disease and pest outbreaks while contributing to thermal pollution deadly to coldwater fisheries.
- Climate policy makers in Oregon can remedy the situation by accounting for timber industry emissions just like other sectors, promoting alternatives to clearcutting, lengthening timber harvest rotations, protecting state forestlands and reforming the timber tax code to incentivize carbon storage.

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Oregon prides itself on being a leader in the fight against climate change

In 2007 the Oregon Legislature codified an ambitious agenda to both fight and adapt to global warming by passing House Bill 3543. The law sets targets for reducing Oregon's greenhouse gas emissions (GHG) and directs the Oregon Global Warming Commission (OGWC) and Oregon Climate Change Research Institute (OCCRI) to recommend policies to meet those targets. The legislation also charges these entities with the task of helping communities adapt to the effects of reduced snowpack, changes in the timing of stream flows, extreme or unusual weather events, rising sea levels, increased occurrences of vector-borne diseases and impacts on forest health. An initial set of strategies for both mitigation and adaptation have been set forth in the OGWC's Roadmap to 2020, the OCCRI's Climate Assessment Report, and the Oregon Climate Change Adaptation Framework prepared by multiple state agencies.[5]

The GHG goals established by HB 3543 call for Oregon to: (1) arrest the growth of Oregon's GHG emissions and begin to reduce them by 2010; (2) achieve GHG levels that are 10 percent below 1990 levels by 2020, and (3) achieve GHG levels that are at least 75 percent below 1990 levels by 2050. The State reports that the 2010 goal has been met, at least with respect to slowing emissions growth. Yet those same reports acknowledge that Oregon is not even close to being on a trajectory to meet its 2020 or 2050 goals without implementation of significant new initiatives.[6] Serious attention to the emissions associated with forest practices must be part of the equation.

Yet emissions from forest practices are simply assumed to be zero

The GHG emissions from forest practices in Oregon have not been measured other than a single assessment in 2002.[7] That assessment – prepared by an intern with the Department of Energy – relied on a metric called "carbon flux." Carbon flux measures how many metric tons of carbon dioxide are emitted by the forest sector each year taking emissions from clearcutting, decay of forest products, and wildfire into account together with the amount of carbon absorbed by residual forest cover. The key conclusion of that assessment was that GHG emissions from the forest sector were, for the most part, net negative for most of the period assessed (1989-2002) and therefore the timber industry should be exempted from further scrutiny. Indeed, the Roadmap to 2020 makes this bold assertion: "Oregon's forests are a carbon sink, capturing more carbon than they release. As such, Oregon's forests and its forest sector have and will continue to contribute to the goal of achieving reductions in greenhouse gas emissions by remaining a robust and sustainable sector in Oregon."[8]

The problem is that this assertion is not supported by the data. To understand why, it is important to quickly review how GHG emissions from the forest sector are typically addressed. Rightly so, the 2002 assessment considered most of the standard components relevant to forest

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[5] Each of these documents is available online via the OGWC's site at: http://www.keeporegoncool.org/content/roadmap-2020.

[6] Drumheller, Bill. 2014. Oregon Greenhouse Gas Emissions and Recent Climate Change Developments. Salem, OR: Interagency Sustainability Coordinators Network, Oregon Department of Energy.

[7] Kelly, Peter. 2013. A Greenhouse Gas Inventory of Oregon's Forests. Salem, OR: Oregon Department of Energy, Oregon Global Warming Commission.

[8] Oregon Global Warming Commission (OGWC). 2010. Interim Roadmap to 2020. Salem: OGWC.

sector GHG emissions in best practice protocols established at the international, national, and state level.[9] These are summarized in Table 1, below. There are three major entries in the balance sheet, and their total is referred to as carbon flux – which can be positive or negative depending on many factors.

Table 1: Greenhouse Gas Emissions from Forest Practices

Major component	Sub components	Effect on emissions
Net ecosystem productivity		-
	• Forest cover	-
	• Forest degradation	+
	• Climate	+/-
Emissions from timber harvest		-
	• Volume harvested	+
	• Longevity of forest products	-
	• Efficiency of harvest and processing	-
	• Substitution of non-wood alternatives	+/-
	• Application of forest chemicals	+
Emissions from wildfires		+
	• Acreage burned	+
	• Intensity of burn	+
	• Density of stored carbon	+

Net ecosystem productivity

The first component influencing emissions is net ecosystem productivity (NEP). NEP is the net effect of photosynthetic carbon uptake and release of carbon to the atmosphere from respiration by autotrophs (plants) and heterotrophs (animals and fungi).[10] NEP is inversely correlated with emissions because higher NEP means more carbon uptake. NEP is dependent on many factors, but the most important are the amount of forest cover, the extent of forest degradation, and climate variability. More forest cover means more carbon uptake.[11] But if that forest cover is degraded – for example, by skid trails and roads left over from earlier logging or intensive

[9] See, e.g. (1) Watson, Robert T., Kan R. Nobel, Bert Bolin, N.H. Ravindranath, David J. Verardo and David J. Dokken (Eds.) 2000. Intergovernmental Panel on Climate Change - 2000. Land Use, Land Use Change and Forestry. UK: Cambridge University Press; (2) H.S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe (Eds.) 2006. Intergovernmental Panel on Climate Change. 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Hayama, Kanagawa: The National Greenhouse Gas Inventories Programme; (3) U.S. Environmental Protection Agency (EPA). 2015. Inventory of Greenhouse Gas Emissions and Sinks: 1990-2013. EPA 430-R-15-004.

[10] Turner, David, Michael Guzy, Michael Lefsky, William D. Ritts, Steve Van Tuyl, and Beverly E. Law. 2004. Monitoring Forest Carbon Sequestration with Remote Sensing and Carbon Cycle Monitoring. *Environmental Management* 33(4): 457-466.

[11] The standard definition for forest cover is trees of at least 5 meters in height (16.4 feet) covering at least 30% of the ground as seen from above.

thinning – NEP can be significantly less.[12] And with respect to climate variability, warmer and drier years are associated with significantly lower NEP in Northwest forests.

Emissions from timber harvest

When a forest is logged, emissions are generated through a number of channels, the most important being (a) the release of carbon once stored in trees when slash, needles, roots and stumps decay or are burned on site; (b) carbon released when soils are disturbed and eroded; (c) carbon released when wood waste is generated and then decays during processing, and; (d) carbon released as wood products decay in landfills.[13] Transportation and heavy equipment use also generates emissions, but these are counted in other sectors (transportation and industry). As harvest volumes increase – whether through additional acres harvested or harvesting of larger, older trees – emissions increase accordingly. But not all stored carbon is lost at once. A proportion of carbon once stored by forests is stored in the wood products made from them. Longer-lived wood products, such as structural wood for homes and buildings, last longest while short-lived paper products or biomass for energy releases stored carbon quickly. Longer rotations (years between timber harvest) allow trees to grow larger and generate more valuable long-lived wood products and thus have a significant effect on emissions.[14]

Estimates of the amount of stored carbon lost from a given acre logged vary depending on these factors. Ingerson (2009) completed one of the most comprehensive reviews on this issue, tracing the amount of the original live tree volume (and thus carbon stored) remaining after logging, primary processing, secondary processing, and construction.[15] Compiling and calibrating estimates from a variety of sources, she concluded that these losses amount – on average – to 82% of the original live tree volume. In other words, when a site is logged and the wood converted into long-lived wood products, only 18% of the original carbon stores are preserved, and then only for a few decades at most before those longer lived wood products start to decay. The remaining 82% of the carbon stocks are released into the atmosphere in a relatively short period of time. This value is essentially 100% for short-lived wood and paper products.

Some have argued for consideration of the product substitution effect in accounting for emissions associated with timber harvest.[16] For example, if wood beams are used in place of steel, overall emissions from fossil fuel combustion may drop. However, the substitution effect is ambiguous. There are now alternatives, such as use of bamboo for flooring, that can be produced with essentially zero emissions and so the presumption that emissions increase as wood substitutes

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[12] Pearson, Timothy R.H., Sandra Brown and Felipe M. Casarim. 2014. Carbon emissions from tropical forest degradation caused by logging. *Environmental Research Letters* 9 (2014): 034017 (11pp). doi:10.1088/1748-9326/9/3/034017.
[13] Harmon, M.E., W.K Ferrel and J. F. Franklin. 1990. Effects on carbon storage of conversion of old – growth forests to young forests. *Science* 247:699-702.
[14] Lippke, Bruce, Elaine Oneil, Rob Harrison, Kenneth Skog, Leif Gustavvsson and Roger Sathre. 2011. Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. *Carbon Management* 2(3): 303-333.
[15] Ingerson, A. 2009 Wood Products and Carbon Storage: Can Increased Production Help Solve the Climate Crisis? Washington, D.C.: The Wilderness Society.
[16] Lippke et al. (2011), Note 14.

increase cannot be substantiated.[17] Finally, accounting protocols also acknowledge the emissions from carbon intensive pesticides, herbicides, and fertilizers applied on forestlands. As more land is harvested through even-aged clearcutting techniques, more chemicals are required and more emissions are generated relative to harvest techniques that rely on natural regeneration.

Emissions from wildfires

The third major component that contributes to GHG emissions from forests is the emissions associated with forest fires. Emissions from forest fires consist of carbon dioxide, methane and nitrous oxide. Forest fires can be broken into two major types: (1) low-intensity fires set intentionally for a variety of conservation goals, and (2) unintentional wildfires of mixed intensities triggered by lighting or human causes that are typical of large fires in Oregon's dry forest provinces. Typically, it is only the latter emissions that are tracked by GHG accounting protocols, and even then only for fires that burn over 50% of crown cover. Emissions associated with unintentional fires vary considerably, and depend upon the total acreage burned, intensity, and the density of carbon stored. In any given year, emissions from wildfires can represent a significant percent of overall emissions for a given state or region.

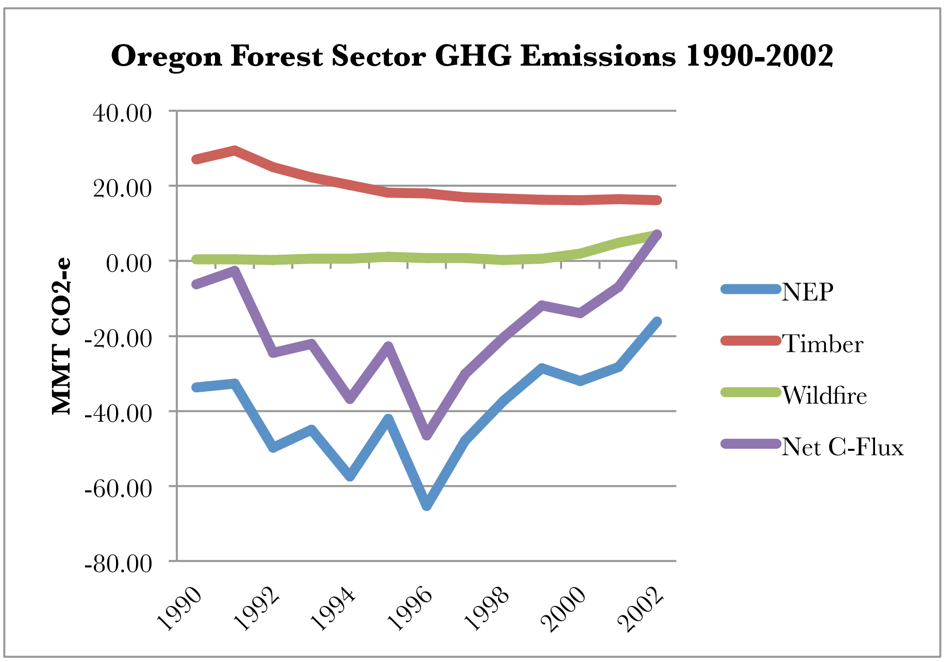

Figure 1: Oregon GHG Inventory Results, 3-year rolling averages

As previously noted, decision makers in Oregon assume that the net effect of these GHG accounting entries are on average, negative – meaning that the forest sector as a whole is responsible for sequestering more carbon dioxide than it emits in a given year. The assumption is based on a single Greenhouse Gas Inventory of Oregon's Forests published in 2009 for the years 1990 to 2002.[18]

As shown in Figure 1, the GHG inventory showed a negative forest carbon flux for most study years but a positive spike of 7.00 million metric tons (MMT) CO_2-e in 2002 (3-year average). Emissions from timber harvest (including carbon stores in long-lived wood products) dropped

[17] Van der Lugt, P., J.G. Vogtlander, J.H. van der Vegte, J.C. Brezet. 2011. Life Cycle Assessment and Carbon Sequestration: The Environmental Impact of Industrial Bamboo Products. Zwagg, the Netherlands: MOSO Research and Development Center.
[18] Kelly, Peter 2013, Note 7.

during the early 1990s but have leveled off since that time in the 16 MMT CO2-e range. Emissions from wildfire were insignificant for most years but increased for 2001 and 2002 when they rose to 6.87 MMT CO2-e in 2002, largely attributable to the Biscuit Fire in southern Oregon. Notably, these findings are similar to related studies using alternative forest carbon datasets that showed the loss of "high-biomass" (older) forests to logging on private lands exceeded that of fire across all ownerships.[19]

While the 2009 GHG analysis for Oregon forests is certainly useful, it was never designed for use as policy, nor should it be. There are too many unwarranted assumptions and accounting irregularities that were never vetted. Moreover, even if the inventory were accurate for 1990 to 2002, it notes an alarming trend that continues today – that of rapidly decreasing carbon sequestration and increasing carbon flux. Due to industrial forest practices and a warming and drying climate, conditions in all likelihood have worsened since that time and imply that net emissions are consistently positive, and significant, at least on industrial forestlands.

The timber industry is, in fact, a significant source of greenhouse gas emissions

There are three key assumptions that render the 2009 GHG analysis for Oregon forests invalid for use by policy makers. First, the inventory assumes that the dramatic trend of steadily declining NEP and increasing carbon flux noted between 1996 and 2002 was an aberration. The report simply concludes "[t]he end of the study period indicates that Oregon's forests became a source rather than a sink for CO2 , due to a combination of low forest growth (due to dry weather) and high fire emissions. The 2001 and 2002 removals from the carbon pool are abnormal, however, and it is expected that future years will show that Oregon's forests revert to being a carbon sink."[20]

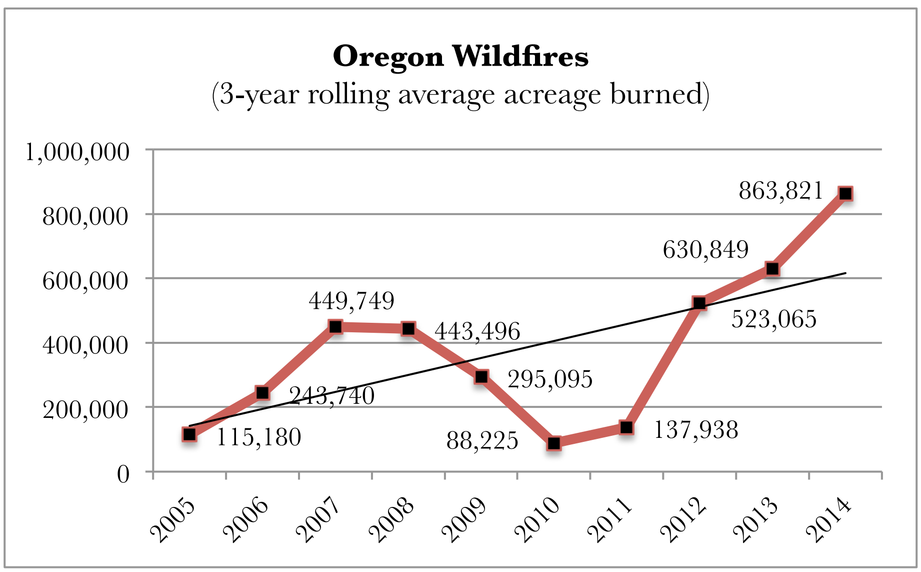

Figure 2: Oregon Wildfire Trends

Almost every future year since the analysis was completed has been dry, warm, and characterized by more frequent wildfires (Figure 2). Statewide precipitation totals were well below average in 2004, 2007, 2008, 2009, 2013 and 2015 to date.[21] The year 2015 is

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[19] Krankina, O.N., D.A. DellaSala, J. Leonard, and M. Yatskov. 2014. High-biomass forests of the Pacific Northwest: who manages them and how much is protected? *Environmental Management* 54:112-121.
[20] Kelly, 2013, Note 18 at 5.
[21] NOAA maintains a useful web portal for monitoring statewide precipitation and temperature trends. These data are available at: http://www.ncdc.noaa.gov/cag/time-series/us.

currently the fifth driest year since record keeping began. Temperatures through the entire period (2002-2015) have been well above historical averages. Rather than an aberration, the trend towards lower NEP and higher carbon flux as climate change continues to unfold is in all likelihood an unfortunate long-term trend.

Secondly, and an assumption which had no actual basis in fact was that forest cover is static and that land use changes make no significant contribution to emissions. The inventory states: "[t]he model fixes the total forested land in Oregon at the 2001 estimates for the purposes of scaling up from site surveys. Forested land lost due to land- use change has been constant and small relative to Oregon's total forest land, and so is not considered a primary driver of the emissions profile of Oregon's forests."[22]

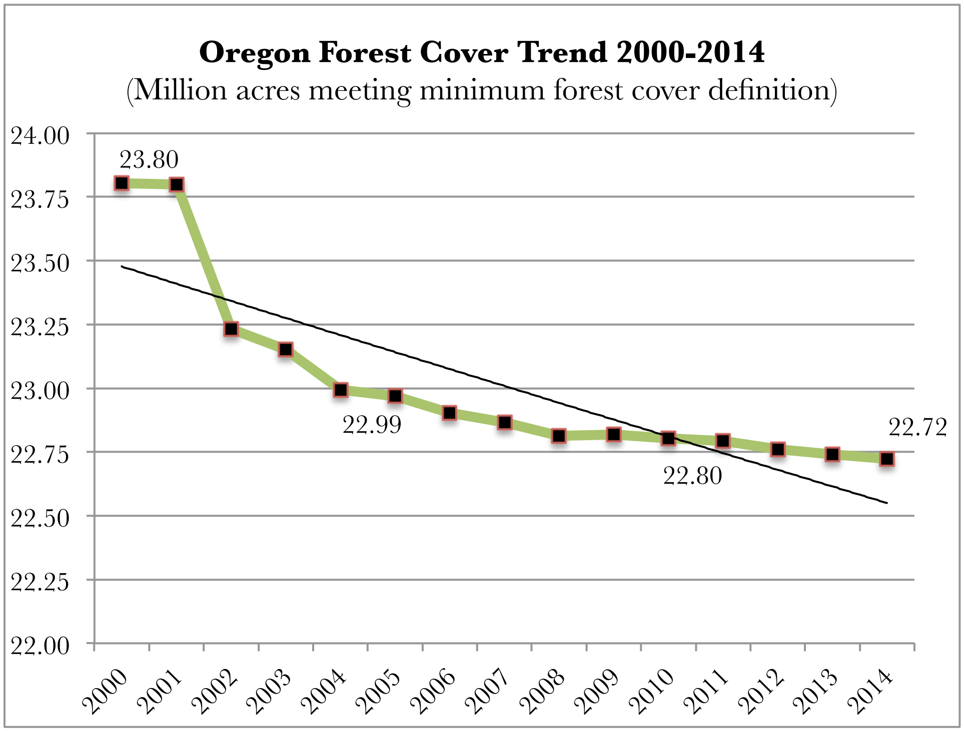

Figure 3: Oregon Forest Cover Trends

In other words, the report simply assumes that since the amount of land in Oregon zoned for forest uses has changed little, that actual forest cover has changed little as well. This is a serious source of error that results in significant over-estimation of NEP since loss of forest cover (deforestation) irrespective of how land is zoned is a significant source (roughly 10%) of global GHG emissions. The reality is that Oregon, since 2000, has lost over 1.08 million acres of forest cover (Figure 3) according to annual satellite measurements available through the World Resources Institute's Global Forest Watch database.[23] More than half of this loss took place on state and private timberlands in western Oregon where it is almost entirely attributable to clearcutting far beyond the rate of forest regrowth.[24] Other factors include fire and urbanization.

The third erroneous aspect of the official inventory, is the lumping of all forestland owners into one aggregate category called the "forest sector" when in fact from a proper GHG accounting stance it is important to separate out emissions from the timber industry from the carbon

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[22] Kelly, 2013, Note 7 at 11.
[23] The online portal for the Global Forest Watch program is accessible at: http://www.globalforestwatch.org/.
[24] Talberth, John and Erik Fernandez. 2015. Deforestation, Oregon Style. Lake Oswego, OR: Center for Sustainable Economy. Available online at: http://sustainable-economy.org/deforestation-oregon-style/.

sequestered on lands protected and sustainably managed by non-profit organizations, small landowners, and government. These protected lands exist and sequester as much carbon as they do despite, and not because of the timber industry's management decisions. The irony of the current assumption is that the segment of the forest sector that is responsible for substantial GHG emissions is able to "take credit" for the emissions reductions of actors outside the industry – actors that are in constant political struggles to protect forests from industry's reach. This is directly analogous to a scenario whereby the fossil fuel industry could deduct from its emissions inventory the carbon sequestered by lands and waters inside the Arctic National Wildlife Refuge, marine sanctuaries, and other protected marine ecosystems.

Taken together, these erroneous assumptions included in the 2009 GHG inventory underscore the necessity of accurate annual accounting of the timber industry's GHG emissions in Oregon rather than continuing to assume these emissions are negligible. Here, we completed a preliminary analysis of such emissions over the 2000-2014 timber period for State and private forestlands in western Oregon. We estimated gross emissions from three sources: (1) timber harvest; (2) lost sequestration capacity, and (3) forest pesticides, herbicides, and fertilizers. We then adjusted these gross emissions figures to account for NEP on lands not affected by timber harvest during this time period to estimate average annual carbon flux. We completed this analysis for industrial timberlands alone and for all State and private forestlands together in western Oregon.

Estimated emissions from timber harvest

To estimate emissions associated with timber harvest, we overlaid two spatial data sets depicting forest carbon stocks in 2000 and another depicting forest cover loss attributable to clearcutting between 2000 and 2014. For initial forest carbon stocks, we relied on the Woods Hole Research Center's high resolution National Biomass and Carbon Dataset for the year 2000 (published in 2012) the first ever spatially explicit inventory of its kind.[25] The dataset was produced as part of a project funded under NASA's Terrestrial Ecology Program with additional support from the Landscape Fire and Resource Management Planning Tools Project (LANDFIRE). The project has generated a high-resolution year-2000 baseline estimate of basal area-weighted canopy height, aboveground live dry biomass, and standing carbon stock for the conterminous United States. The inventory estimates carbon stocks per square meter with a 90 square meter resolution.

For forest cover change, we made use of satellite-based data made available by the World Resources Institute (WRI), through its Global Forest Watch (GFW) Program.[26] The GFW hosts a

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[25] 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), U.S.A., 2000. Data set. Available on-line at http://daac.ornl.gov from ORNL DAAC, Oak Ridge, Tennessee, U.S.A. http://dx.doi.org/10.3334/ORNLDAAC/1081.

[26] Hansen, M. C., P. V. Potapov, R. Moore, M. Hancher, S. A. Turubanova, A. Tyukavina, D. Thau, S. V. Stehman, S. J. Goetz, T. R. Loveland, A. Kommareddy, A. Egorov, L. Chini, C. O. Justice, and J. R. G. Townshend. 2013. "High-Resolution Global Maps of 21st-Century Forest Cover Change." Science 342 (15 November): 850–53. Data available on-line from: http://earthenginepartners.appspot.com/science-2013-global-forest.

platform enabling users to download and analyze Landsat-based forest change data dating back to 2001.[27] As with the forest carbon dataset, each pixel measures an area of 90 square meters, making the data sets comparable. After removing federal lands from the analysis as well as forest cover loss due to wildfire, we calculated the amount of stored carbon removed by clearcutting on state and private lands during the 2000-2014 period. Appendix 1 shows carbon stock removals by county. In accordance with the GHG accounting protocols discussed above, we then deducted 18% of the carbon stock removed to account for carbon that is possibly still stored in longer-lived wood products, and then converted the residual carbon figure to CO2-e units.

Since 2000, the GFW data indicates that nearly1.6 million acres had been clearcut. According to the Woods Hole dataset, these clearcut acres once stored at least 89 MMT-C, roughly 56 metric tons per acre.[28] After converting to CO2-e and deducting the amount possibly stored in longer-lived wood products (58.49 MMT-CO2-e) this translates into 266.47 MMT-CO2-e emissions associated with timber harvest – an average of 19.03 MMT CO2-e per year over the fourteen-year period. Of this amount, 14.28 MMT CO2-e is attributable to industrial forestlands, while 4.75 MMT CO2-e is attributable to the state and small forestland owners. This apportionment is based on the ratio of industrial timber harvests to the overall harvest from state and private forestlands in western Oregon over the past several years. That ratio is roughly 0.75.[29]

Estimated emissions from lost carbon sequestration capacity

Because the rate of clearcutting on state and private timberlands has exceeded the rate of forest regrowth over the 2000-2014 period, deforestation is occurring. During this time period, the GFW dataset indicates that nearly 1.6 million acres have been clearcut. During this same time period, just about 1.1 million acres of previously clearcut land attained minimum tree height and canopy closure to meet minimum forest cover definitions. Thus, western Oregon has lost roughly 500,000 acres of forest cover, roughly half the loss (1.08 million acres) for the State as a whole over the same time period.[30] This is resulting in significantly less net ecosystem productivity (or carbon sequestration) than would otherwise occur if timber harvests were sustainable and not resulting in a net loss of forest cover. And as recognized by the Legislature and climate policy makers statewide, "losing this potential to sequester carbon will have a significant negative effect on the reduction of carbon levels in the atmosphere."[31]

Using carbon sequestration estimates compiled by researchers studying NEP in both the Coast Range and Cascades we made a rough estimate of these timber industry emissions. For clearcuts and newly established plantations between one and 13 years old sequestration is negative – meaning these lands are emitting more carbon dioxide than they absorb. NEP for these lands was estimated to range between -6 and -146 grams per cubic meter per year ($gC/m^2/yr$). To be conservative, and because emissions from timber harvest already account for the biomass

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[27] WRI's Global Forest Watch portal can be accessed here: http://www.globalforestwatch.org
[28] Researchers at Woods Hole and others acknowledge this as a conservative figure. The USDA's Forest Inventory and Analysis (FIA) team in 2010 estimated net storage on private lands as 75 metric tons per acre.
[29] Department of Forestry, Western Oregon Harvests, Years 2010-2014.
[30] Talberth and Fernandez, Note 24.
[31] ORS 468A.200(5).

decaying on-site we simply assume it is zero. For newly established plantations between 14 and 29 years the NEP estimates are 389 $gC/m^2/yr$ in the Coast Range and 254 $gC/m^2/yr$ in the western Cascades. And for stands 30-99 years old the estimates were 299 and 354 $gC/m^2/yr$ for the Coast Range and Cascades, respectively.[32]

We used the 30-99 age class NEP figures to represent what the level of sequestration would be on lands deforested (i.e. experiencing forest cover loss) during the 2000 and 2014 timber period or still too young (aged 1-13 years) to achieve positive NEP. We then accounted for the increase in sequestration associated with forest cover that was reestablished during this time period and old enough (14+ years) to achieve positive sequestration status to determine the net effect. We applied the mean NEP estimates for the 14-29 year age class to these lands. Both forest loss and forest gain figures were segregated between the Coast Range and Cascades to make use of the different NEP estimates, and also disaggregated to the county level. The results indicate that loss of sequestration capacity associated with clearcutting beyond the rate of forest regrowth on state and private lands in western Oregon is responsible for 3.57 MMT-CO2-e emissions per year at this time above and beyond the emissions associated with timber harvest. Of this amount, 2.68 MMT-CO2-e is attributable to practices on industrial timberlands while 0.89 MMT CO2-e is attributable to state and small forestland owners.

Estimated emissions from forest pesticides, herbicides, and fertilizers

Estimating emissions from forest pesticides, herbicides, and fertilizers is difficult because there is no regular reporting of annual applications. There are, however, a few periodic estimates we can make use of as well as the rates of application recommended on product labels. For forest chemicals, the three most common applied regularly include Atrazine, 2,4 D, and Glyphosate. There are at least two applications of this chemical cocktail on all lands newly clearcut. Atrazine is typically applied at a rate of 2 kilograms (kg) per acre while both 2,4 D and Glyphosate are applied at a rate of 0.91 kg per acre. The most ubiquitous fertilizer applied to forestlands is Urea, and is applied at a rate of 91 kg per acre.

The carbon content of these chemicals and fertilizers has been calculated and published in the scientific literature. The relevant units are kilograms carbon per kilogram product (kg/kg). The respective values are reported to be 3.8 kg/kg for Atrazine, 1.7 kg/kg for 2,4 D, 9.1 kg/kg for Glyphosate, and 1.3 kg/kg for Urea.[33] In terms of annual acres treated, we assume there have been two applications on each acre of newly clearcut and replanted land in the 2000-2014 period for chemicals and continuation of an overall application rate on state and private forestlands (95,000 acres per year) for Urea recently documented by researchers at the U.S. Geological Survey.[34] Taken together, this suggests total emissions of 56,710 metric tons CO2-e annually associated with chemicals and fertilizers on state and private forestlands in western Oregon. Of

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[32] All these NEP estimates were taken from Turner, David P., Michael Guzy, Michael A. Lefsty, William D. Ritts, Steve Van Tutyl and Beverly E. Law. 2004. Monitoring forest carbon sequestration with remote sensing and carbon cycle modeling. *Environmental Management* 33 (4): 457-466.
[33] Lal, R. 2004. Carbon emissions from farm operations. *Environmental International* 30: 981-990.
[34] Anderson, Chauncey W. 2002. Ecological Effects on Streams from Forest Fertilization – Literature Review and Conceptual Framework for Future Study in the Western Cascades. Portland, OR: US Department of Interior, US Geological Survey.

this amount, 42,532 metric tons CO2-e is attributable to applications on industrial forestlands while 14,178 metric tons CO2-e is attributable to state and non-industrial forestland owners.

Net ecosystem productivity on lands not affected by timber harvest

The biggest unknown in this analysis is the amount of carbon sequestered (NEP) by lands not affected by timber harvest. Under a proper GHG accounting framework, emissions from timber harvest and chemicals for any particular owner should be offset by the amount of carbon sequestered on that owner's residual lands. But there are no reliable NEP estimates for Oregon's forests that distinguish between categories of private forestland ownership (i.e State, industrial vs. non-industrial) and as they stand now, GHG accounting frameworks inappropriately lump all forest sector owners together. However, the apportionment of NEP between owners can at least be estimated based on the amount of land they manage and maintain as forest cover.[35] To do this, we relied on two separate sources of NEP information.

The first is the data relied upon in the 2009 inventory, prepared by the Oregon-California (ORCA) carbon assessment project at the Oregon State University.[36] For Oregon's 28 million acres of forestland as a whole the three-year rolling average NEP for 2002 was calculated as -16.50 MMT-CO2-e. It is reasonable to assume that this average NEP did not increase since that time due to consistently warmer, drier conditions. We can therefore use this figure to "distribute" NEP among owners. For state and private lands in western Oregon, if we first back out acres on industrial vs. state and non-industrial that were clearcut or newly replanted during the 2000-2014 period (because these lands are net emissions sources) and then proportionally allocate NEP among these ownership categories,[37] it suggests an average of -1.12 MMT-CO2-e for private industrial forestlands in western Oregon, and -2.15 MMT-CO2-e on forestlands managed by state and non-industrial owners.

The second method relies on information from a somewhat more recent (2003-2007) NEP analysis for forests in northern California, Oregon and Washington within the range of the northern spotted owl.[38] While the analysis distinguishes between private and public ownerships, it does so at this regional scale and so may miss significant differences in forest practices among the

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[35] The method of apportionment probably over-estimates NEP on industrial forestlands since these lands are managed far more intensively that lands managed by the state or other owners. But in the absence of specific data, we err on the conservative side.

[36] Oregon State University, Department of Forest Ecosystems and Society. ORCA (Oregon and California) Synthesis of Remote Sensing and Field Observations to Model and Understand Disturbance and Climate Effects on the Carbon Balance of Oregon and Northern California. Beverley Law, Principal Investigator. Project summary online at: http://terraweb.forestry.oregonstate.edu/orca.htm.

[37] Forestland ownership acres in western Oregon and (%) of Oregon's total are: Federal - 7.9 million acres (28%); State and other public – 1 million acres (4%); Private industrial – 3.2 million (11%); private non-industrial – 3.2 million (11%). Source: Campbell, Sally, Dave Azuma and Dale Weyermann. 2004. Forests of Western Oregon: An Overview. PNW-GTR-525. Portland, OR: USDA Forest Service, Pacific Northwest Research Station.

[38] Turner, David P., William D. Ritts , Zhiqiang Yang, Robert E. Kennedy, Warren B. Cohen, Maureen V. Duane, Peter E. Thornton and Beverly E. Law. 2011. Decadal trends in net ecosystem *production and net ecosystem carbon balance for a regional socioecological system. Forest* Ecology and Management 262 (2011): 1318-1325.

states. For private lands, the analysis reports an average NEP of -42.24 MMT-CO2-e for 18.7 million acres. Distributing this proportionally suggests an average of -4.43 MMT-CO2-e on industrial forestlands in western Oregon and -8.48 MMT-CO2-e on forestlands managed by State and non-industrial owners.

Estimates of average annual carbon flux across ownerships 2000-2014

Tables 2 and 3, below, tie all this information together. For two ownership categories – industrial and State/non-industrial forestland owners – we report annual average emissions from timber harvest, deforestation, and forest chemicals and fertilizers as well as adjustments to account for carbon stored in long lived wood products and sequestered on residual lands not affected by timber harvest during the 2000 to 2014 period. We use two different NEP assumptions as previously discussed based on the ORCA analysis (Table 2) and the Turner et al. (2011) analysis (Table 3).

As shown in Table 2, both industrial forestlands and those managed by state and non-industrial owners are likely a significant source of carbon dioxide emissions at 19.39 MMT-CO2-e using the ORCA NEP assumption, but industry emissions (15.88 MMT-CO2-e) outpace those of state and other private owners (3.69 MMT-CO2-e) by a factor of 4.3. As shown in Table 3, only industrial forestlands are likely a significant source of carbon dioxide emissions at 12.57 MMT-CO2-e using the Turner et al. (2011) NEP assumption, but state and non industrial owners are a net emissions sink at -2.82 MMT CO2-e. Combined, overall emissions from state and private forestlands in western Oregon are 9.75 MMT CO2-e. How do these emissions stack up against emissions of other sectors?

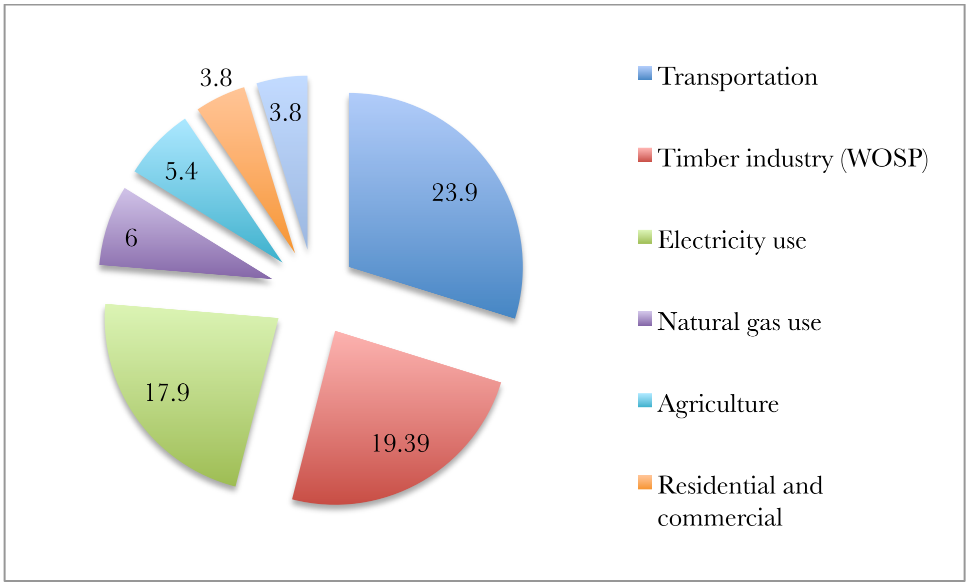
Table 2: Carbon Flux Annual Average 2000 – 2014 with ORCA NEP
(Western Oregon state and private forestlands MMT-CO2-e)

GHG accounting component	Industry	State/non-industry	Total
Emissions from timber harvest	17.41	5.80	23.21
Emissions from lost carbon sequestration	2.68	0.89	3.57
Emissions from chemicals and fertilizers	.04	0.2	.06
Net wood product sink	(3.13)	(1.05)	(4.18)
Net ecosystem productivity	(1.12)	(2.15)	(3.27)
Net carbon flux (emissions)	15.88	3.69	19.39

Table 3: Carbon Flux Annual Average 2000 – 2014 with Turner et al. NEP
(Western Oregon state and private forestlands MMT-CO2-e)

GHG accounting component	Industry	State/non-industry	Total
Emissions from timber harvest	17.41	5.80	23.21
Emissions from lost carbon sequestration	2.68	0.89	3.57
Emissions from chemicals and fertilizers	.04	.02	.06
Net wood product sink	(3.13)	(1.05)	(4.18)
Net ecosystem productivity	(4.43)	(8.48)	(12.91)
Net carbon flux (emissions)	12.57	(2.82)	9.75

Figure 4: Comparison with Oregon Greenhouse Gas Emissions (2012 estimates by sector in MMT CO2-e)



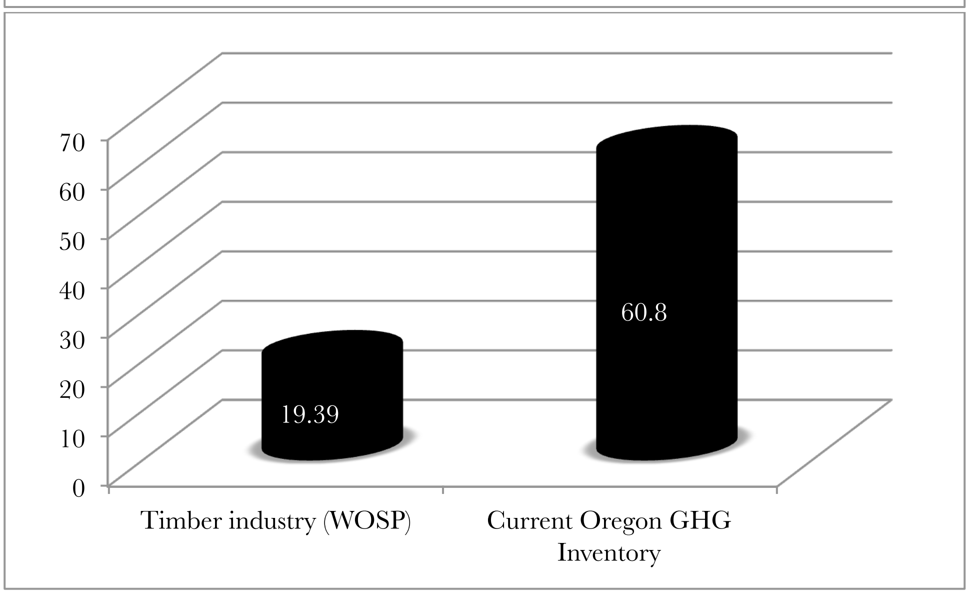


Figure 4 puts the data from Table 2 into perspective. The pie chart compares GHG emissions from western Oregon state and private forestlands (WOSP) with emissions from five other sectors last estimated in 2012.[39] Under the ORCA NEP assumption, the timber industry would rank second (19.39 MMTCO2-e) overall, behind transportation (23.9 MMT-CO2-e) and ahead of the industrial, residential, commercial, agriculture, natural gas, and electricity sectors. Emissions from Oregon power plant coal combustion in 2012 were 2.65 MMT-CO2-e according to DEQ. Therefore, this analysis suggests that the timber industry has the equivalent emissions of up to 7

[39] Oregon Department of Environmental Quality (DEQ). Oregon Greenhouse Gas Emissions Data 1990 through 2012. Available online at: http://www.oregon.gov/deq/AQ/Pages/Greenhouse-Gas-Inventory-Report.aspx.

coal-fired power plants. These emissions are also equivalent to putting 2-4 million new vehicles on the road.[40] If the timber industry's emissions were counted, Oregon's overall emissions would rise to 80.19 MMT-CO2-e, far surpassing the historical peak emissions of 71.9 MMT CO2-e in 1999.

The column chart compares timber industry WOSP emissions with GHG emissions estimated in 2012 for the five other sectors. The timber industry's emissions represent 32% of the current state's inventory under the ORCA NEP analysis. This drops to 16% of state emissions under the Turner et al. (2011) NEP approach. Either way, it is likely that current emissions from logging activities on state and private lands in western Oregon is a significant source that should be addressed and mitigated in the state's climate agenda.

State and private forest practices undermine climate adaptation goals

In addition to generating significant GHG emissions, clearcutting on state and private forestlands in western Oregon is undermining climate adaptation goals that seek to protect rural communities from the risks of wildfires, floods and landslides, degradation of water supplies, and loss of critical natural resources like fisheries and forest health.[41] Current forest practices that include short rotations, extensive timber plantations, clearcutting, and a rapid rate of harvest maintain lands in a high-risk condition for all these threats due to homogenization of fuel loads and retention of logging slash on site among other factors.[42]

Figure 5: Dense, young timber plantations exacerbate fire risk.

Rapid clearcutting on state and private forestlands has transformed vast areas of more fire-resistant native forests into open brush fields and dense young tree plantations that are more prone to explosive, hot crown fires that kill more trees and spread fires faster and farther.[43] As noted by the US Forest Service in an evaluation of the 2003 fire season "plantations experienced a disproportionately high amount of stand- crown

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[40] Calculated using the EPA's greenhouse gas equivalences calculator at: http://www2.epa.gov/energy/greenhouse-gas-equivalencies-calculator

[41] Department of Land Conservation and Development (DLCD). 2010. The Oregon Climate Change Adaptation Framework. Salem, OR: DLCD.

[42] Odion, D.C., J.R. Strittholt, H. Jiang, E. Frost, D.A. DellaSala, and M. Moritz. 2004. Fire severity patterns and forest management in the Klamath National Forest, northwest California, USA. *Conservation Biology* 18:927-936.

[43] Odion, D.C., J.R. Strittholt, H. Jiang, E. Frost, D.A. DellaSala, and M. Moritz. 2004. Fire severity patterns and forest management in the Klamath National Forest, northwest California, USA. *Conservation Biology* 18:927-936

fires as compared to older, unmanaged forests… Plantations had a tendency to increase the rate of fire spread and increased the overall area of stand-replacement fire effects by spreading to neighboring stands."[44]

The vast network of clearcuts and logging roads present on state and private forestlands also presents a big risk for landslides, especially during extreme precipitation events such as the 1996 floods. Under almost all climate change scenarios for Oregon, the frequency of these events will increase even if overall precipitation patterns remain unchanged. Maintenance of strong root systems is an important factor in stabilizing soils during these events. Clearcutting reduces the strength of these root systems dramatically, and thus is a major factor in increased landslide risk.[45] Logging roads channel water runoff and result in debris torrents that can travel many miles downstream, pick up momentum, and cause widespread destruction.[46] Studies indicate that clearcuts exhibit landslide rates up to 20 times higher than the background rate. Near logging roads, landslide rates are up to 300 times higher than forested areas.[47]

Increasing water temperatures are another climate change challenge worsened by current practices on state and private lands. On these lands, practices that are consistent with the Oregon Forest Practices Act (OFPA)'s no-cut buffer restrictions of 20 feet or less for most streams and rivers can be expected to increase mean water temperatures by 2.61 °F over and above ambient warming due to climate change.[48] The consequences of warmer waters were apparent in the summer of 2015 when over half of the Columbia River basin's 500,000 spawning sockeye salmon were thought to have died due to stream temperatures that passed the 70 °F threshold of concern. Similar heat related die-offs have been noted for the Klamath River.

Policy reforms can transform forest practices from an emissions source to a sink

Oregon's State and private forests are faced with a significant choice in policy direction. On the one hand, the State can no longer avoid emissions from logging as these are likely 7 times greater than the annual power plant emissions from the Boardman coal fired plant and will be a significant impediment to achieving the State's emissions goals. On the other hand, State and private forests contain significant amounts of stored and sequestered carbon that could be part of the solution to reducing GHG emissions. There are many short and long term policy reforms the State can pursue to address this urgent challenge and help expedite the transition of State and private forestlands from a significant net source to an increasingly vital sink for GHG emissions

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[44] USDA Forest Service, Umpqua National Forest. 2003. Wildfire Effects Evaluation Project. Roseburg, OR: USDA Forest Service, Umpqua National Forest.

[45] Schmidt, K.M, J. J. Roering, J.D. Stock, W.E. Dietrich, D.R. Montgomery and T. Schaub. 2001. "The variability of root cohesion as an influence on shallow landslide susceptibility in the Oregon Coast Range." *Can. Geotech. J* (38): 995-1024.

[46] Swanson, F. J., J. L. Clayton, W. F. Megahan and G. Bush. 1989. "Erosional processes and long-term site productivity," pp. 67-81 in *Maintaining the Long-Term Productivity of Pacific Northwest Forest Ecosystems*. D. A. Perry, R. Meurisse, B. Thomas, R. Miller, J. Boyle, J. Means, C.R. Perry, R. F. Powers, eds. Portland, Oregon: Timber Press.

[47] Heiken, Doug. 2007. Landslides and Clearcuts: What Does the Science Really Say? Eugene, OR: Oregon Wild.

[48] Oregon Department of Forestry. 2015. Riparian Rule Analysis: Analysis of riparian prescriptions and expected changes in restrictions. Salem, OR: Oregon Department of Forestry.

in the years ahead. It is beyond the scope of this report to provide an in-depth treatment, but five key areas of reform are immediately obvious.

1. Account for the timber industry's emissions.

The timber industry should be treated like any other sector in Oregon that has significant GHG emissions. Those emissions should be monitored and included in the state's regular inventory, last published in 2012. HB3543 requires the Oregon Global Warming Commission to track and evaluate "[g]reenhouse gases emitted by various sectors of the state economy…" as well as "the carbon sequestration potential of Oregon's forests" on a continuous basis and report the results of this monitoring to the legislature in odd numbered years.[49] Including the timber industry's emissions and the adverse impacts of unsustainable forest practices on the carbon sequestration and carbon storage potential of Oregon's forests will be an important first step in achieving the forest sector goals outlined in the Roadmap 2020. In calculating these emissions, and as demonstrated here, it is of the utmost importance to distinguish between various forestland ownerships so that those owners who are responsible for emissions and lost sequestration and storage capacity are no longer able to mask the impacts of their management activities by, in effect, claiming credit for carbon sequestered or stored on forests the industry had nothing to do with protecting.

2. Promote alternatives to short rotations, clearcutting, and chemicals.

Oregon's climate legislation also requires the Oregon Global Warming Commission to track and evaluate "alternative methods of forest management that can increase carbon sequestration and reduce the loss of carbon sequestration to wildfire."[50] Long rotations and alternatives to clearcutting and chemical spraying should be high on the list of such strategies the OGWC pursues with the Department of Forestry and other agencies since these techniques can help transform the industrial forestlands from a source to a sink for GHG emissions and help rebuild forest structure to be more tolerant of drought and wildfire.[51] For instance, if timber harvest rates were lengthened by 50 years compared to status quo logging, carbon stores would increase by 15%, thereby reducing emissions[52].

3. Manage state forests to maximize their carbon storage value.

State-owned forests make up just 10% of the land base in western Oregon but are the only forests managed under the Oregon Forest Practices Act that can be controlled directly by public decision makers and managed for public benefits. Ostensibly, these lands are managed to achieve the greatest permanent value to all Oregonians. As provided in the governing statutes and regulations, "greatest permanent value" means healthy, productive, and sustainable forest

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[49] ORS 468A.250 (f); (i).
[50] ORS 468A.250 (i).
[51] Von Hagen, Bettina and Michael S. Burnett. 2003. Emerging Markets for Carbon Stored by Northwest Forests. Chapter 8 in OFRI, Forests, Carbon and Climate Change: A Synthesis of Scientific Findings. Corvallis, OR: Oregon Forest Resources Institute (OFRI)
[52] Hudiburg, T., et al. 2009. Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage. *Ecol. Applic*. 19:163-180; Law, B. et al. 2001. Carbon storage and fluxes in ponderosa pine forests at different developmental stages. *Global Change Biology* 7:755-777.

ecosystems that over time and across the landscape provide a full range of social, economic, and environmental benefits to the people of Oregon.[53]

But recent forest planning efforts have, instead, focused on strategies that continue the conversion of these lands into industrial tree plantations managed on short (40-60 year) rotations. As discussed here, such management strategies will increase GHG emissions, reduce sequestration capacity, and undermine climate adaptation goals. In a recent testimony and analysis provided to the Board of Forestry, CSE demonstrated how long rotations can achieve the greatest permanent value standard by maximizing the value of carbon stored, the value of the land for conservation purposes and the value of standing timber because state forests would contain more volume and command higher market value as prime veneer logs rather than logs only suitable for pulp, paper, and other low value products (Figure 6).[54] These simple win-win solutions for climate and state forests should be vigorously explored and implemented.

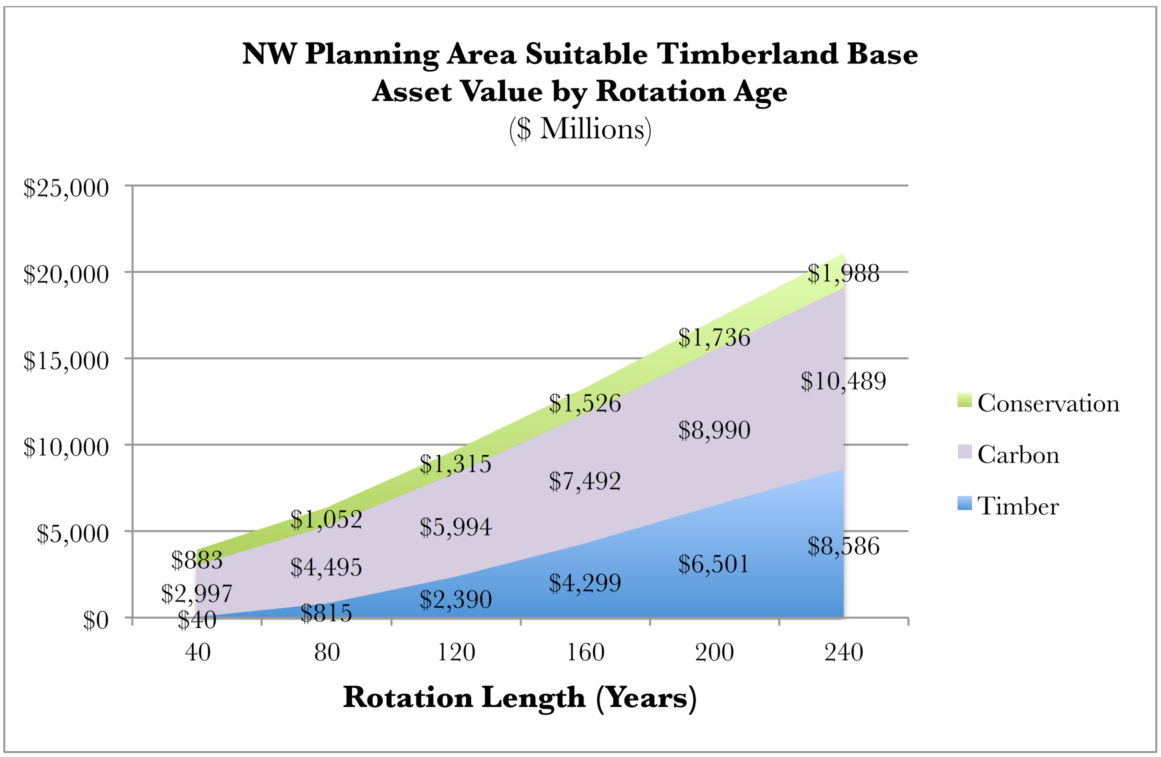

Figure 6: Long Rotations and State Forest Asset Value

4. Close tax loopholes that increase emissions from industrial forestlands.

Oregon's forestland tax structure encourages practices that generate GHG emissions. In 1999, the Legislature, at former Governor Kitzhaber's behest, exempted large, industrial forestland owners (> 5,000 acres) from paying the timber harvest privilege tax (a tax on volume removed)

53 ORS 530.0505; OAR 629-035-0000 et seq.

54 Talberth, John. 2015. Testimony of Dr. John Talberth Before the Board of Forestry, Subcommittee on Alternative Forest Management Plans for Northwest State Forests, October 19th.

while keeping that tax intact for small forestland owners.[55] As demonstrated in this analysis, this is exactly the opposite of a tax structure to incentivize carbon storage since it is the small, non-industrial forestland owners that are managing their lands better from a GHG emissions standpoint. All forestland owners also receive major tax breaks under Oregon's Forestland and Small Tract Forestland programs.[56] Property tax breaks of 90% are provided to any land managed for timber production regardless of the condition of that land. Ending the practice of applying this tax break to open clearcuts and logging roads would generate tremendous revenues for the state, counties, and the school system, help incentivize carbon storage, and provide badly needed funds for climate adaptation.

There are many other options for modernizing Oregon's outdated Forest Practices Act, timber tax codes, and other aspects of the regulatory framework to help fulfill HB3543's mandate to enroll Oregon's forests in the fight against global warming. Over the next year, CSE and Geos Institute will be exploring these reforms in more detail.

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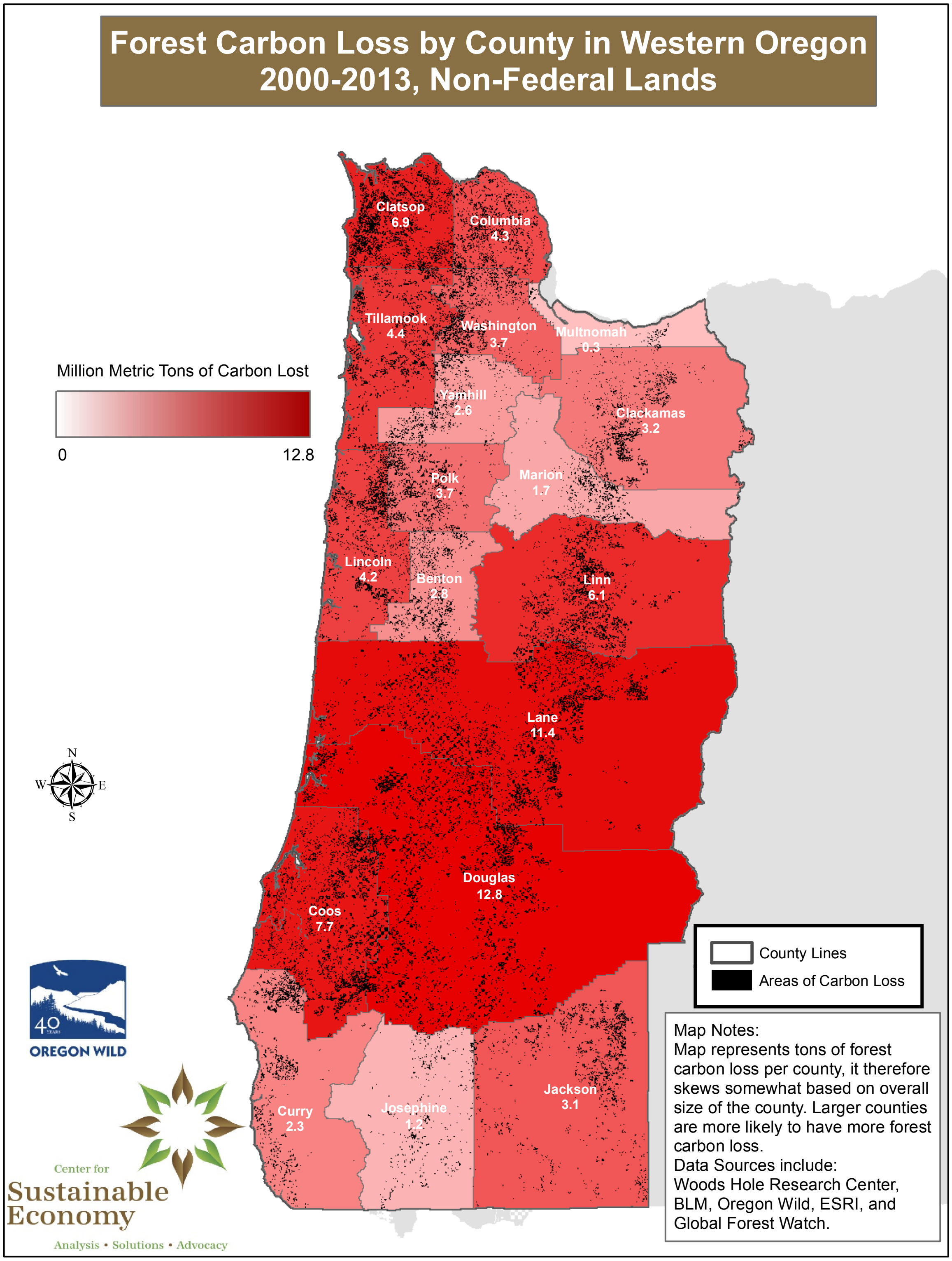
[55] For a brief overview of this tax break, see Keene, Roy. 2015. "Elliott State Forest sale won't solve anything." Guest Opinion. Eugene: Eugene Register Guard.

[56] A description of the programs are accessible online at: http://www.oregon.gov/dor/TIMBER/Pages/forestland.aspx.

Appendix 1



Forest Carbon Loss by County in Western Oregon
2000-2013, Non-Federal Lands
Million Metric Tons of Carbon Lost
0
12.8
Clatsop 6.9
Columbia 4.3
Tillamook 4.4
Washington 3.7
Multnomah 0.3
Yamhill 2.6
Clackamas 3.2
Polk 3.7
Marion 1.7
Lincoln 4.2
Benton 2.8
Linn 6.1
Lane 11.4
Douglas 12.8
Coos 7.7
Curry 2.3
Josephine 1.2
Jackson 3.1
N
W
E
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County Lines
Areas of Carbon Loss
Map Notes:
Map represents tons of forest carbon loss per county, it therefore skews somewhat based on overall size of the county. Larger counties are more likely to have more forest carbon loss.
Data Sources include:
Woods Hole Research Center, BLM, Oregon Wild, ESRI, and Global Forest Watch.
OREGON WILD
Center for Sustainable Economy
Analysis • Solutions • Advocacy

Harris et al. Carbon Balance Manage (2016) 11:24
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Carbon Balance and Management

RESEARCH

Open Access

Attribution of net carbon change by disturbance type across forest lands of the conterminous United States

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Abstract

Background: Locating terrestrial sources and sinks of carbon (C) will be critical to developing strategies that contribute to the climate change mitigation goals of the Paris Agreement. Here we present spatially resolved estimates of net C change across United States (US) forest lands between 2006 and 2010 and attribute them to natural and anthropogenic processes.

Results: Forests in the conterminous US sequestered -460 ± 48 Tg C year^{-1}, while C losses from disturbance averaged 191 ± 10 Tg C year^{-1}. Combining estimates of net C losses and gains results in net carbon change of -269 ± 49 Tg C year^{-1}. New forests gained -8 ± 1 Tg C year^{-1}, while deforestation resulted in losses of 6 ± 1 Tg C year^{-1}. Forest land remaining forest land lost 185 ± 10 Tg C year^{-1} to various disturbances; these losses were compensated by net carbon gains of -452 ± 48 Tg C year^{-1}. C loss in the southern US was highest (105 ± 6 Tg C year^{-1}) with the highest fractional contributions from harvest (92%) and wind (5%). C loss in the western US (44 ± 3 Tg C year^{-1}) was due predominantly to harvest (66%), fire (15%), and insect damage (13%). The northern US had the lowest C loss (41 ± 2 Tg C year^{-1}) with the most significant proportional contributions from harvest (86%), insect damage (9%), and conversion (3%). Taken together, these disturbances reduced the estimated potential C sink of US forests by 42%.

Conclusion: The framework presented here allows for the integration of ground and space observations to more fully inform US forest C policy and monitoring efforts.

Keywords: Forests, Disturbance, Harvest, Insects, Fire, Drought, Greenhouse gas, Land use, Climate change, FIA, UNFCCC

Background

The 2015 Paris Climate Change Agreement, with consensus from 192 signatories, calls for achieving a balance between anthropogenic emissions by sources and removals by sinks in the second half of this century [1]. Forests are currently responsible for the capture and storage of an estimated 25% of global anthropogenic emissions [2]. If Paris goals are to be achieved, further enhancement of forest-based carbon (C) removals to mitigate emissions in other sectors will be a critical component of any collective global strategy [3], especially as no alternative sink technologies have yet been proven at scale. Thus, spatially identifying terrestrial sources and sinks of carbon, and understanding them well enough to predict how they will respond to management decisions or future climate change, will pose major science and policy challenges in the years to come.

Remote sensing products can provide regular and consistent observations of Earth's surface to help identify the condition of forest ecosystems and changes within them at a range of spatial and temporal scales [4]. Over the past

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several years, the remote sensing research community has used these products to monitor tropical deforestation, forest C stocks and associated C emissions, largely in support of REDD+ initiatives in developing countries [5–12]. In many developed countries, periodic national forest inventories form the basis of annual greenhouse gas (GHG) reporting to the United Nations Framework Convention on Climate Change (UNFCCC). The sample-based design of these inventories may offer little in the way of detailed and spatially-explicit information on the distribution of forest biomass [13], timing and location of timber harvesting in managed forests, or the cause and timing of other types of forest disturbances. If the ultimate aim of the Paris Agreement is to introduce practices that lead to reduced emissions and enhanced removals of C from the world's managed forests, including in temperate and boreal biomes, then a lack of disaggregated, spatially-explicit information could pose challenges over the coming years related to knowledge of where changes are occurring and where interventions are likely to be most effective.

Several C budget models have been developed to simulate ecosystem response to climate drivers and other disturbances, and these models represent an established approach to estimating C fluxes at national to regional scales. For example, Canada's National Forest Carbon Monitoring Accounting and Reporting System (NFCMARS) uses the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3), and is used also as a decision support tool for forest managers to quantify forest C dynamics at a landscape scale. Different models emphasize different aspects of ecosystem dynamics, with some accounting for competition between plant functional types, nutrient limitation, and natural disturbances. Time series of anthropogenic land-cover changes are usually prescribed based on spatially explicit data. The models can reflect spatial and temporal variability in C density and response to environmental conditions, but their modeled C stocks may differ markedly from observations [14].

Such models are not used explicitly in the GHG inventory for the US to report forest C fluxes. Instead, the current US inventory system uses the C stock-difference accounting approach [15] enabled by the annual national forest inventory conducted by the United States Department of Agriculture (USDA) Forest Service Forest Inventory and Analysis (FIA) program. The difference in C stocks in five C pools is estimated via sequential re-measurements of permanent ground inventory plots. When forest stocks decline, it is assumed that C emissions have occurred from the land to the atmosphere if not reconciled with a transfer to another land use category. Conversely, when forest C stocks increase it is assumed that C has been sequestered from the atmosphere by terrestrial vegetation. In this way, estimated net C change in the US forest sector is the integrated result of both anthropogenic and natural processes—harvest, land use change, fire, drought, insect infestation, wind damage—all of which influence the magnitude of forest C stocks in each pool. Results are most statistically robust when compiled at large spatial scales (e.g., state or regional), such that quantification of finer-scale spatial patterns is less precise. Though changes are well constrained via sequential re-measurements on inventory plots, the US [16, 17] has only recently begun using methods to disaggregate the effects of various disturbance types on forest stocks and fluxes (although this separation is not a requirement of IPCC Good Practice Guidance, [18]).

The objective of this study was to synthesize information from remote sensing observations of forest carbon stocks and disturbance with information collected by various US agencies into a framework that (1) more explicitly attributes C losses to major disturbance types (land use change, harvesting, forest fires, insect damage, wind damage and drought); and (2) disaggregates net C change into relevant IPCC reporting categories of non-forest land converted to forest land, forest land converted to non-forest land, and forest land remaining forest land. This framework allows for the integration of ground and space observations to more fully inform US forest C policy and monitoring efforts.

Methods

We built a spatially-explicit empirical model that combines information from many data sources to infer disturbance and resulting C dynamics within each hectare of forest land in the 48 conterminous states of the US, totaling an area of more than 2.1 million km^2. For the purposes of regional comparison and analyses, we divided the US into three broad regions (North, South, West) based on similar histories of forestland use ([19], Fig. 1) and into nine smaller subregions based on those used in the US FIA program. Forest types were defined as hardwood or softwood, following the National Land Cover Data (NLCD) classification (deciduous forest class: hardwoods; evergreen forest class: softwoods). The time period of analysis is 1 January 2006 to 31 December 2010.

Data inputs

Forest area map (2005)

Forest extent in the base year 2005 was determined from the NLCD and the global tree cover and tree cover change products of Hansen et al. [8]. Specifically, an area was determined to be forested if categorized as

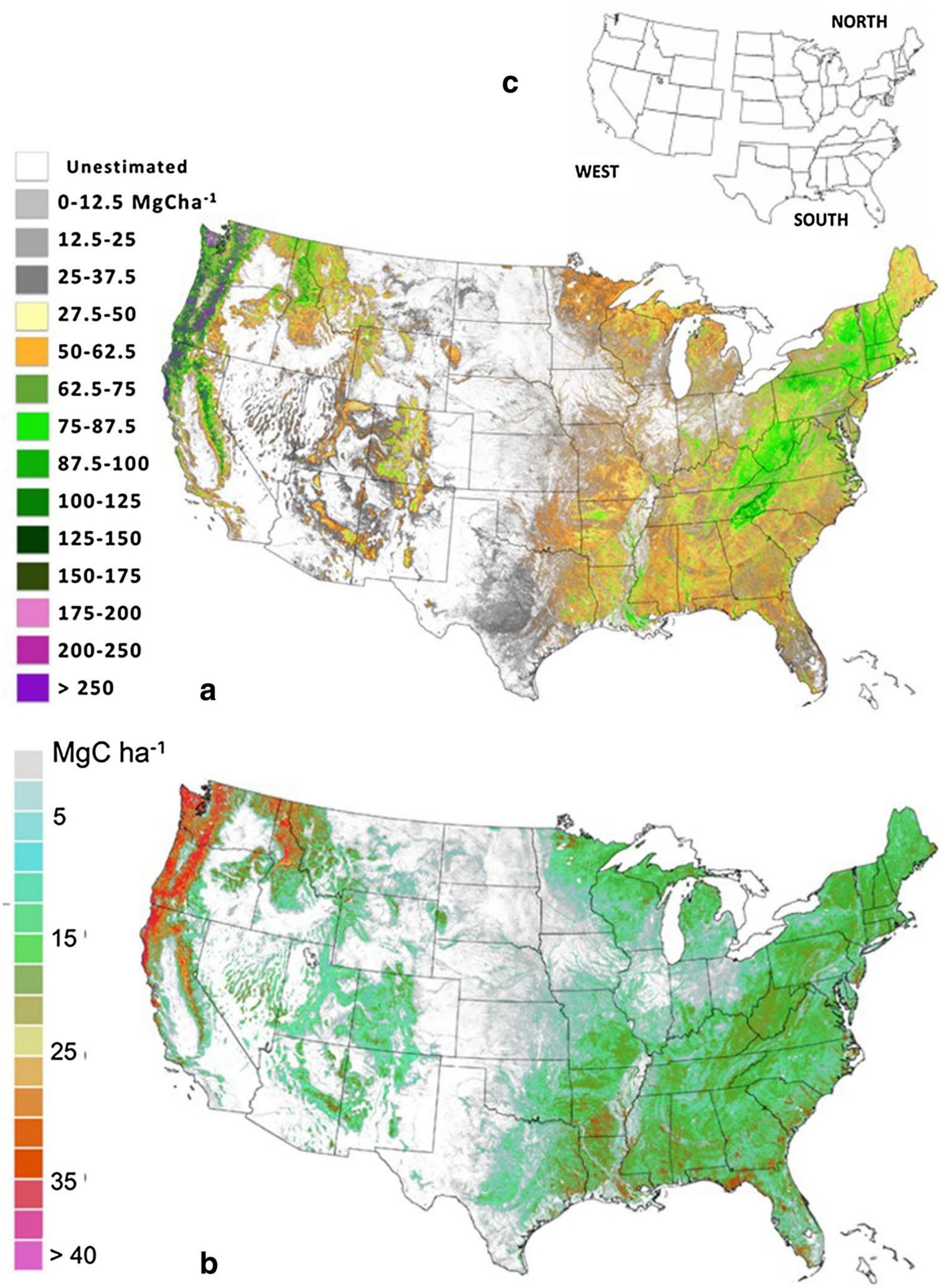

Fig. 1 **a** Map of aboveground live woody biomass carbon density (Mg C ha^{-1}) and **b** uncertainty across forest lands of the conterminous US at 1-ha resolution for circa the year 2005. **c** The regional analysis was performed by dividing the US into three sub-regions as recommended by Heath and Birdsey [19]. The above and belowground carbon density maps and the uncertainty maps can be downloaded from NASA's distributed Data Active Archive Center (http://dx.doi.org/10.3334/ORNLDAAC/1313)

hardwood or softwood in the NLCD 2006 dataset[1] and, according to the Hansen et al. [8] dataset, it (a) met the tree cover threshold of 25% in the year 2000 and was not lost between 2001 and 2005 or (b) did not meet the tree cover threshold of 25% in 2000 but was identified as having gained tree cover (i.e., afforestation/reforestation) between 2000 and 2012. The NLCD has been shown to significantly underestimate tree cover [20] and thus the forest area estimates used in this analysis—defined by both NLCD and Hansen et al. [8]—are likely to be conservative. However, these two data products currently represent the best available spatially explicit data for forest extent in the conterminous US (CONUS).

Forest biomass density maps (circa 2005)

We developed maps of C stocks (50% of biomass) in aboveground live biomass in US forest land as part of NASA's C Monitoring System (CMS) program based on a combination of remote sensing observations and FIA data (Fig. 1). The overall methodology used in mapping the aboveground live forest biomass C density is described in Saatchi et al. [5]. After filtering for cloud effects, slopes, and signal-to-noise ratio, more than 700,000 samples of lidar (light detecting and ranging) data acquired between 2003 and 2008 from the Geoscience Laser Altimeter System (GLAS), onboard the Ice, Cloud and land Elevation Satellite (ICESat) were used as samples of the vertical structure of US forest land. We used the Lorey's height [21] measured in 65,000 single-condition FIA plots (i.e., plots with a single domain mapped on each plot) to calibrate the lidar-derived height metric and used the relationship between Lorey's height and aboveground C density for 28 forest types to convert the lidar data into estimates of aboveground live C density. All FIA plots with a probability of disturbance causing reduced canopy cover (<50%) were removed from the height-biomass model development to reduce any potential discrepancy between ground and lidar height metrics. Lidar-derived biomass samples were then extrapolated over the landscape using a combination of optical and radar satellite imagery that captures the variations of forest structure and cover to create wall-to-wall maps of forest aboveground live biomass C density. We used nine remote sensing imagery layers as spatial predictor variables. Optical and thermal data from Landsat imagery (bands 3, 4, 5 and 7) were aggregated to 100 m spatial resolution from 30 m native resolution along with the leaf area index derived from Landsat imagery [22]. In addition, we used the advanced land observing satellite (ALOS) phased area L-band synthetic aperture radar (PALSAR) imagery at two polarizations (HH and HV backscatter) along with topographical data of surface elevation and slope from Shuttle Radar Topography Mission (SRTM) resampled to 100 m resolution from 20 and 30 m native resolutions, respectively. ALOS PALSAR plays an important role in quantifying variation in forest biomass. In particular, the HV polarization provides the largest contribution among the data layers to predicted biomass because it has a strong direct sensitivity to biomass up to 100–150 $Mgha^{-1}$ (depending on forest type), is less impacted by soil moisture and other environmental variables, and may contribute significantly in extrapolating larger biomass forests through texture and spatial correlation. Similarly, SRTM data include information on topography and also forest height. We used the national elevation data (NED) to represent the ground surface elevation and used the difference between SRTM and NED as an indicator of forest height. This variable also contributed significantly to explaining the spatial variation of biomass over forests with biomass values >150 $Mgha^{-1}$.

The aboveground C density samples derived from GLAS data were combined with satellite imagery using the maximum entropy estimation (MaxEnt) algorithm to estimate aboveground biomass density for each 1-ha pixel. MaxEnt is a probability-based algorithm that estimates the posterior likelihood distribution of a variable by maximizing the entropy of said probability distribution while maintaining the constraints provided by the training samples [23]. We selected a random subset consisting of 70% of the samples (~500,000 samples) for model input and used the remaining 30% for model evaluation and validation. The product from the MaxEnt estimator includes both the mean aboveground carbon (AGC) density for each 1-ha pixel and the estimation of the error derived from a Bayesian probability estimator for each pixel. Spatial uncertainty analysis and uncertainty propagation were used to evaluate the overall uncertainty of AGC at the pixel level. This process included the quantification of error at each step of the process and the use of the Gaussian error propagation approach:

$$\mathrm{Error} = \sqrt{\varepsilon^2_{\mathbf{measurement}} + \varepsilon^2_{\mathbf{allometry}} + \varepsilon^2_{\mathbf{sampling}} + \varepsilon^2_{\mathbf{prediction}}}$$

where each of the terms are the relative errors at that pixel and represent the measurement errors of lidar for capturing the forest height, the error associated with the lidar aboveground C allometry model for each forest type, the error associated with sampling the 1-ha pixel

[1] Within each 1 ha pixel, the wet woodland class was included as forest but was not used to determine whether the pixel was hard- or softwood. Hard- or softwood was determined based on the plurality of NLCD hard- or softwood 30 m pixels within the hectare, ignoring the sub-fraction of wet woodlands and selecting softwood when hard- and softwood fractions were equal.

with GLAS footprint size (~0.25 ha), and the MaxEnt prediction error. In evaluating the errors at the state and county level, we also included the spatial correlation of the prediction error from the MaxEnt approach [24].

In the FIA, belowground forest biomass is quantified using a root-shoot ratio [25]. Knowledge of root biomass dynamics is fundamental to improving our understanding of carbon allocation and storage in terrestrial ecosystems [26]. We used the relationship between belowground carbon (BGC) and AGC from the FIA data to develop a BGC spatial distribution at the same scale as AGC [5, 27]. In estimating the uncertainty in BGC, we followed the same approach as AGC with the addition of including the errors associated with the model used in relating AGC to BGC.

FIA stock change data (2006–2010)

To estimate average net changes in the stock of live AGC and BGC between 2006 and 2010 in forests disaggregated by disturbance type, we queried the FIA database (http://apps.fs.fed.us/fiadb-downloads/datamart.html) to extract more than 141,000 records associated with remeasured permanent plots, where each extracted record represents a "condition" (i.e., domain(s) mapped on each plot according to attributes such as land use, forest type, stand size, ownership, tree density, stand origin, and/or disturbance history) of a measured plot at two points in time, typically 5 years apart. Disturbed plots were stratified into a lookup table by geographic region (North, South, or West), forest type (hardwood or softwood), disturbance type (fire, insect, wind, conversion, or harvest), and disturbance intensity (Table 1). A similar lookup table was developed for undisturbed plots stratified by geographic region, forest type, and base C stock in the year 2005 (Table 2).

Disturbance maps (2006–2010)

Sources of disturbance data used in this analysis are summarized in Table 3 and include spatially-explicit data on locations of fire, insect damage, wind damage, land use change, drought, and timberlands. The timberlands map was used to attribute net carbon gains occurring within vs. outside timberland areas. Because harvested wood may come from intermediate treatments (treatments not intended to cause regeneration), partial harvest or clearcutting forests, deforestation, and non-forest land trees, the area of clearcuts as observed within timberland areas through remote sensing imagery cannot represent all these wood sources [28]. Therefore for estimating C losses from timber harvest, we used data collected in the US based on mill surveys rather than remote sensing observations.

Timber product output data (TPO 2007)

The volume of roundwood products, mill residues and logging residues reported in the TPO database (Table 3), separated by product class and detailed species group, were used to estimate C losses from wood harvest. The spatial resolution of the data was the "combined county", which represented the minimum reportable scale from the timber product output (TPO; FIA Fiscal Year 2013 Business Report, [29]) data while retaining necessary confidentiality.

Model assumptions

IPCC Tier 2 estimation

The terrestrial C cycle includes changes in C stocks due to both continuous processes (i.e., growth, decomposition) and discrete events (i.e., disturbances such as harvest, fire, insect outbreaks, land-use change). Continuous processes can affect C stocks in all areas every year, while discrete events (i.e., disturbances) cause emissions and redistribute C in specific areas in the year of the event. In accounting for net C change in this analysis, we use country-specific data (Tier 2) and apply the simplifying methodological assumption [15] that all post-disturbance emissions (after accounting for C storage in harvested wood products) occur as part of the disturbance event, i.e., in the year of disturbance, rather than modeling these emissions through time as in IPCC's Tier 3 approach. The application of lower tier methods also assumes that the average transfer rate into dead organic matter (dead wood and litter) is equal to the average transfer out of dead organic matter, so that the net stock change in these pools is zero [15]. This assumption means that dead organic matter (dead wood and litter) C stocks need not be quantified for land areas that remain forested. The rationale for this approach is that dead organic matter stocks, particularly dead wood, are highly variable and site-specific, depending on forest type and age, disturbance history and management. Because the FIA data used in this analysis do not include measurements of soil C or dead C pools and no robust relationships currently exist that relate these pools to a more easily measured pool (such as the derivation of belowground biomass from aboveground biomass using root:shoot ratios), we excluded the soil C and dead C pools from our analysis. As a result, our estimate of net C change using the stock-difference approach is equal to the net change in C stocks in the aboveground and belowground live biomass pools only, with a fraction of the aboveground live biomass assumed to be transferred to the wood products pool, where a portion is permanently sequestered in long-lived products and the remainder emitted to the atmosphere (see below).

Table 1 Look-up table of annual fractional change (average = μ; standard error = σ) in aboveground carbon (AGC) and belowground carbon (BGC) in disturbed forests based on FIA plot data

Region	Forest type	Disturbance	Initial C	N	AGC μ	AGC σ	BGC μ	BGC σ
North	Softwood	Fire	Low	2	−0.003	0.012	−0.001	0.013
North	Softwood	Fire	Medium	3	−0.052	0.031	−0.053	0.031
North	Softwood	Fire	High	5	−0.150	0.030	−0.157	0.030
North	Softwood	Weather	Low	63	−0.013	0.016	−0.014	0.016
North	Softwood	Weather	High	10	−0.163	0.013	−0.169	0.013
North	Softwood	Insect	Low	85	−0.003	0.007	−0.003	0.008
North	Softwood	Insect	Medium	82	−0.044	0.023	−0.046	0.023
North	Softwood	Insect	High	45	−0.126	0.035	−0.133	0.032
North	Softwood	Harvested	Low	521	−0.046	0.035	−0.048	0.036
North	Softwood	Harvested	High	246	−0.152	0.026	−0.158	0.025
North	Hardwood	Fire	Low	40	−0.003	0.009	−0.003	0.009
North	Hardwood	Fire	Medium	29	−0.045	0.024	−0.048	0.023
North	Hardwood	Fire	High	11	−0.131	0.034	−0.136	0.034
North	Hardwood	Weather	Low	412	−0.011	0.016	−0.011	0.016
North	Hardwood	Weather	High	34	−0.160	0.017	−0.164	0.016
North	Hardwood	Insect	Low	656	−0.002	0.008	−0.002	0.008
North	Hardwood	Insect	Medium	432	−0.045	0.020	−0.046	0.020
North	Hardwood	Insect	High	118	−0.132	0.029	−0.136	0.028
North	Hardwood	Harvested	Low	2177	−0.047	0.035	−0.047	0.035
North	Hardwood	Harvested	High	806	−0.154	0.023	−0.157	0.023
South	Softwood	Fire	Low	127	−0.002	0.007	−0.003	0.008
South	Softwood	Fire	Medium	174	−0.048	0.021	−0.052	0.022
South	Softwood	Fire	High	52	−0.124	0.027	−0.131	0.028
South	Softwood	Weather	Low	78	−0.016	0.016	−0.017	0.016
South	Softwood	Weather	High	16	−0.161	0.026	−0.168	0.023
South	Softwood	Insect	Low	46	−0.002	0.008	−0.004	0.008
South	Softwood	Insect	Medium	66	−0.054	0.022	−0.059	0.023
South	Softwood	Insect	High	60	−0.135	0.030	−0.142	0.029
South	Softwood	Harvested	Low	1787	−0.044	0.034	−0.048	0.036
South	Softwood	Harvested	High	586	−0.149	0.025	−0.157	0.024
South	Hardwood	Fire	low	112	−0.002	0.008	−0.003	0.008
South	Hardwood	Fire	Medium	86	−0.042	0.021	−0.045	0.022
South	Hardwood	Fire	High	37	−0.131	0.033	−0.139	0.030
South	Hardwood	Weather	Low	484	−0.014	0.016	−0.015	0.016
South	Hardwood	Weather	High	32	−0.162	0.019	−0.167	0.017
South	Hardwood	Insect	Low	145	0.000	0.013	−0.002	0.011
South	Hardwood	Insect	Medium	121	−0.047	0.022	−0.051	0.022
South	Hardwood	Insect	High	38	−0.133	0.031	−0.138	0.031
South	Hardwood	Harvested	Low	1235	−0.048	0.036	−0.051	0.036
South	Hardwood	Harvested	High	609	−0.146	0.029	−0.152	0.027
West	Softwood	Fire	Low	13	−0.007	0.008	−0.007	0.008
West	Softwood	Fire	Medium	8	−0.049	0.023	−0.050	0.026
West	Softwood	Fire	High	0	*−0.126*	*NA*	*−0.133*	*NA*
West	Softwood	Weather	Low	5	−0.003	0.008	−0.003	0.008
West	Softwood	Weather	High	0	*−0.162*	*NA*	*−0.168*	*NA*
West	Softwood	Insect	Low	12	0.001	0.007	0.001	0.007
West	Softwood	Insect	Medium	3	−0.041	0.016	−0.044	0.018
West	Softwood	Insect	High	0	*−0.131*	*NA*	*−0.138*	*NA*

Table 1 continued

Region	Forest type	Disturbance	Initial C	N	AGC μ	AGC σ	BGC μ	BGC σ
West	Softwood	Harvested	Low	28	−0.027	0.030	−0.028	0.031
West	Softwood	Harvested	High	0	*−0.150*	*NA*	*−0.157*	*NA*
West	Hardwood	Fire	Low	4	−0.002	0.008	−0.002	0.008
West	Hardwood	Fire	Medium	3	−0.057	0.021	−0.059	0.021
West	Hardwood	Fire	High	0	*−0.131*	*NA*	*−0.138*	*NA*
West	Hardwood	Weather	Low	0	*−0.013*	*NA*	*−0.013*	*NA*
West	Hardwood	Weather	High	0	*−0.161*	*NA*	*−0.165*	*NA*
West	Hardwood	Insect	Low	13	−0.003	0.008	−0.003	0.009
West	Hardwood	Insect	Medium	3	−0.041	0.025	−0.044	0.028
West	Hardwood	Insect	High	0	*−0.132*	*NA*	*−0.136*	*NA*
West	Hardwood	Harvested	Low	4	−0.039	0.031	−0.039	0.033
West	Hardwood	Harvested	High	0	*−0.151*	*NA*	*−0.155*	*NA*

Italics imputed from other regions

Disturbance attribution

Forest land was assumed to be disturbed if included in at least one of the disturbance maps (Table 3) during the 2006–2010 time period: (1) maximum burn severity score of at least two (low) over the 5 years of fire data; (2) insect damage of at least three trees per acre over the 5 year study period; (3) within a path of a tornado or a buffered region around the hurricane path where wind speeds typically exceeded 95 miles per hour (category 2 hurricane)[2] between 2006 and 2010; (4) converted to agriculture, barren land or settlement in the NLCD layer between 2006 and 2011 (considered as deforestation events); or (5) had an average drought intensity score of more than two in the NDMC Drought Monitor map between the years of measurement. For fire and insect disturbance, three levels of disturbance intensity were assigned based on burn severity score (from the MTBS dataset) or insect damage per acre (from the Aerial Detection Survey), respectively. Two levels of wind disturbance intensity were assigned and areas determined to have been converted to agriculture or settlement were assumed to experience one uniform intensity of disturbance. All other forest land was assumed to be undisturbed between 2006 and 2010. In areas where multiple types of disturbance were identified within a 1 ha forest land pixel, we assumed only one disturbance type was driving the C loss. Disturbance type priority was set based on the intensity of the disturbance and level of confidence in the data sets. In general, more intense disturbances and higher quality products took priority over less intense disturbances and those products assessed as having more uncertainty. The disturbance location and intensity products were assumed to be in the following quality order, from least to most inherent uncertainty: conversion, fire, wind, insect damage. For instance, a pixel identified as experiencing an intense fire disturbance and a low intensity insect disturbance was assigned the high intensity fire disturbance as the single disturbance driving loss. This assumption simplified the processing but added additional uncertainty to the estimates. The assigned disturbance type priority varied across multiple iterations of our uncertainty analysis. It was not possible to attribute harvest disturbance to specific pixels, therefore C losses from harvest were estimated at the county scale using TPO data.

Estimation of net carbon change

Net carbon change from fire, wind, insect damage, land use change, and drought

If a hectare of forest land in the US was categorized as disturbed between 2006 and 2010 based on the disturbance maps, then the intensity and type of disturbance was identified. The pixel was then linked to an annualized percent net change in C stock estimate, based on its identified category in the FIA-based lookup tables. These annualized percent change values were multiplied by the initial base C stock in 2005 in each pool (aboveground biomass, belowground biomass) and multiplied by 5 years to estimate total net change in C within the pixel between 2006 and 2010.

Net carbon change from harvest

Annual C losses associated with harvest activities were estimated using mill surveys compiled into the USDA

[2] This wind speed threshold was selected based on the Saffir Simpson Hurricane Wind Scale, which indicates that trees start to be uprooted and fall at category 2 sustained wind speeds between 96 and 110 mph. The hurricane tracks were buffered to a symmetrical width of 100 km.

Table 2 Look-up table of annual fractional change (average = μ; standard error = σ) in aboveground carbon (AGC) and belowground carbon (BGC) in undisturbed forests, based on FIA plot data

Region	Forest type	Drought	Initial C	n	AGC μ	AGC σ	BGC μ	BGC σ
North	Softwood	No	<25	5167	0.064	0.135	0.080	0.199
North	Softwood	No	25–50	3459	0.023	0.034	0.023	0.034
North	Softwood	No	50–100	2085	0.016	0.024	0.016	0.024
North	Softwood	No	≥100	345	0.013	0.034	0.013	0.034
North	Softwood	Yes	<25	50	0.028	0.030	0.031	0.035
North	Softwood	Yes	25–50	50	0.008	0.034	0.008	0.035
North	Softwood	Yes	50–100	12	0.016	0.040	0.016	0.040
North	Softwood	Yes	≥100	2	0.013	0.017	0.013	0.016
North	Hardwood	No	<25	12,559	0.074	0.102	0.087	0.131
North	Hardwood	No	25–50	13,656	0.025	0.036	0.025	0.036
North	Hardwood	No	50–100	14,173	0.014	0.026	0.014	0.026
North	Hardwood	No	≥100	3265	0.010	0.030	0.010	0.030
North	Hardwood	Yes	<25	19	0.016	0.058	0.016	0.062
North	Hardwood	Yes	25–50	12	0.006	0.040	0.006	0.041
North	Hardwood	Yes	50–100	7	0.001	0.026	0.000	0.027
North	Hardwood	Yes	≥100	1	0.006	NA	0.005	NA
South	Softwood	No	<25	3648	0.314	0.355	0.452	0.621
South	Softwood	No	25–50	2940	0.082	0.069	0.085	0.072
South	Softwood	No	50–100	2345	0.039	0.049	0.039	0.050
South	Softwood	No	≥100	673	0.021	0.050	0.020	0.051
South	Softwood	Yes	<25	464	0.340	0.407	0.487	0.694
South	Softwood	Yes	25–50	348	0.081	0.071	0.084	0.074
South	Softwood	Yes	50–100	299	0.038	0.039	0.038	0.041
South	Softwood	Yes	≥100	110	0.020	0.038	0.020	0.039
South	Hardwood	No	<25	6585	0.133	0.191	0.176	0.291
South	Hardwood	No	25–50	6180	0.040	0.044	0.041	0.045
South	Hardwood	No	50–100	8244	0.021	0.032	0.021	0.032
South	Hardwood	No	≥100	2697	0.014	0.032	0.014	0.032
South	Hardwood	Yes	<25	630	0.140	0.184	0.185	0.272
South	Hardwood	Yes	25–50	498	0.042	0.062	0.044	0.064
South	Hardwood	Yes	50–100	756	0.021	0.029	0.021	0.030
South	Hardwood	Yes	≥100	275	0.011	0.029	0.011	0.029
West	Softwood	No	<25	56	0.061	0.102	0.079	0.123
West	Softwood	No	25–50	45	0.027	0.048	0.028	0.049
West	Softwood	No	50–100	61	0.022	0.026	0.022	0.027
West	Softwood	No	≥100	80	0.014	0.019	0.014	0.019
West	Softwood	Yes	<25	0	*0.310*	*NA*	*0.443*	*NA*
West	Softwood	Yes	25–50	0	*0.072*	*NA*	*0.075*	*NA*
West	Softwood	Yes	50–100	0	*0.037*	*NA*	*0.037*	*NA*
West	Softwood	Yes	≥100	0	*0.020*	*NA*	*0.020*	*NA*
West	Hardwood	No	<25	33	0.037	0.055	0.043	0.061
West	Hardwood	No	25–50	26	0.023	0.026	0.025	0.028
West	Hardwood	No	50–100	45	0.026	0.041	0.027	0.043
West	Hardwood	No	≥100	38	0.019	0.025	0.020	0.027
West	Hardwood	Yes	<25	0	*0.137*	*NA*	*0.180*	*NA*
West	Hardwood	Yes	25–50	0	*0.041*	*NA*	*0.043*	*NA*

Table 2 continued

Region	Forest type	Drought	Initial C	n	AGC μ	AGC σ	BGC μ	BGC σ
West	Hardwood	Yes	50–100	0	*0.021*	*NA*	*0.021*	*NA*
West	Hardwood	Yes	≥100	0	*0.011*	*NA*	*0.011*	*NA*

Italics imputed from other regions

Table 3 Fourteen independent datasets were integrated and used to produce net carbon change estimates by disturbance type

Product	Source	Spatial coverage	Temporal coverage	Url
Tree cover Tree cover change	[8]	Complete CONUS	Tree cover: single snapshot in 2000 Loss: annual 2001–2010 Gain: 2000–2012	http://earthenginepartners.appspot.com/science-2013-global-forest/download_v1.1.html
Fire	Monitoring trends in burn severity	Complete CONUS	Annual 2006–2010	http://www.mtbs.gov/products.html
Wind	NOAA's storm prediction center—tornado tracks	Complete CONUS	Annual 2006–2010	http://www.spc.noaa.gov/gis/svrgis/
Wind	NOAA's storm prediction center—hurricane paths	Complete CONUS	Annual 2006–2010	http://nhc.noaa.gov/gis/
Insect	USFS aerial detection survey	Sub-set of CONUS	Annual 2006–2010	http://www.fs.fed.us/foresthealth/technology/adsm.shtml
Forest type	National land cover database—hardwood or softwood	Complete CONUS	Single snapshot in 2000	http://www.mrlc.gov/
Conversion	National land cover database	Complete CONUS	Snapshots in 2006 and 2011	http://www.mrlc.gov/
Drought	NDMC drought monitor	Complete CONUS	Weekly between 2006 and 2011	http://droughtmonitor.unl.edu/
Timberlands	Mark Nelson USFS for 2007 resources planning act	Complete CONUS	Snapshot in 2007	N/A
Biomass density Carbon stocks	Sassan Saatchi	Complete CONUS	Snapshot in 2005	http://dx.doi.org/10.3334/ORNLDAAC/1313)
Harvest	USFS timber products output	Combined county CONUS	Survey in 2007	http://www.fia.fs.fed.us/program-features/tpo/
FIA	USFS forest inventory and analysis program	Sites in CONUS	Between 1997 and 2013	http://www.fia.fs.fed.us/

TPO database for the year 2007. Due to the periodic nature of the TPO report for 2007 data, harvest emission estimates were assumed to be representative for all 5 years included in our analysis (2006–2010). Volumes of roundwood products, mill residue and logging residues were converted to biomass using oven-dry wood densities [30]. The fraction of C in primary wood products remaining in end uses or in landfills after 100 years per product class[3] was assumed to be permanently sequestered, and was estimated from values published in Smith et al. [31]. Fuelwood, posts/poles/pilings and miscellaneous product classes were assumed to be fully emitted. Emissions from mill residues were considered equal to the summed mill residues from fuel by-products, miscellaneous by-products and unused mill residues, plus emissions from fiber by-products. All fiber by-products were assumed to form pulp and to follow the emissions assumptions of pulp products. All logging residues were assumed to be emitted. Timberlands were delineated based on the boundaries of the US timberlands map (Table 3), and annual net C gains within timberlands were estimated following the look-up tables for growth in undisturbed forests as described below.

Net carbon change from forest growth/regrowth

Forest land in the US that did not experience deforestation through land use conversion or significant damage by wind, insect, fire, or drought over the analysis period, as well as new forest land (i.e., afforestation/reforestation), were linked to values of annual net change

[3] The TPO and Smith et al. [31] product classes were mapped to one another as follows: Sawlog = softwood/hardwood lumber (depending on species); veneer = softwood plywood; pulp = paper; composite = oriented strandboard.

in C stock, based on the area's identified category in the lookup tables derived from FIA measurement data. These annualized percent change values were multiplied by the initial C stock in 2005 in each pool (aboveground biomass, belowground biomass) and multiplied by 5 years to estimate total net change in C within each 1-ha pixel between 2006 and 2010.

Total annual net carbon change

The FIA-based estimated net change in C represents the sum of net C losses (caused by disturbances) and net C gains (caused by forest growth) that occurred between FIA measurement dates at the site. Similarly, our estimate of net C change (ΔC_{net}) during the 5-year period at the combined county scale was calculated as:

$$\Delta C_{net} = \Delta C_{undist} + \Delta C_{A/R} + \Delta C_{conversion} + \Delta C_{timberlands} + \Delta C_{insect} + \Delta C_{fire} + \Delta C_{wind} + \Delta C_{drought}$$

where ΔC_{undist} is the net C change in forest land outside of timberlands that did not experience land use conversion or significant damage by wind, insects, fire or drought. $\Delta C_{A/R}$ is the net C change in new forest land. $\Delta C_{conversion}$, ΔC_{wind} ΔC_{insect}, and ΔC_{fire} represent the net C change in forestland that was converted or significantly disturbed by conversion, wind, insects, and fire, respectively. $\Delta C_{drought}$ is the net C reduction in sequestration in forest land experiencing drought from what was expected during non-drought periods. $\Delta C_{timberlands}$ is the net C change on timberlands (as delineated by the timberlands map), calculated as the sum of net C gains (as estimated from FIA lookup tables) and C losses (as estimated from the TPO data, accounting for the fraction of harvested C stored permanently in the long-lived product pool). By convention, C losses are represented as positive values and C gains as negative values. Consequently, various forms of disturbance result in a weaker (i.e., less negative) overall sink than would occur otherwise in the absence of disturbance.

Uncertainty analysis

We estimated statistical bounds for the estimates of net C change by conducting a Monte Carlo uncertainty analysis [32]. The four sources of uncertainty included in the simulation were associated with the forest biomass density maps, the stock-change lookup tables derived from FIA data, each of the disturbance maps, and the TPO data. The simulation was conducted at the combined county scale. Uncertainty in the biomass density maps was derived from a secondary simulation in which the input datasets were resampled to generate 100 replicate training datasets, or realizations, that had the same qualities of the original training dataset, but different random error. A new MaxEnt model was fit to each of these 100 replicated datasets and used to create 100 full resolution biomass maps. Uncertainty in the FIA-based ΔC values were calculated using the variance in the look-up tables:

$$uncertainty\% = \frac{\frac{\sigma}{\sqrt{n}} * 1.96}{\mu} * 100$$

Uncertainty in the area affected by disturbance was estimated to be 30%, with an estimated 5% bias in under reported area. We conducted the simulation using three separate rule sets for selecting a disturbance type for pixels identified as experiencing multiple disturbances during the 5-year study period. Uncertainty in the TPO data at the combined county scale was also assumed to be 30%.

We ran 10,000 Monte Carlo simulations with stochastic elements in place for the four uncertainty components. We assumed that 80% of the randomly generated error was random and 20% of the error was systematic within the simulation. To implement this assumption, we estimated the error associated with each component twice—once at the simulation iteration level and again for each individual combined county. The iteration level uncertainty was multiplied by 0.2 before it was added to the original combined county estimate, while the combined county level stochastic element was multiplied by 0.8 before it was added. In this way, we accounted for both random error as well as systematic error in our estimates.

This uncertainty analysis was intended to provide context to the estimates and assist in the process of identifying methods and data in need of refinement or replacement. The uncertainty analysis is not exhaustive, in the sense that additional sources of uncertainty exist that are not accounted for in the analysis presented here. These additional sources include but are not limited to (a) potential temporal mismatch between the biomass data providing initial carbon stocks in 2005 and the activity data beginning in 2006 and (b) uncertainty in the equations and factors used in the FIA to convert tree measurements to estimates of wood volume and carbon stocks. Given these additional sources of uncertainty, the uncertainty bounds presented here are almost certainly an underestimate of the actual uncertainty.

Results

Forest land in the conterminous US, as defined here totaling 221 million ha in 2005, sequestered -460 ± 48 Tg C year^{-1} between 2006 and 2010, while average C losses from forest disturbances were 191 ± 10 Tg C year^{-1}. Combining estimates of net C gains and net C losses results in net C change of -269 ± 49 Tg C year^{-1} (Fig. 2). These results are broadly

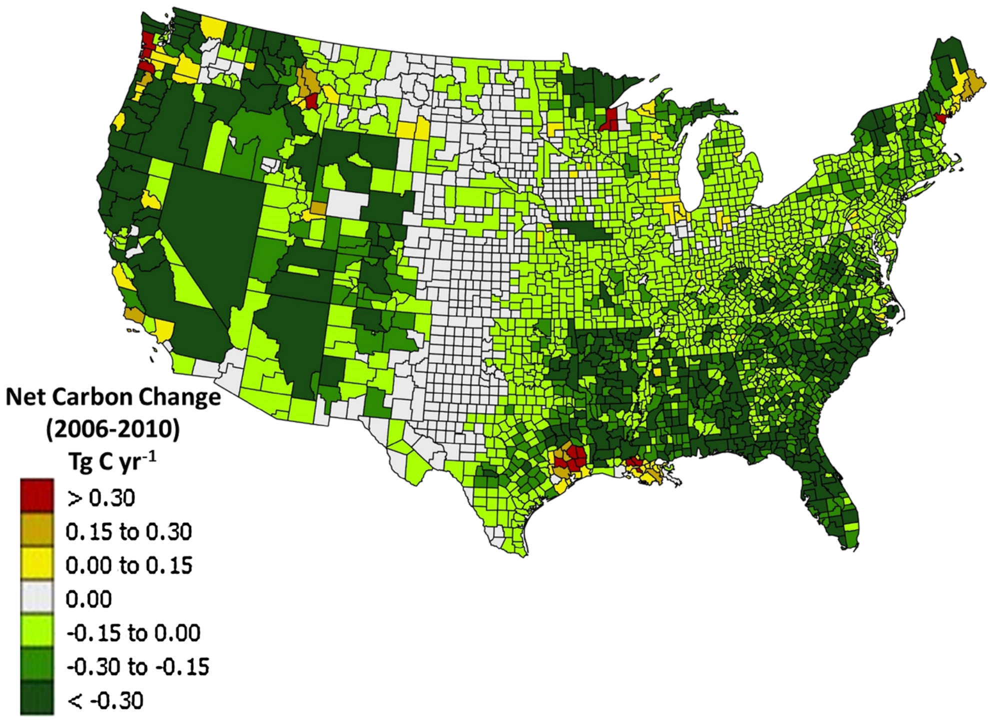

Fig. 2 Average annual net carbon change (Tg C $year^{-1}$) at the combined county scale across the CONUS. Most combined counties (91%) are net C sinks while areas with extensive forest disturbance can be net C sources to the atmosphere

consistent with estimates reported in the US. GHG inventory for forests in 2010 (−293 Tg C $year^{-1}$, [33]) but we estimate a larger net sink than reported in Zheng et al. [28] (−181 Tg C $year^{-1}$), although the spatial and temporal domains varied across these analyses, as did the C pools included.

New forests, averaging 0.4 million ha per year, sequestered −8 ± 1 Tg C $year^{-1}$, while deforestation, averaging 0.1 million ha per year, resulted in C losses of 6 ± 1 Tg C $year^{-1}$. Forest land remaining forest land lost 184 ± 10 Tg C $year^{-1}$ to disturbance (13% from natural disturbance, 87% from harvest); these were compensated by net carbon gains of 452 ± 48 Tg C $year^{-1}$, 75% of which occurred within timberland areas (Table 4). C losses from natural and human induced disturbances reduced the potential net C sink in US forests by 42% compared to the potential sink estimated without disturbance effects included, an estimate that is similar to other studies [28, 34].

Regional variation in net C change across the nation was substantial. The South sequestered more C in growing forests (−271 ± 28 Tg C $year^{-1}$) than the North (−97 ± 10 Tg C $year^{-1}$) or the West (−92 ± 11 Tg C $year^{-1}$), while at the same time losing more C to the atmosphere from disturbances (105 ± 6 Tg C $year^{-1}$) than the other regions (41 ± 2 Tg C $year^{-1}$ for the North and 44 ± 3 Tg C $year^{-1}$ for the West). Forest C change in the South was substantial, in terms of both C losses and gains, because this region is home to a majority of the wood harvest occurring in the US (60% of all C loss from harvest occurred in the South), and is therefore also home to the largest area of regenerating forests that are sequestering C at high rates. At the state level, the highest C losses occurred in the forests of Georgia, Alabama, Washington, Mississippi, Louisiana, and Oregon, with each of these states losing more than 11 Tg C $year^{-1}$ (Table 5). Georgia, Florida, Alabama, Mississippi, and North Carolina gained the most forest C in the time period, with each sequestering at least 24 Tg C $year^{-1}$. C gains exceeded C losses in all states. Forests in approximately 6% of combined counties were a net source of C to the atmosphere (Fig. 2).

We estimated net C losses from six separate disturbance processes: fire, insect infestation, wind, timber harvest, land use conversion, and drought (Fig. 3). C losses from harvest (162 ± 9.9 Tg C $year^{-1}$) were more than five times higher than losses from all other processes combined (30 ± 2.6 Tg C $year^{-1}$). Fire (7 ± 1.0 Tg C $year^{-1}$), wind (5 ± 0.7 Tg C $year^{-1}$), insect infestation (10 ± 1.3 Tg C $year^{-1}$), and deforestation (6 ± 0.7 Tg C $year^{-1}$) each contributed a similar magnitude of C losses across the CONUS, while drought

Table 4 Average annual net C change (Tg C year^{-1}) across US forests between 2006 and 2010, disaggregated into categories of non-forest land to forest land, forest land to non-forest land, and forest land remaining forest land

Category	Area (Mha year^{-1})	Net C gain (Tg C year^{-1})	Net C loss (Tg C year^{-1})
Non-forest land to forest land	0.4	−8 ± 1	
Forest land to non-forest land	0.1		6 ± 1
Forest land remaining forest land	221.1	−452 ± 47	185 ± 10
Insect damage	0.9		9 ± 1
Forest fire	0.6		7 ± 1
Wind damage	0.6		5 ± 1
Drought	0.8		1 ± 0
Timberlands	152.0	−342 ± 42	162 ± 10
Undisturbed forest	54.9	−109 ± 19	
Total	221.6	−460 ± 48	191 ± 10
Net C change			−269 ± 49

Results are further disaggregated by disturbance type within the forest land remaining forest land category

accounted for about 1 ± 0.2 Tg C year^{-1}. Individual disturbances had spatially distinct distributions (Fig. 4a). On average, drought affected areas had C sequestration rates 20% lower than drought-free areas.

C losses in the South were highest (105 ± 6 Tg C year^{-1}) with the highest fractional contributions from harvest (92%) and wind (5%), with a particularly high concentration of loss coming from the South Central region (including the states of Texas, Oklahoma, Mississippi, Louisiana, Kentucky, Tennessee, Alabama, and Arkansas; Fig. 4b). The West had the second highest C loss (44 ± 3 Tg C year^{-1}) with significant contributions from harvest (66%), fire (15%), and insects (13%). The North had the lowest C loss (41 ± 2 Tg C year^{-1}) with most significant proportional contributions coming from harvest (86%), insect damage (9%), and conversion (3%).

Our results can also be used to estimate net C impacts of localized disturbances at finer spatial scales. A tornado struck Lakewood, Wisconsin on 7 June 2007 and caused severe forest damage, resulting in net C loss of more than 0.3 Tg C across a 13,000 ha swath (Fig. 5a). The wild fire in southern California's Santa Barbara County, termed the "Zaca" fire, started on 4 July 2007 and caused extensive damage to more than 97,000 ha of forest in the Los Padres National Forest, resulting in net C loss of more than 4 Tg C (Fig. 4b).

The highest fractional contribution of C loss in all states was from harvest (Table 4), and 64% of these losses were from logging residues [both above- (19%) and below-ground (23%)] and mill residues (22%). Across all wood product classes, the production of pulpwood resulted in the highest forest C losses (26 Tg C year^{-1}), followed by saw logs (18 Tg C year^{-1}), although a high proportion of C in saw logs is in use or in landfills, both which are considered to be long-term C storage (Fig. 6).

Discussion

Comparison with other studies

We estimate that Hurricanes Gustav and Ike in 2008, the only two hurricanes above category 2 to make landfall during the study period, damaged forests in Texas and Louisiana and led to net C change of more than 22 ± 2 Tg C (or 4 ± 0.5 Tg C year^{-1} on average over the 5 year period). Other studies report average annual C loss in US forests due to hurricane damage in the 20th century of 14 Tg C year^{-1} [35]. Zhou et al. [36] estimate total C emissions from wood harvest in 35 eastern US states as 168 Tg C year^{-1} between 2002 and 2010, while our estimate for the same geographic extent is 132 ± 8 Tg C year^{-1} between 2006 and 2010. Other national scale estimates of emissions from wood harvest are lower, such as that of Williams et al. [37] (107 Tg year^{-1} in 2005) and Powell et al. [34] (74 Tg C year^{-1} between 1986 and 2004). Hicke and Zeppel [38] estimated that bark beetles and fire together resulted in gross emissions of 32 Tg C year^{-1} in the western US between 1997 and 2010. We estimate that insects and fire resulted in net C change of 17 ± 2 Tg C year^{-1} between 2006 and 2010. We conclude that, given the different spatial extents, time periods and C pools included, results from our analysis that cover all disturbance types are broadly consistent with these and other more specialized studies (see Williams et al. [39] for a comprehensive review).

Priorities for improved forest carbon change estimates

Results generated from this analysis are dependent on the algorithm that assigns each hectare of forest land to a category that is then associated with a C stock change value. By including spatial data sets of carbon stocks and disturbance from remote sensing observations, the

Table 5 State level estimates of forest area in 2005 (millions of ha), net C gains, net C losses, and net C change (Tg C year^{-1}) together with the percent of C loss attributable to harvest, drought, fire, wind, insect infestation, and land use conversion within the state

State	Forest area	C gain	C loss	Net C change	Fire (%)	Insect (%)	Wind (%)	Conversion (%)	Drought (%)	Harvest (%)
Alabama	8.5	−27.3	12.5	−14.9	0	1	0	1	0	97
Arizona	2.0	−2.4	0.4	−1.9	22	0	1	0	0	77
Arkansas	7.4	−22.6	8.6	−14.0	1	2	0	2	0	95
California	9.3	−16.8	9.4	−7.4	32	0	7	1	0	60
Colorado	5.1	−6.7	0.3	−6.3	8	0	0	1	0	92
Connecticut	0.9	−1.2	0.2	−1.0	0	0	1	31	0	68
Delaware	0.2	−0.2	0.1	−0.1	0	0	0	4	0	95
District of Columbia	<0.1	0.0	0.0	0.0	0	0	0	100	0	0
Florida	6.4	−28.5	6.3	−22.2	3	0	0	3	0	94
Georgia	9.4	−33.2	14.4	−18.8	1	1	0	2	0	96
Idaho	7.1	−10.2	4.9	−5.3	29	0	23	0	0	48
Illinois	2.3	−2.8	1.1	−1.7	0	0	0	3	0	97
Indiana	2.3	−2.8	1.7	−1.1	0	0	3	1	0	95
Iowa	1.2	−1.5	0.4	−1.1	0	1	0	3	0	96
Kansas	0.9	−1.1	0.2	−0.9	0	1	0	3	0	95
Kentucky	5.7	−11.5	3.3	−8.2	1	0	0	6	0	93
Louisiana	5.4	−18.0	11.1	−6.9	0	19	0	1	0	79
Maine	6.8	−7.7	6.7	−0.9	0	0	15	1	0	84
Maryland	1.2	−1.5	0.8	−0.8	0	0	6	7	0	86
Massachusetts	1.5	−1.9	0.6	−1.3	0	0	4	18	0	78
Michigan	8.5	−10.3	4.3	−6.0	0	0	1	1	11	87
Minnesota	7.7	−9.5	3.2	−6.3	1	0	3	1	0	96
Mississippi	7.0	−24.3	11.6	−12.7	0	2	0	2	0	96
Missouri	7.1	−8.7	2.7	−6.0	1	2	0	4	0	93
Montana	7.3	−8.6	5.0	−3.5	14	0	49	0	0	37
Nebraska	0.3	−0.4	0.1	−0.2	2	1	0	0	0	97
Nevada	0.7	−0.8	0.1	−0.7	15	0	0	0	0	84
New Hampshire	2.1	−2.6	0.8	−1.8	0	2	4	6	0	88
New Jersey	1.0	−1.3	0.5	−0.8	2	0	40	14	0	43
New Mexico	2.6	−3.2	0.3	−2.8	33	0	16	0	0	51
New York	8.3	−10.7	3.1	−7.6	0	0	5	4	0	91
North Carolina	7.6	−23.7	9.6	−14.1	0	0	0	1	2	95
North Dakota	0.2	−0.3	0.0	−0.3	0	1	0	2	0	96

Table 5 continued

State	Forest area	C gain	C loss	Net C change	Fire (%)	Insect (%)	Wind (%)	Conversion (%)	Drought (%)	Harvest (%)
Ohio	3.6	−4.4	1.2	−3.2	0	0	7	7	0	86
Oklahoma	3.6	−9.0	1.6	−7.3	2	2	0	3	0	94
Oregon	9.2	−20.6	11.1	−9.6	4	0	2	6	0	88
Pennsylvania	7.6	−9.8	4.0	−5.8	0	0	13	3	0	84
Rhode Island	0.2	−0.2	0.1	−0.2	0	0	3	11	0	85
South Carolina	4.8	−18.4	6.5	−11.9	1	1	0	2	0	97
South Dakota	0.5	−0.6	0.2	−0.3	2	0	0	0	0	98
Tennessee	6.2	−14.2	4.0	−10.1	0	1	0	3	0	95
Texas	7.9	−23.3	9.8	−13.6	1	23	0	2	0	74
Utah	2.2	−2.2	0.3	−1.8	24	0	38	0	0	38
Vermont	2.0	−2.5	0.6	−1.9	0	0	2	1	0	96
Virginia	6.7	−16.5	6.1	−10.4	1	0	0	2	0	97
Washington	7.9	−17.3	11.7	−5.6	3	0	8	19	0	70
West Virginia	5.3	−6.9	2.5	−4.4	0	0	1	6	0	93
Wisconsin	7.2	−8.4	6.3	−2.0	0	1	23	0	5	70
Wyoming	2.7	−3.3	0.8	−2.5	21	0	25	0	0	54
Total	221.5	−459.5	191.1	−268.4	4	3	5	3	1	85

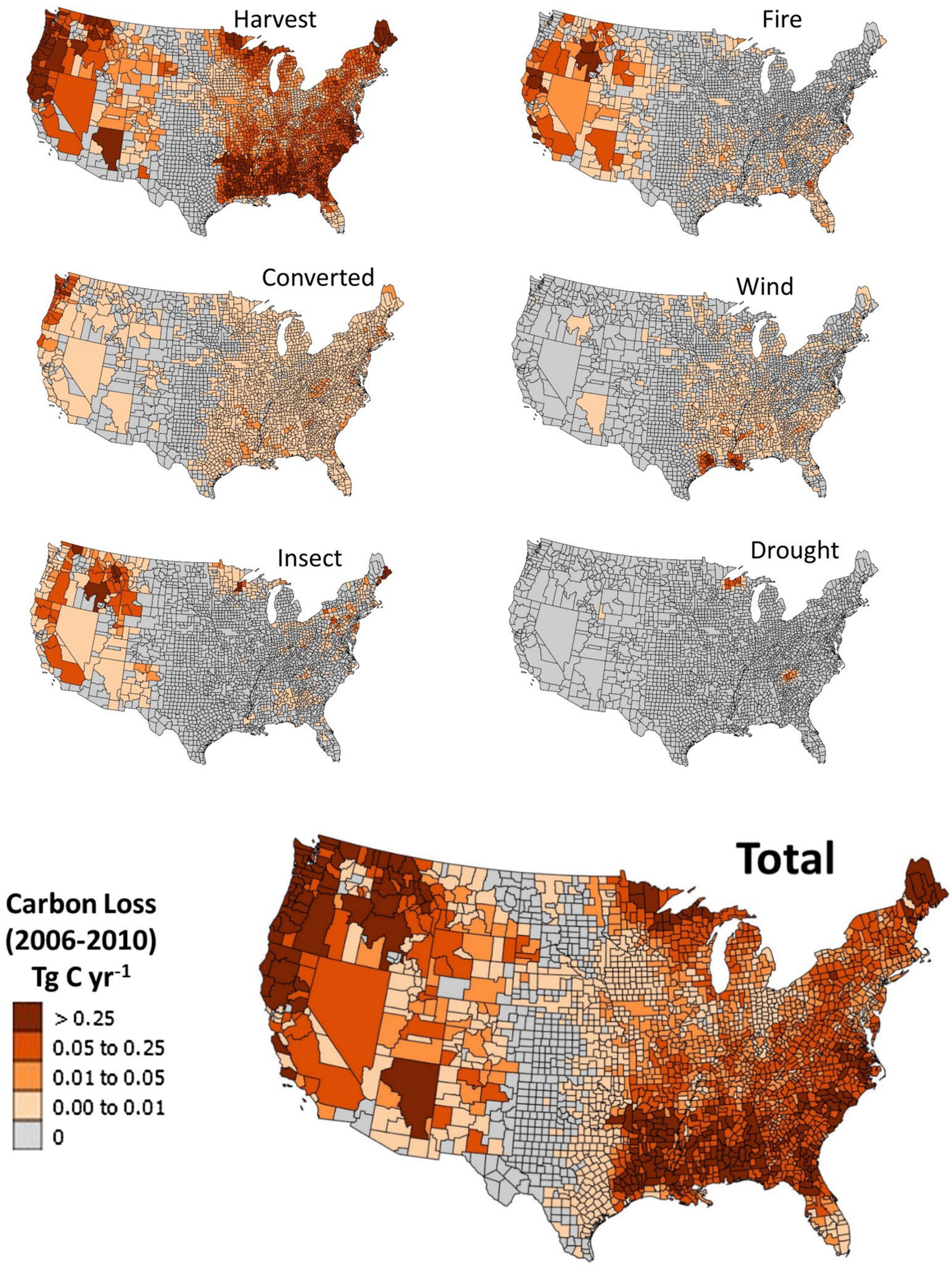

Fig. 3 Average annual net carbon loss (Tg C year^{-1}) attributed to the most likely disturbance type and estimated at the combined county scale for harvest, fire, land use conversion, wind, insect, and drought. Combining these six sources results in estimates of total annual net C loss from disturbance occurring between 2006 and 2010

methodology avoids making gross assumptions on the regional distribution of carbon stocks and disturbance, thus improving estimates of C loss. The strength of this approach is estimated in the uncertainty analysis. Our framework is therefore completely dependent on the underlying data sources and, as the data improve, so will the estimates. Although the US is among the world's leaders in technology and open data, where high quality geospatial datasets are publicly available and inventory programs are maintained by various federal and state agencies, opportunities for improvement remain.

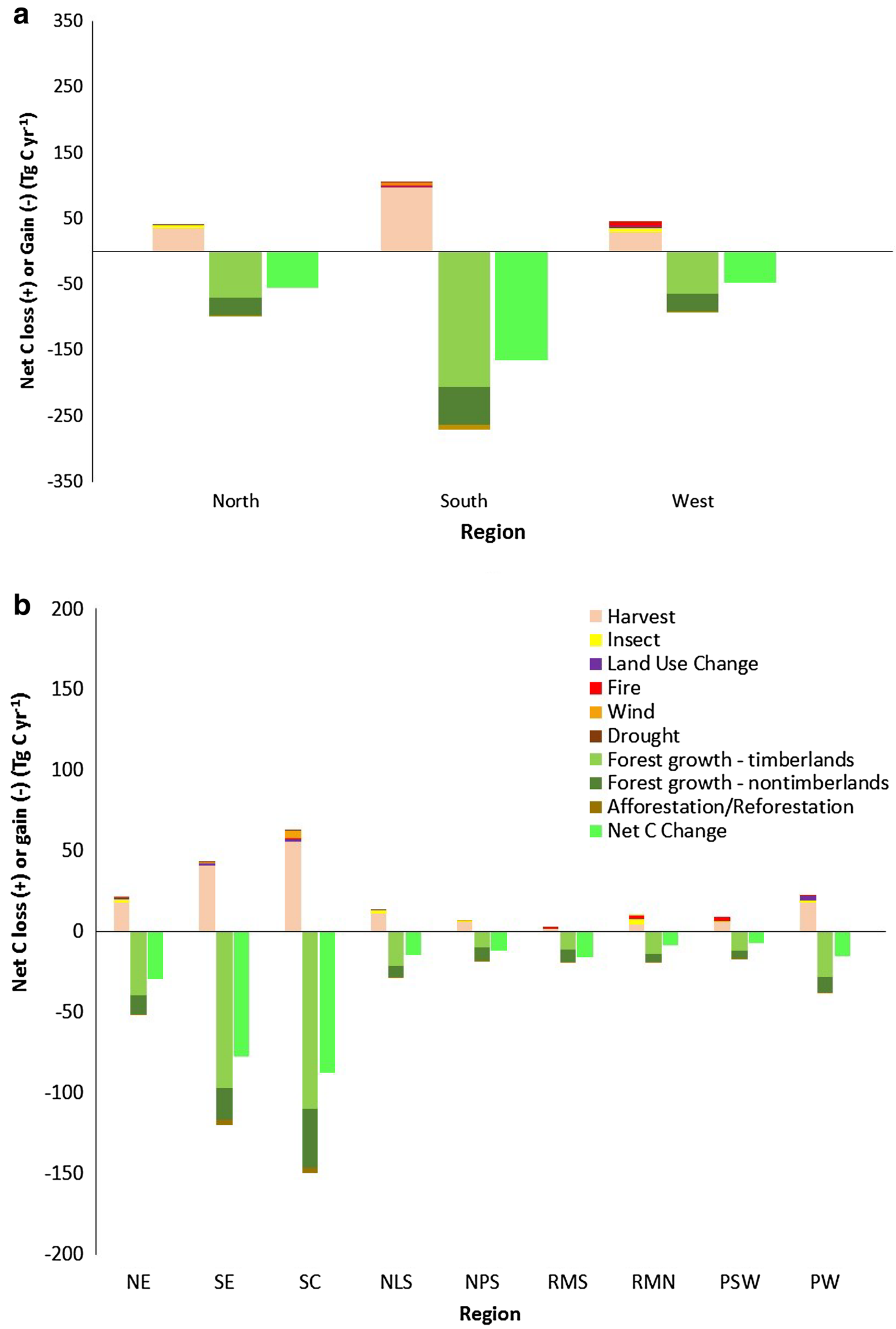

Fig. 4 Average annual net carbon change by disturbance type in **a** the North (79 million ha of forest), South (87 million ha), and West (56 million ha) regions and **b** by FIA region: northeast (NE; 41 million ha), southeast (SE; 35 million ha), southcentral (SC; 52 million ha), northern lake states (NLS; 23 million ha), northern plains states (NPS; 15 million ha), pacific west (PW; 17 million ha), rocky mountain northern (RMN; 14 million ha), rocky mountain southern (RMS; 15 million ha), and the pacific southwest (PSW; 9 million ha)

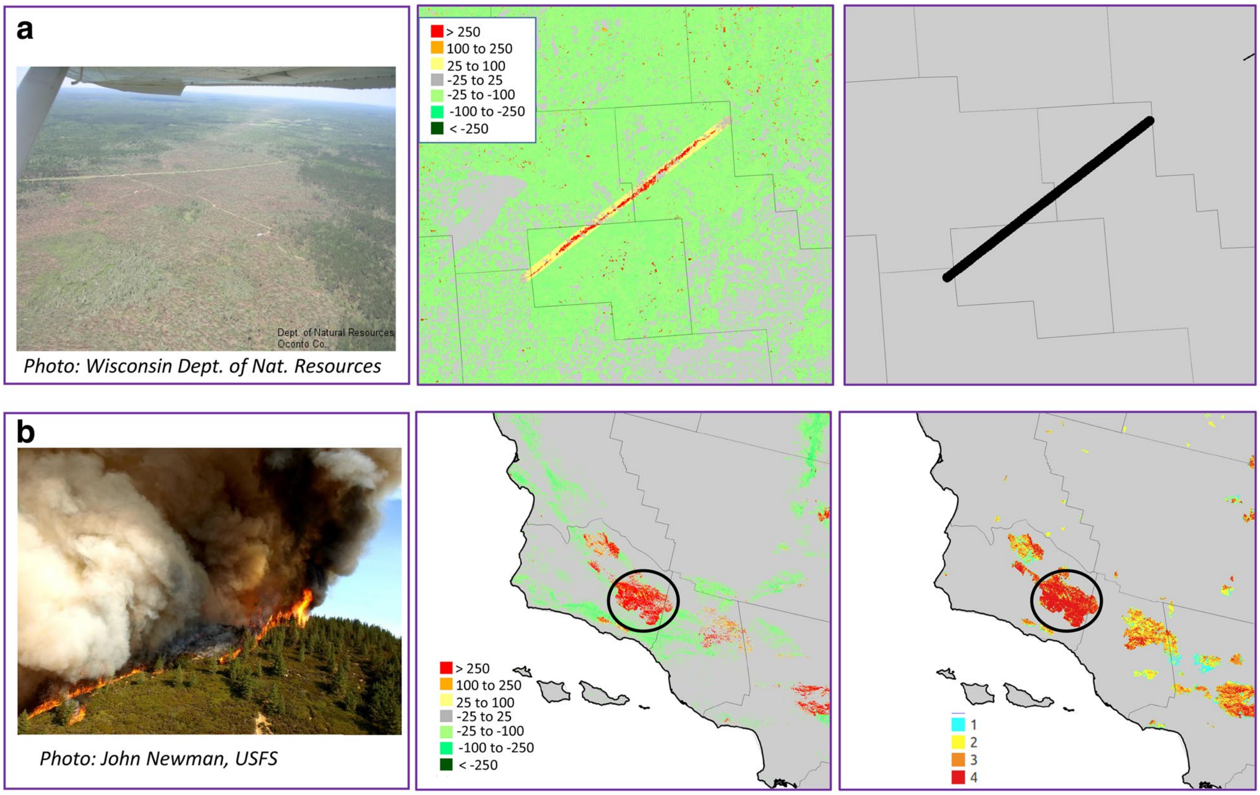

Fig. 5 The forest carbon accounting framework implemented here can be useful in assessing carbon impacts of localized disturbances. **a** 2007 tornado in Lakewood, Wisconsin. The tornado track from NOAA (*right*) resulted in extensive impacts to the forest, which is evident in an aerial photo (*left*) and in the resulting estimate of net carbon change (center, in units of Mg C ha^{-1}). **b** 2007 wild fire in southern California's Santa Barbara County, termed the "Zaca" fire. A photo of the blaze (*left*) highlights the fire intensity, which is mirrored in the burn severity map (*right*, MTBS) and the resulting net carbon change estimate (*center*, in units of Mg C ha^{-1})

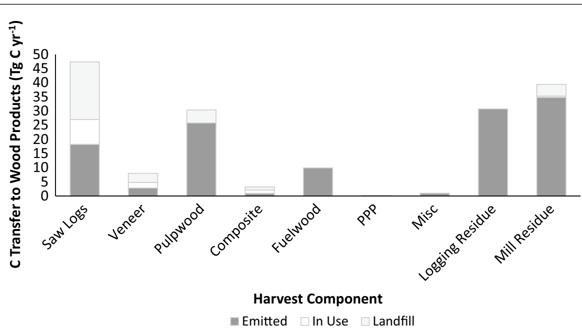

Fig. 6 Fate of C harvested from US forestlands in the year 2007, with some stored in use and landfills and the rest emitted within 100 years. *PPP* posts, poles and pilings

Priorities for FIA data collection

All forest inventory data used to estimate changes in the above- and belowground C stocks in this analysis come from FIA plots measured more than once. However, many more FIA plots have been re-measured in the North and South regions of the US than in the West. The limited number of re-measured FIA plots in the West resulted in higher uncertainties in net C stock change estimates and, in some disturbance categories, required the imputation of estimates obtained from other regions (Tables 1, 2). As the FIA program continues national implementation of an annual inventory (including re-measurement), the FIA data used in this analysis can be revised accordingly so that the sample size of plots per disturbance type increases and uncertainties decrease. Until the early 2000s, the FIA program measured only live tree attributes (e.g., tree diameter) allowing for the estimation of aboveground C and modelling of the other pools based on regions, live tree, and site characteristics (although the dead wood pool was measured in some states). Therefore, we estimated changes in the aboveground C pool using measured data while we relied on models to estimate belowground C. The FIA program is in the process of replacing model predictions of C in the dead wood, litter, and soil organic C pools with estimates obtained from measurements of these pools on a subset of FIA plots [40]. These pools, excluded from the current analysis, can be included in our framework as new data are collected.

Priorities for non-forest lands

Our analysis focused on forest areas defined in part by the NLCD data that is based on the interpretation of Landsat imagery. Comparison of our 1-ha map of carbon density of forestlands based on NLCD with high resolution lidar data over the state of Maryland has shown a significant underestimation of carbon stocks in highly fragmented and mixed urban and forest landscapes [41]. These small scale forests cover substantial areas of densely populated and fragmented landscapes of the eastern United States and appear to be highly dynamic. There is information on the disturbance and recovery of these forests over the time frame of our study, but our analysis has ignored carbon sources and sinks from these lands. By improving the carbon inventory and satellite observations to capture small scale changes, the uncertainty of carbon fluxes, particularly over the Eastern states, may be reduced. In the future (post-2020), planned satellite observations of the aboveground structure of forests by GEDI and NISAR from the National Aeronautics and Space Administration (NASA) and BIOMASS from the European Space Agency should improve the annual inventory of forest C change, as should the planned collection of FIA plot data in urban and woodland areas.

Priorities for UNFCCC reporting

Although the US has data on the magnitude of area change across land use categories, it does not have reliable and comprehensive estimates of C stocks across the entire reporting time series (e.g., 1990–2014 for the most recent UNFCCC submission) and full matrix of land use and land-use change categories to report these changes separately. For this reason, in its GHG inventory submission the US has historically deviated from IPCC guidance by reporting together C stock changes from afforestation and forest management as "forest land remaining forest land", while emissions associated with a land use conversion from forest land to a non-forest land use are reported in the non-forest land use category (per IPCC guidance). For the first time in its 2016 submission [16, 17], the US delineated net C stock changes from afforestation separately from forest land remaining forest land. An additional data need is refined C stock monitoring on non-forest lands and better coordination among land use categories to ensure complete accounting and avoidance of double counting. Our spatially resolved analysis approach allowed us to disaggregate net C change into subcategories of non-forest land to forest land (-8 ± 1 Tg C year^{-1}), forest land to non-forest land (6 ± 1 Tg C year^{-1}), and forest land remaining forest land (-267 Tg C year^{-1}). While the sole focus on net processes within the forest land use category in this study does not fully solve complete C accounting issues across all land uses, the methods used in this research are an incremental improvement toward resolving components of net C change within the forest land category, and these results can help inform and refine US reporting in the future.

Priorities for improving disturbance attribution

Insect and disease aerial detection surveys (ADS) are conducted annually using a variety of light aircraft by the USDA Forest Service in collaboration with other state and federal cooperators. Overview surveys map the current year's forest impact, and some regions have been conducting ADS for more than 60 years while others have become more active only within the last decade. Therefore, annual maps of insect damage with full coverage of all US forestlands are not available, but areas most likely to be affected by insect damage are surveyed more frequently. We accounted for the lack of continuous data coverage in our uncertainty analysis by assuming a 5% bias in underreported area. The Monitoring Trends in Burn Severity (MTBS) dataset, sponsored by the Wildland Fire Leadership Council, consistently maps the burn severity and perimeters across all lands of the US since 1984. Although 30 m resolution imagery is used for analysis, the minimum mapping unit for delineating fire perimeters is greater than 1000 acres (404 ha) in the West and 500 acres (202 ha) in the East. Therefore, burned forest areas smaller than these patch sizes were excluded from our analysis.

Priorities for wood harvest data collection

Information on the primary anthropogenic source of C loss in US forests—wood harvest—is available only at the level of combined counties. TPO data allow for the estimation of C losses from the extraction of wood products that are not readily detected by remote sensing observations, including the most recent Landsat based tree cover loss data from Hansen et al. [8]. We examined the relationship between TPO estimated C losses and a remote sensing-based estimate of C losses from forest disturbance that could not be readily linked to another disturbance type (i.e. wind, insect, fire, or conversion). For this comparative analysis, we assumed all tree cover loss pixels in Hansen et al. [8] data that could not be linked to another disturbance type were harvested, and subsequent C loss was estimated via our FIA look-up table approach. When aggregated to the state level, these two independent estimates of C loss associated with harvest were highly correlated (Fig. 7), and the remote

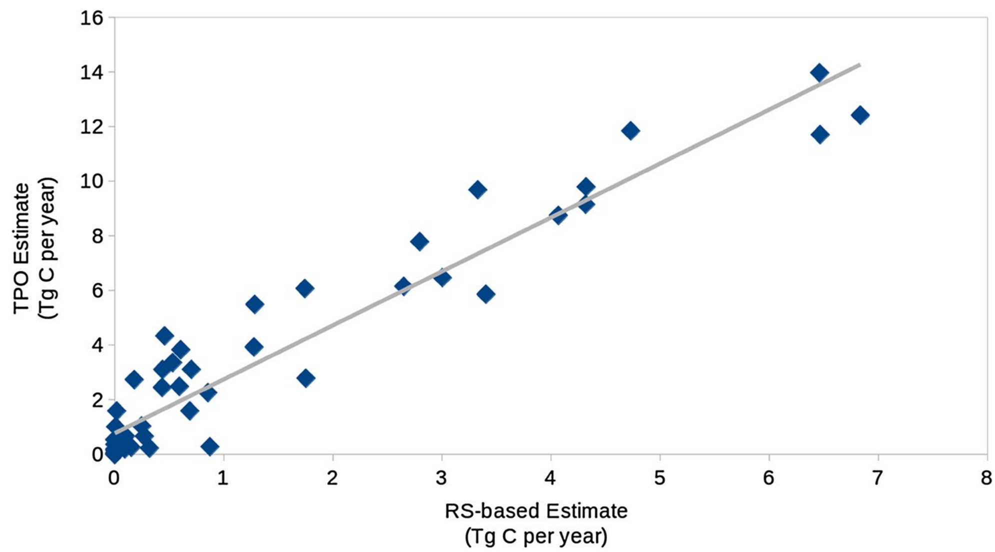

Fig. 7 Relation between C losses from harvest as estimated from timber product output (TPO) data and from an independent remote sensing-based estimate. TPO = 1.98 × RS + 767,777; $R^2 = 0.91$). Data points represent results aggregated to the state-level

sensing-based estimates of (net) C loss from harvest were approximately half of the (gross) TPO-based estimates. This provides indications that: (1) Landsat-based remote sensing observations likely miss a significant proportion of harvest activity due to partial loss, rather than full loss, of tree canopy cover; and (2) the additional C loss not identified by the remote sensing approach is spatially proximate to larger scale C losses from harvest, at least at the state scale. Increased transparency on the spatial location, timing and type of harvesting occurring across the US would allow more explicit attribution of forest C fluxes to specific forest management activities.

Managing US forests for climate change mitigation

Globally, the US ranks fourth in terms of forest area [42, 8]. Although large C losses occur from US forests as a result of an active wood products industry, particularly in the US South, 76% of the total US net carbon sink (342 Tg C year^{-1}) occurred within timberland areas, more than half of which are privately owned [43]. The income received by landowners from Intensive forest management may reduce the likelihood of forest conversion to development, but in the absence of all disturbance effects, we estimate a potential C sink between 2006 and 2010 of −460 and −436 Tg C year^{-1} if only non-harvest disturbance effects (fire, drought, wind, insect damage, land-use conversion) are considered. The US has also committed to restoring 15 Mha of forest land [44], which could further increase the C sink capacity of US forests. This implies that the US C sink could be increased substantially if existing forest land were managed to achieve this goal.

In addition to sequestering and storing atmospheric carbon, US forests also generate wood products that support the energy, industry, transport and building sectors both domestically and internationally. Given that wood harvest represents the majority of C losses from US forests, increasing the US net forest C sink would require shifts in current forest management practices as well as more refined and disaggregated information to reduce the uncertainty of these estimates and resolve these with correct estimation of net C change. For example, national debate has grown over the production of wood pellets as a renewable energy source, particularly from the southeast US, with demand driven by European policies to reduce emissions of greenhouse gases and increase the use of renewable energy. Georgia, Florida, Alabama and Virginia currently account for nearly all US wood pellet exports [45]. Although wood pellets are claimed by the industry to be made from residues at lumber mills or logging sites, the industry's growth could lead to a substantial increase in demand on Southern forests, potentially creating incentives to expand plantations. The potential of bioenergy to reduce greenhouse gas emissions inherently depends on the source of the biomass and its net land use effects; bioenergy reduces greenhouse gas emissions only if the growth and harvesting of the biomass used for energy sequesters carbon above and beyond what would be sequestered anyway [46]. This additional carbon must result from land management changes that increase tree C uptake or from the use of biomass that would otherwise decompose rapidly.

New global emphasis on climate change mitigation as one of the many benefits that forests provide gives US decision makers the opportunity to re-evaluate national and state policy agendas to consider not only the production of merchantable wood volume and biomass for bioenergy, but also enhanced C sequestration and storage for climate change mitigation. As recognized in the 2014 Farm Bill [47], there is a growing need to both reduce the uncertainty associated with estimating forest biomass and the associated monitoring of C dynamics across US forests. As it currently stands, the statistical power of detecting changes in forest C stocks exists only at large regional scales [48], disallowing the detection of C change at policy-relevant scales such as encountered in the pellet industry. Continued research to both downscale forest C inventories and correctly attribute C change to natural and anthropogenic disturbance events is needed to empower forest management policy decisions.

Conclusions

Achieving a global, economy-wide "balance between anthropogenic emissions by sources and removals by sinks" [1] will require both more emission reductions and more C sequestration from the forest sector. Results from this analysis indicate the location and estimated magnitude of C losses from different disturbances in absolute and relative terms, and can be used to track more explicitly which losses result from natural or anthropogenic disturbances. Our national net C change estimate of -269 ± 49 Tg C year^{-1} is within the range of previously reported estimates, and provides spatially explicit estimates and attribution of changes to different types of disturbances. Data are synthesized from various US agencies into a common framework, which could improve inter-agency dialogue to ensure complete accounting and to avoid double counting within and between land use categories. This work may also improve collaboration that drives a more efficient and participatory process for allocating resources towards activities that meet common goals, including an increased focus on climate change mitigation. The methodological framework and accompanying results allow US policymakers and negotiators to better understand the causes of forest C change more completely so that they can participate more effectively in domestic policy discussions about forest management and monitoring as well as in international negotiations. Integration of results from this and other studies should further enable the development of future US GHG inventories that include disturbance attribution and full land use change accounting in expectation of post-2020 commitment requirements.

Authors' contributions

NH, SH, SS, CW, SB and WS designed the study. SH, NH and TP conducted the analysis. CW, BW and GD compiled the FIA and TPO datasets. SS, YY and AF produced the biomass maps. SH implemented model runs and designed and conducted the uncertainty analysis. BB provided guidance on C modeling and on technical implementation of the methods. SH produced figures and tables. NH and SH wrote the paper. All authors reviewed the final manuscript. All authors read and approved the final manuscript.

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Competing interests

The authors declare that they have no competing interests.

Availability of data and materials

The datasets supporting the conclusions of this article are available on the ORNL DAAC website (http://daac.ornl.gov/).

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Is It Better to Plant Trees or Let Forests Regrow Naturally?

Nations are pledging to plant billions of trees. But a new study shows that we've underestimated the power of natural forest regrowth to fight climate change.

PHOTOGRAPH: VICTOR MORIYAMA/GETTY IMAGES

THIS STORY ORIGINALLY *appeared on Yale Environment 360 and is part of the Climate Desk collaboration.*

When Susan Cook-Patton was doing a postdoc in forest restoration at the Smithsonian Environmental Research Center in Maryland seven years ago, she says, she helped plant 20,000 trees along Chesapeake Bay. It was a salutary lesson. “The ones that grew best were mostly ones we didn’t plant,” she remembers. “They just grew naturally on the ground we had set aside for planting. Lots popped up all around. It was a good reminder that nature knows what it is doing.”

What is true for Chesapeake Bay is probably true in many other places, says Cook-Patton, now at the Nature Conservancy. Sometimes, we just need to give nature room to grow back naturally. Her conclusion follows a new global study that finds the potential for natural forest regrowth to absorb atmospheric carbon and fight climate change has been seriously underestimated.

Tree planting is all the rage right now. This year's World Economic Forum in Davos, Switzerland, called for the world to plant a trillion trees. In one of its few actions to address climate concerns, the US administration—with support from businesses and nonprofits such as American Forests—last month promised to contribute close to a billion of them—855 million, to be precise—across an estimated 2.8 million acres.

The European Union this year promised 3 billion more trees as part of a Green Deal; and existing worldwide pledges under the 2011 Bonn Challenge and the 2015 Paris Climate Accord set targets to restore more than 850 million acres of forests, mostly through planting. That is an area slightly larger than India, and it provides room for roughly a quarter-trillion trees.

Planting is widely seen as a vital "nature-based solution" to climate change—a way of moderating climate change in the next three decades as the world works to achieve a zero-carbon economy. But there is pushback.

Nobody condemns trees. But some critics argue that an aggressive drive to achieve planting targets will provide environmental cover for land grabs to blanket hundreds of millions of acres with monoculture plantations of a handful of fast-growing and often nonnative commercial species such as acacia, eucalyptus, and pine. Others ask: Why plant at all, when we can often simply leave the land for nearby forests to seed and recolonize? Nature knows what to grow and does it best.

Cook-Patton's new study, published in *Nature* and coauthored by researchers from 17 academic and environmental organizations, says estimates of the rate of carbon accumulation by natural forest regrowth, endorsed last year by the UN's Intergovernmental Panel on Climate Change, are on average 32 percent too low, a figure that rises to 53 percent for tropical forests.

The study is the most detailed attempt yet to map where forests could grow back naturally and to assess the potential of those forests to accumulate carbon. "We looked at almost 11,000 measurements of carbon uptake from regrowing forests, measured in around 250 studies around the world," Cook-Patton told Yale Environment 360.

She found that current carbon accumulation rates vary by a factor of a hundred, depending on climate, soils, altitude, and terrain. This is much greater than previously assessed. "Even within countries there were huge differences." But overall, besides being better for biodiversity, the study showed, natural regeneration can capture more carbon more quickly and more securely than plantations.

Cook-Patton agrees that as climate change gathers pace in the coming decades, rates of carbon accumulation will change. But while some forests will grow more slowly or even die, others will probably grow faster due to the fertilization effect of more carbon dioxide in the air, an existing phenomenon sometimes called global greening.

The study identified up to 1.67 billion acres that could be set aside to allow trees to regrow. This excludes land under cultivation or built on, along with existing valuable ecosystems such as grasslands and boreal regions, where the warming effects of dark forest canopy outweigh the cooling benefits of carbon take-up.

Combining the mapping and carbon accumulation data, Cook-Patton estimates that natural forest regrowth could capture in biomass and soils 73 billion tons of carbon between now and 2050. That is equal to around seven years of current industrial emissions, making it "the single largest natural climate solution."

Cook-Patton said the study's local estimates of carbon accumulation fill an important data gap. Many countries intent on growing forests to store carbon have data for what can be achieved by planting, but lack equivalent data for natural regeneration. "I kept getting emails from people asking me what carbon they would get from [natural] reforesting projects," she says. "I had to keep saying it depends. Now we have data that allow people to estimate what happens if you put up a fence and let forest regrow."

The new local estimates also allow comparisons between the potential of natural regrowth and planting. “I think planting has its place, for instance where soils are degraded and trees won’t grow,” she said. “But I do think natural regrowth is hugely underappreciated.”

THE GREAT THING about natural restoration of forests is that it often requires nothing more than human inaction. Nature is constantly at work restoring forests piecemeal and often unseen on the edges of fields, on abandoned pastures, in scrubby bush, and wherever forests lie degraded or former forest land is abandoned.

But because it requires no policy initiatives, investments, or oversight, data on its extent is badly lacking. Satellites such as Landsat are good at identifying deforestation, which is sudden and visible; but the extent of subsequent recovery is slower, harder to spot, and rarely assessed. Headline grabbing statistics on the loss of the world’s forests generally ignore it.

In a rare study, Philip Curtis of the University of Arkansas recently attempted to get around the problem by devising a model that could predict from satellite imagery what had caused the deforestation, and hence the potential for forest recovery. He found that only about a quarter of lost forests are permanently taken over for human activities such as buildings, infrastructure, or farming. The remaining three-quarters suffered from forest fires, shifting cultivation, temporary grazing, or logging, and at least had the potential for natural recovery.

Another study published this year found that such recovery was widespread and rapid even in an epicenter of deforestation such as the Amazon. When Yunxia Wang of the University of Leeds in England analyzed recently-released Brazilian data from the Amazon, she found that 72 percent of the forest being burned by ranchers to create new cattle pasture is not pristine forest, as widely assumed, but is actually recent regrowth. The forest had been cleared, converted to cattle pasture and then abandoned, whereupon the forest returned so fast that it was typically only six years before it was cleared again. Such was the confusion caused by this rapid forest turnover that regular land-use assessments frequently wrongly categorized this new growth as degraded old-growth forest.

Wang noted that if Brazil’s president, Jair Bolsonaro, wanted to fulfill a promise made by one of his predecessors, Dilma Rousseff, at the 2015 Paris climate summit to restore 30 million acres of forest by 2030, then he need not plant at all. He could just allow regrowth to proceed in the Amazon without further clearing.

Brazil's other great forest, the Atlantic Forest, is already on that path, recovering slowly after more than a century of clearance for coffee and cattle. The government has an Atlantic Forest Restoration Pact that subsidizes landowners to replant, often with trees intended to supply the paper industry. Yet Camila Rezende of the Federal University of Rio de Janeiro says most of the forest regrowth is not from planting but from "spontaneous" regrowth, as forest remnants colonize neighboring abandoned farmland. She estimates that some 6.7 million acres of Atlantic Forest have naturally regenerated in this way since 1996. That land now makes up about a tenth of the forest.

Much the same has been happening in Europe, where forest cover is now up to 43 percent, often from naturally recolonizing farmland rather than planting. Italy, for instance, has grown its forest cover by 2.5 million acres. In the former communist nations of central Europe, 16 percent of farmland in the Carpathian Mountains was abandoned in the 1990s, much of it reclaimed by the region's famed beech forests. Across Russia, an area of former farmland about twice the size of Spain has been recolonized by forests. Irina Kurganova of the Russian Academy of Sciences calls this retreat of the plow "the most widespread and abrupt land-use change in the 20th century in the Northern Hemisphere."

The United States has also seen natural forests regenerate as arable farmland has declined by almost a fifth in the past 30 years. "The entire eastern United States was deforested 200 years ago," says Karen Holl of UC Santa Cruz. "Much of that has come back without actively planting trees." According to the US Forest Service, over the past three decades the country's regrowing forests have soaked up about 11 percent of national greenhouse gas emissions.

With nature on the march, a major concern is whether a push for planting might grab land for plantations that natural forests might otherwise recolonize. The result would be less wildlife, less amenity for humans, and often less carbon stored.

ECOLOGISTS HAVE TRADITIONALLY dismissed the ecological gains from natural restoration of what is often called "secondary" forest. Such regrowth is often regarded as ephemeral, rarely sought out by wildlife, and prone to being cleared again. This has led many to regard planting to mimic natural forests as preferable.

Thomas Crowther, coauthor of a widely publicized study last year calling for a "global restoration" of a trillion trees to soak up carbon dioxide, emphasizes that, while nature could do the job in places, "people need to help out by spreading seeds and planting saplings."

But a reappraisal is going on. J. Leighton Reid, director of Restoration Ecology at Virginia Tech, who recently warned against bias in studies comparing natural regeneration with planting, nonetheless told e360, "Natural regeneration is an excellent restoration strategy for many landscapes, but actively reintroducing native plants will still be a better option in highly degraded sites and in places where invasive species dominate."

Others make the case that most of the time, natural restoration of secondary forests is a better option than planting. In her book, *Second Growth*, Robin Chazdon, a forest ecologist formerly at the University of Connecticut, says that secondary forests "continue to be misunderstood, understudied, and unappreciated for what they really are—young self-organizing forest ecosystems that are undergoing construction."

Yes, she agrees, they are work in progress. But they generally recover "remarkably fast." Recent research shows that regrowing tropical forests recover 80 percent of their species richness within 20 years, and frequently 100 percent within 50 years. That seems to be better than what human foresters achieve when trying to replant forest ecosystems.

A review of more than 100 tropical forest restoration projects by Renato Crouzeilles of the International Institute for Sustainability in Rio de Janeiro, with Chazdon as a coauthor, found that success rates were higher for secondary forests allowed to regenerate naturally than for those subjected to the "active restoration" techniques of foresters. In other words, planting can often worsen outcomes for everything from the number of bird, insect, and plant species to measures of canopy cover, tree density, and forest structure. Nature knows best.

Now, Cook-Patton has extended the reappraisal to the carbon-accumulating potential of natural forest regeneration. It too may often be superior.

This scientific rethink requires a policy rethink, says Holl. "Business leaders and politicians have jumped on the tree-planting bandwagon, and numerous nonprofit organizations and governments worldwide have started initiatives to plant billions or even trillions of trees for a host of social, ecological, and aesthetic reasons."

She concedes that on some damaged lands, "we will need to plant trees, but that should be the last option, since it is the most expensive and often is not successful."

Planting a trillion trees over the next three decades would be a huge logistical challenge. A trillion is a big number. That target would require a thousand new trees in the ground every second, and then for all of them to survive and grow. Once the cost of nurseries, soil preparation, seeding, and thinning are accounted for, says Crouzeilles, it would cost hundreds of billions of dollars. If natural forest growth is cheaper and better, does that make sense?

Ocean Researcher Debunks Ocean Myths

OCEARCH founder and ocean explorer Chris Fischer has been working to protect our oceans and the wildlife that inhabits them. He sits down with WIRED to talk over some common myths associated with the ocean. Is the Bermuda Triangle real? Can sharks smell blood from a mile away? Can the ocean absorb anything?   Footage courtesy of OCEARCH

TOPICS CLIMATE DESK ENVIRONMENT CLIMATE CHANGE ECOLOGY CONSERVATION TREES

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A New Direction for California Wildfire Policy—Working from the Home Outward

February 11, 2019

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A New Direction for California Wildfire Policy—Working from the Home Outward

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Executive Summary

California's state policies on wildfire need to change direction. The current policies are failing. They have not effectively protected homes, while they place dramatically increasing pressures on state and local budgets. Moreover, these policies are often based on notions about the role of fire in California's ecosystems that are not supported by sound science and do not reflect the changing climate. These policies try to alter vast areas of forest in problematic ways through logging, when instead they should be focusing on helping communities safely co-exist with California's naturally fire-dependent ecosystems by prioritizing effective fire-safety actions for homes and the zone right around them. This new direction—working from the home outward—can save lives and homes, save money, and produce jobs in a strategy that is better for natural ecosystems and the climate.

The impetus for this report is the Governor's Executive Order N-05-19, which instructed CalFire to develop wildfire policy recommendations for California. To help Governor Newsom chart a new evidence-based approach to these policies, the Leonardo DiCaprio Foundation invited experts from our partner organizations to prepare concise synopses of key points that are not likely to be included or emphasized in CalFire's recommendations. Those synopses are compiled in this report. In addition, we have prepared a list of specific steps that can help California embark on a new approach to wildfire policy that prioritizes home and community safety and works from the home outward.

Top recommendations include:

- Convene a task force focused specifically on wildfire safety for homes and communities, consisting of experts on home-safety features and community planning
- Ensure that the Governor has a diverse set of advisors on wildfire and forest policy, including experts who are not primarily advocating for logging-based strategies
- Direct SB 901 funds and other resources to prioritize support for retrofitting of homes that need to be more fire-safe and other home-safety actions

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A New Direction for California Wildfire Policy—Working from the Home Outward

Introduction

by Douglas Bevington, PhD, Forest Director, California Program, Leonardo DiCaprio Foundation

The Problem:
California's state policies on wildfire need to change direction. Those policies are currently steering resources into trying to alter vast areas of forest in problematic ways, when instead they should be focusing on helping communities safely co-exist with California's fire-dependent ecosystems by prioritizing effective fire-safety actions for homes and the zone around them.

In order to solve a problem, it needs to be defined clearly. Amid the effects of climate change, California is experiencing unprecedented levels of home destruction and loss of human life during wildfires, and fire suppression spending is bigger than ever. California has a human-safety problem during fires and also an economic problem from spiraling fire suppression costs, but California does not have an unnatural excess of forest fire in terms of either amount or severity. While recent fires are described as "record" in size, those statements are based on records from after California began suppressing fire. Prior to the advent of 20th century mechanized suppression, California's forests naturally experienced much more fire than now. Our forests need fire as an ecosystem process, and they naturally burn in a mixture of low, medium, and high-severity. (For peer-reviewed studies on these points, see pp. 12-13.)

California's current fire policies focus on how to do massive forest alterations, mainly through logging, to try to alter fire severity. Those policies are trying to address the wrong problem. Our forests do not need reduced fire amount or severity to be healthy. Moreover, altering forests to try to change fire severity is largely irrelevant to keeping homes safe during fires. Most home ignitions are not caused by coming into contact with high-severity fire (Syphard et al. 2017). For example, in the 2007 Grass Valley Fire, contact with high-severity forest fire was only responsible for 3% of the burned houses. The other 97% were due to low-severity fire, wind-blown embers, and flames from other houses (https://tinyurl.com/y33bdu9s). (This pattern can be readily seen in other fires in which burned houses are often next to unburned green trees.) Policies to address impacts to communities that are based on more logging as the solution, to try to alter fire severity, are an inefficient and ineffective way to protect homes.

Instead, research shows that the most effective steps to prevent homes from burning involve incorporating fire-safe features on buildings (e.g., roof materials, vent screens) and pruning vegetation in the zone 100 feet around houses (see pp. 8-9). When properly implemented, this approach works effectively even when faced with intense wildfires amid high temperatures and high winds, such as during the La Tuna Fire, in which more than 99% of houses within the fire path remained unburned (http://www.latimes.com/opinion/op-ed/la-oe-hanson-latuna-fire-homes-20180810-story.html). And these home-safety actions can produce jobs for rural communities (http://nreconomics.com/reports/2018-04-28_EnvNow_Report.pdf).

We need a policy focus that starts from the home outward, yet currently much of the attention and resources are being redirected to logging of vast forest areas far away from homes.

Calls for large-scale forest alterations to try to change fire severity are often based on erroneous claims that do not reflect a growing body of scientific research (see pp. 12-13) showing that:

- mixed-severity fire is a natural and necessary component of California's forests
- there is less forest fire of all severities now than there naturally should be
- logging has caused a shortage in the total volume of biomass/carbon in our forests now

Current forest-altering policies promote subsidized logging and biomass extraction that:

- take resources away from the actions that most effectively keep homes safe during fires
- are costly to taxpayers
- cause damage to forest ecosystems
- contribute to global warming by releasing stored forest carbon into the atmosphere

Associated efforts to promote forest extraction by including biomass in the state's Renewable Portfolio Standard and legislation that requires forest bioenergy procurement result in:

- increased costs for utility ratepayers
- utilities forced to select biomass power sources that are more expensive than solar and that emit more carbon dioxide than coal per unit of energy generated
- resources pulled away from zero-emission energy sources such as solar
- California biomass policies that are similar to those of the Trump administration

The Causes of this Problem:

For wildfire-related matters, California's officials and agencies have been relying too heavily on the recommendations of CalFire and the US Forest Service. These agencies have spent many decades promoting logging and intensive fire suppression, an approach that has produced high costs and poor results. Scientists widely agree that fire suppression has harmed forest ecosystems. And efforts to blame forest protection for current forest fire behavior ignore research results showing that forests with the highest levels of restrictions on logging burn at lower severities compared to forests with fewer restrictions on logging (https://esajournals.onlinelibrary.wiley.com/doi/full/10.1002/ecs2.1492). Yet, CalFire and the US Forest Service continue to advocate spending more on large-scale logging (using euphemisms such as "thinning" and "management") as a primary emphasis of fire policies.

The resulting policymaking processes have drawn heavily on US Forest Service-funded scientists while avoiding or misrepresenting the peer-reviewed research of independent scientists whose findings refute the justifications used to promote logging (e.g., https://tinyurl.com/y9sqmp76).

The current approach continues to pull resources away from actions directly around homes that would help communities to safely co-exist with fire-dependent ecosystems in California. And each time homes are lost, the same voices keep on calling for even more funding to be poured into the current failing strategies. It is time for a new direction guided by new voices.

The Purpose for this Report:

The immediate impetus for this report is the Governor's Executive Order N-05-19, which instructed CalFire to develop wildfire policy recommendations for California, due to be released later this month. If past is prologue, this document will be likely focused on redoubling the failed suppression and forest-alteration strategies that have dominated CalFire's approach so far. It will be built on fundamentally erroneous claims about the role of fire in California forests that exclude key scientific research on this subject (for examples, see pp. 12-13 of our report). CalFire may continue to apply what is in effect still a 20th century fire suppression strategy that is not appropriate for our 21st century climate (see pp. 10-11). There may be mention of 21st century technologies such as drones, but they will likely be applied in support of outdated suppression goals. There may even be some greater attention on prescribed fire, but if this tool is simply used in support of an outdated suppression strategy, the outcome will be problematic (see p. 13). And while CalFire may talk about the problem of climate change, its recommended policies are likely to be detrimental to the climate (see pp. 14-17). Above all, while there may be some mention of defensible space and houses, the overall outcome of CalFire's recommendations will likely be to direct funding mainly to suppression and logging, rather than redirecting resources to where they can be most effective by focusing on retrofitting homes and communities to be prepared for the inevitable wildfires in California's fire-dependent ecosystems (see pp. 8-9, 19).

To chart a new approach to wildfire policies in California, Governor Newsom will need to seek advice beyond the voices that have steered us into the current failed policies. To help address this need, the Leonardo DiCaprio Foundation invited experts from our partner organizations to prepare concise synopses of key points that are not likely to be included or emphasized in CalFire's recommendations (pp. 10-19). In addition, we are honored to include a piece by Dr. Jack Cohen, who recently retired from the US Forest Service (pp. 8-9). Dr. Cohen has been a pioneer in the study of the importance of home features and the zone right around them for preventing home ignitions during wildfires. Despite the significant implications of Dr. Cohen's research, not nearly enough has been done to incorporate these findings into current fire policies. As a recent article summarized, "Cohen thought he had come up with a way to save houses and to let fires burn naturally—he thought it was a win-win. And so in 1999, he presented a paper about his findings at a fire conference in front of people from the Forest Service and state fire agencies. These were people who were in a position to change policies. But Cohen says they were totally uninterested. Cohen's research implied that basically everything about how the Forest Service dealt with wildfires was wrong."(https://tinyurl.com/yb4rt45r) Through the research presented in this report, we hope to show that there is now an opportunity to take California's wildfire policies in a positive and effective new direction.

Solutions:

In light of these findings, we urge Governor Newsom to seek guidance beyond the CalFire recommendations before setting the course of California's wildfire policies. In particular, we recommend that he convene a task force focused specifically on wildfire safety for homes, consisting of experts on home-safety features and community planning. (The composition of this task force would therefore be different from the Forest Management Task Force). This task force should identify the most effective and cost-efficient actions to prevent home ignitions during wildfires, including potential roles for state policies and resources to support retrofitting of homes that need to be more fire-safe. By focusing resources on preparing homes and communities to safely coexist with inevitable wildfires through a new approach that works from the home outward, we can save lives and homes, save taxpayers' money, and produce jobs in a strategy that is better for California's natural ecosystems and the climate.

Recommendations

Based on the research cited in this report, we recommend that following steps can help state wildfire policies shift to a focus on safety and cost-efficiency by working from the homes outward, while avoiding subsidizing unnecessary logging:

➢ Convene a task force focused specifically on wildfire safety for homes and communities, consisting of experts on home-safety features and community planning. (The composition of this task force would therefore be different from the Forest Management Task Force). This task force should identify the most effective and cost-efficient actions to prevent home ignitions during wildfires, including potential roles for state policies and resources to support retrofitting of homes that need to be more fire-safe

➢ Ensure that the Governor has advisors on wildfire and forest policy beyond those primarily advocating for logging-based strategies, including:

- Environmental groups that are actively challenging harmful logging projects, so as to better understand the science-based concerns with current projects
- Scientists who are not financially dependent on the US Forest Service
- Experts on defensible space and forest carbon
- Fire management experts affiliated with the National Park Service
- Experts on chaparral and non-conifer forest ecosystems where much of the recent home losses have occurred

➢ Take a leadership role on setting better standards for making homes fire-safe throughout California, and link eligibility for fire/forest-related state funds to the extent to which communities implement these fire-safety measures

➢ Direct SB 901 funds to home-safety actions rather than logging

➢ Remove forest biomass from the Renewable Portfolio Standard and do not mandate utility use of expensive biomass power sources

➢ Conduct independent review and reform of the SB 901-mandated forest carbon calculator

➢ Do not use California state funds to subsidize logging on national forests

➢ Revise CalFire's policies to better fit 21st century climate conditions, including independent review of the costs and impact from CalFire's use of large airtankers (see p. 11)

➢ Shift more resources from wildland fire suppression to municipal fire departments on the frontlines of keeping homes safe

➢ Support research and public education about the many benefits of retrofitting homes to become more fire-safe, including job-creation and reduction of loss of life and property

➢ Prevent unplanned human-caused wildfire ignitions, including by increasing the pace at which utilities bury their powerlines underground. This action will reduce a key fire ignition source while simultaneously avoiding other problems with aboveground powerlines.

A More Effective Approach for Preventing Wildland-Urban Fire Disasters

By Jack Cohen, PhD, Research Physical Scientist, US Forest Service, retired

Summary

Communities exposed to inevitable extreme wildfire conditions do not have to incur inevitable disastrous fire destruction. Research shows that the characteristics of a home and its immediate surroundings within 100 feet (30 meters) principally determine home ignitions. This area, called the home ignition zone (HIZ), defines wildland-urban (WU) fires as a home ignition problem and not a problem of controlling wildfires. Communities can readily reduce home ignitability within the HIZ to prevent WU fire disasters instead of increasing wildfire suppression that fails during extreme wildfire conditions. Reducing the ignition conditions within the HIZ to produce ignition resistant homes provides an effective alternative for preventing WU fire disasters without necessarily controlling extreme wildfires.

Inevitable Wildfires and Extreme Burning Conditions

Wildfire occurrence is inevitable and thus, a small percentage of wildfires will inevitably attain uncontrollable extreme wildfire conditions. For over one-hundred years U.S. fire suppression has successfully controlled 95 to 98 percent of wildfires with initial attack (Stephens and Ruth 2005). However, there is no historical evidence or current fire management trend to suggest that all wildfires can be excluded and if not excluded, controlled with an initial suppression response. Thus, we can assume the inevitability of wildfires and the occurrence of extreme wildfire conditions (Williams 2013). Most wildfires controlled at initial attack occur during moderate to high wildfire conditions. During severe conditions of drought, high winds, low relative humidity and multiple ignitions, 2 – 5 percent of the wildfires producing rapid growth with high burning intensities escape initial attack suppression.

The primary federal, state and local approach for protecting structures from wildfires and preventing community fire disasters is wildfire control using suppression added by pre-suppression fuel breaks and shrub and forest fuel treatments (Finney and Cohen 2003, Cohen 2010). However, disastrous community wildfire destruction (greater than 100 homes destroyed) has only occurred during extreme wildfire conditions when high wind speeds, low relative humidity and continuous flammable vegetation result in rapid fire growth rates and numerous spot ignitions from showers of burning embers (firebrands); that is, the conditions when wildfire control fails (Cohen 2010, Calkin et al. 2014).

Community fire destruction during wildfires will continue as long as wildfire suppression continues to be the primary residential protection approach. The inevitability of uncontrolled extreme wildfires suggests inevitable disastrous home destruction; however, research on how homes ignite during extreme wildfires indicates practical opportunities for effectively creating ignition resistant homes and thereby preventing community fire disasters without necessarily controlling wildfires (Cohen 2000; Cohen 2001; Cohen 2004; Cohen and Stratton 2008; Cohen 2010; Calkin et al. 2014; Cohen 2017). We can immediately see how homes were not ignited during a wildfire from the readily observable patterns of destruction.

Patterns of Home Destruction during Wildfires

Total home destruction surrounded by green tree canopies following the Camp Fire in Paradise, CA (Figure 1, left photo) has been reported as unusual; however, unconsumed vegetation adjacent to and surrounding total home destruction is the typical WU fire pattern associated with extreme wildfire conditions (Cohen 2000; Cohen and Stratton 2003; Cohen 2003; Cohen and Stratton 2008; Graham et al. 2012; Cohen 2017). The center photo (Figure 1) shows an example of a burning home that could have only ignited from lofted burning embers (firebrands) on the home and low intensity surface fire spreading to contact the home. The three photos (Figure 1) of home destruction with adjacent unconsumed shrub and tree vegetation indicate the following:

Figure 1.
Paradise, CA; 2018 Camp Fire

Southwest CO; 2002 Missionary Ridge Fire

S Cal; 2007 Grass Valley Fire

- ***High intensity wildfire did not continuously spread through the residential area as a wave or flood of flame.***
- ***Unconsumed shrub and tree canopies adjacent to homes did not produce high intensity flames that ignited the homes; ignitions could only be from firebrands and low intensity surface fires.***
- ***The 'big flames' of high intensity wildfires did not cause total home destruction.***

Figure 2.

Figure 3.

High intensity wildfires do not spread through residential areas such as Paradise. The continuous tree and shrub canopies required to maintain high intensity wildfire spread (crown fires) are broken by fuel gaps such as streets, driveways and home sites (Cohen 2010). Figure 2 shows how a crown fire spread to but could not continue beyond the first residential street. Although the crown fire terminated at the street, firebrands showered downwind into the residential area initiating fires resulting in several blocks of total home destruction (Cohen 2010). Extreme wildfire conditions initiate ignitions within residential areas but the residential fuels, structures and vegetation continue the residential burning resulting in total home destruction. Commonly, homes ignite and burn hours after the wildfire has ceased active burning near the community (Cohen and Stratton 2008, Cohen 2010).

Furthermore, the typical WU fire patterns indicate that conditions local to a home principally determine home ignitions with firebrands the principal source of ignitions within the residential area. The totally destroyed home in Figure 3 indicates firebrands as the only possible ignition source, potentially igniting the home directly and the flammable materials adjacent to the home. Firebrands are a given during extreme WU fire conditions; however, regardless of the distance firebrands were lofted, firebrand ignitions depend on the local conditions of the ignitable surfaces on or adjacent to a home.

An Effective Approach for Preventing WU Fire Disasters

Research (Cohen 2004) has quantified "local ignition conditions" to be an area of a home and its immediate surroundings within 100 feet (30 meters). This area is called the home ignition zone (HIZ) (Cohen 2010; NFPA 2018). The relatively small area of the HIZ principally determines home ignitions during extreme wildfires and defines WU fire destruction as a home ignition problem that can be prevented by readily addressing home ignition vulnerabilities within the HIZ without necessarily controlling wildfires. For example, an ignition resistant home does not have a flammable wood roof, flammable tree debris on the roof, in the rain gutters, on decks or on the ground within 5 feet (1.5 m) of flammable siding, no open firewood within 30 feet (9 m), or unscreened vents. Clearing the HIZ of vegetation is not necessary. As indicated by the typical patterns of WU fire destruction, shrub and tree canopies are not spreading high intensity fires through communities. The inevitability of uncontrolled extreme wildfires spreading to communities does not mean WU fire disasters are inevitable if we address the problem with the readily available approach of reducing home ignitability. Ignition resistant communities increase community fire protection effectiveness, life-safety options for residents and firefighters, and decrease wildfire suppression costs while preventing WU fire disasters without attempting to protect communities by controlling wildfires.

CalFire's 20th Century Fire Suppression Policy is Not Appropriate for a 21st Century Climate

by Timothy Ingalsbee, PhD, Executive Director, Firefighters United for Safety, Ethics, and Ecology

Up until the mid-20th century, we had a lot more fire on the land
Hundreds of fire history studies document that wildland fires burned significantly more area than burns now. Even in the 20th century up until the 1950s, several tens of millions of acres burned in the U.S. each year (NIFC).

Then we began mechanized firefighting in the 20th century
Federal agencies such as the U.S. Forest Service began fighting fires in 1905, but with minimal effectiveness due to the large expanse of undeveloped wildlands, the limited size of its workforce, and primitive technology. This changed in the post-World War II period with an influx of military surplus vehicles and equipment in fire suppression (Pyne 1982). Cutting firelines with bulldozers and airtankers dropping chemical retardants brought annual burned acreage crashing down. In California alone there was a 36% decline in area burned from the 1940s to the 1950s, the start of a trend of rapidly declining acres burned that continued until the 1980s (CalFire-A n.d.). This created a historically unprecedented shortage of fire on the landscape that is still adversely affecting fire-adapted ecosystems across the west.

But the post-war surge of suppression success accompanied a change in climate
At the same time that mechanized firefighting was pushing deeper into backcountry wildlands and containing nearly all wildfires at a small size, the climate had changed. A prolonged cool, wet period from a natural cycle of climate variability called the Pacific Decadal Oscillation (PDO) greatly aided firefighters' efforts in stopping wildfire spread (Littell et al. 2009, Peterson et al. 2011). This created an unprecedented shortage of fire on the landscape during the 1950s and 60s. During this post-war period with its anomalously and artificially low level of wildfire activity, people developed a distorted perception of wildfires as absolutely bad, along with a false sense of security that firefighters could put them all out (Murphy et al. 2018).

21st century climate change is making wildfires start easier and spread faster
At the end of the 20th century that cool, wet PDO cycle ended and was replaced with much warmer and drier conditions that are now being amplified by global warming from fossil-fuel emissions. Prolonged droughts punctuated by frequent severe fire weather conditions (high temperatures, high winds, and low relative humidity) are making vegetation ignite much easier and fires spread more rapidly. Beginning in the 1980s but accelerating after 2000, the signal of anthropogenic climate change is now registering in greatly increased wildfire activity that is leading to longer fire seasons and increased amount of acres burned.
But even this recent increase in large fires masks the fact that there still much less fire on the land than is necessary for maintenance of California's fire-adapted forest ecosystems (Sugihara et al 2006).

21st century climate is ending the efficacy of conventional firefighting
Conventional firefighting tactics of dumping retardant, cutting firelines, and lighting backfires cannot stop wind-blown flames from jumping over firelines or firebrands lofting in the sky and landing on flammable rooftops miles away from a wildfire's flaming front. Now that 21st century anthropogenic global warming is causing severe fire weather conditions to become more frequent, the efficacy of conventional suppression is further declining. Conventional firefighting strategies and tactics are unable to either prevent or suppress large wildfires that are now being driven by climatic conditions that will be with us for the far foreseeable future.

Suppression spending is soaring

In response to increasing wildfire activity, both federal and state agencies have been dramatically escalating their suppression spending over the last 30 years. For example, in 1986 CalFire spent only $15 million total on suppression, but in 2017 the agency spent a record $947 million, far exceeding its budget (CalFire 2018). In all but one year in the 21st century CalFire has spent over $100 million—and sometimes several hundreds of millions—on firefighting, a huge surge in spending from earlier decades. But CalFire's tactics remain rooted in a suppression-based approach that is proving more and more expensive and less and less effective in a 21st century climate. In fact, the last four years have seen the highest suppression spending in CalFire's existence—accompanied by huge urban fire disasters and record numbers of homes destroyed.

Expanding the fleet of airtankers would be a poor investment of taxpayer dollars

A signature example of a costly and increasingly ineffective 20th century approach to fire suppression is the emphasis on airtankers. Airtankers are one of the most expensive resources used in wildfire suppression, but several recent studies have found that airtankers are routinely deployed at times, places, and conditions where they are least useful or effective (Stonesifer et al. 2016; Stonesifer et al. 2015; Calkin et al. 2014; Thompson et al. 2012). They are particularly likely to be impaired by high winds associated with severe fire weather. CalFire regulation 8362.3.1.1 requires airtankers to be grounded when there is even moderate turbulence or windspeeds exceeding 35 mph (CalFire-B n.d.) Heavy smoke is another impediment to effective airtanker use. For example, while the Camp Fire raged through Paradise, a fleet of airtankers located literally next door in Chico was grounded by high winds and dense smoke.

Fighting fires in backcountry wildlands depletes resources needed to protect communities

Systematic attempts to exclude or suppress all fires regardless of whether or not they are near communities is costly to taxpayers and puts communities at risk from lack of suppression crews and resources actually protecting homes. For example, in 2016 a joint CalFire/USFS effort spent over $262 million on the Soberanes Fire that burned mostly in the Ventana Wilderness Area and became the most expensive wildfire suppression operation in U.S. history (Ingalsbee et al. 2018). A USFS internal investigation (USDA-FS 2017) concluded that the excessive spending reflected "systemic fire management issues" revolving around lack of fiscal accountability that have yet to be solved. These large expenditures on fire suppression in remote areas pull limited resources away from the actions that are most effective at preventing home loss during fires.

Recommendations:

➢ Wildland fires are ecologically necessary and inevitable, but losses of life and property in urban fire disasters need not be inevitable if we adopt new fire management policies and practices suitable for 21st century climate conditions. We need to move away from 20th century mechanized fire suppression strategies, tactics, and tools (e.g., large airtankers) that are inappropriate and increasingly ineffective in the current climate.

➢ Suppression resources should be redirected away from fighting fires in remote wildlands where fire is ecologically necessary and instead focused on directly protecting communities.

➢ Invest in preparing communities to live safely and sustainably in a fire-prone environment: retrofit homes to reduce home ignitability, improve emergency communications, maintain safe evacuation routes, construct community fire shelters, bury powerlines, and implement other infrastructure projects that could be part of a Green New Deal.

Common Myths about Forests and Fire

by Chad Hanson, PhD, Ecologist and Director, John Muir Project

Do We Currently Have an Unnatural Excess of Fire in our Forests? No. There is a broad consensus among fire ecologists that we currently have far less fire in western US forests than we did historically, prior to fire suppression (Hanson et al. 2015). For example, currently, we have about 200,000 acres of fire in California's forests per year on average, and 500,000 to 900,000 in the very biggest years. Historically, before fire suppression, an average year would see 1-2 million acres in California's forests (Stephens et al. 2007, Baker 2017). California's forests have always burned with a mixture of intensities, including patches of high-intensity fire. We have less fire of all intensities now, including less high-intensity fire (Stephens et al. 2007; Mallek et al. 2013; Baker et al. 2018).

Do Current Fires Burn Mostly at High-Intensity Due to Past Fire Suppression? No. Current fire is mostly low/moderate-intensity in western US forests, including the largest fires (Mallek et al. 2013, Baker et al. 2018). The most long-unburned forests experience mostly low/moderate-intensity fire (Odion and Hanson 2008; Miller et al. 2012; van Wagtendonk et al. 2012).

Do Large High-Intensity Fire Patches Destroy Wildlife Habitat or Prevent Forest Regeneration? No. Hundreds of peer-reviewed scientific studies find that patches of high-intensity fire create "snag forest habitat", which is comparable to old-growth forest in terms of native biodiversity and wildlife abundance (DellaSala and Hanson 2015). In fact, more plant, animal, and insect species in the forest are associated with this habitat type than any other (Swanson et al. 2014). Forests naturally regenerate in ecologically beneficial ways in large high-intensity fire patches (DellaSala and Hanson 2015, Hanson 2018).

Is Climate Change a Factor in Recent Large Fires? Yes. Human-caused climate change increases temperatures, which influences wildland fire. Some mistakenly assume this means we must have too much fire but, due to fire suppression, we still have a substantial fire deficit in our forests. For example, historically, snag forest habitat, from high-intensity fire and patches of snag recruitment due to drought and native bark beetles, comprised 14% to 30% of the forests in the Sierra Nevada (Show and Kotok 1925; Safford 2013; Baker 2014; Baker et al. 2018). Currently, based on federal Forest Inventory and Analysis data, it comprises less than 8% of Sierra Nevada forests.

Are Our Forests Unnaturally Dense and "Overgrown", and Do Denser Forests Necessarily Burn More Intensely? No. We currently have somewhat more small trees than we had historically in California, but we have fewer medium/large trees, and less overall biomass—and therefore less carbon (McIntyre et al. 2015). Our forests actually have a carbon deficit, due to decades of logging. Historical forests were variable in density, with both open and very dense forests (Baker et al. 2018). Recent studies by U.S. Forest Service scientists, regarding historical tree density, omitted historical data on small tree density and density of non-conifer trees. When the missing historical data were included, it was revealed that historical tree density was 7 times higher than previously reported in ponderosa pine forests, and 17 times higher than previously reported in mixed-conifer forests (Baker et al. 2018). Wildland fire is driven mostly by weather, while forest density is a "poor predictor" (Zald and Dunn 2018).

Are Recent Large Fires Unprecedented? No. Fires similar in size to the Rim fire and Rough fire, or larger, occurred prior to modern fire suppression (Bekker and Taylor 2010, Caprio 2016).

Do Occasional Cycles of Drought and Native Bark Beetles Make Forests "Unhealthy"? Actually, it's the opposite. During droughts, native bark beetles selectively kill the weakest and least climate-adapted trees, leaving the stronger and more climate-resilient trees to survive and reproduce (Six et al. 2018). In areas with many new snags from drought and native bark beetles, most bird and small mammal species increase in numbers in such areas because snags provide such excellent wildlife habitat (Stone 1995).

Do Forests with More Dead Trees Burn More Intensely? Small-scale studies are mixed within 1-2 years after trees die, i.e., the "red phase" (Bond et al. 2009, Stephens et al. 2018), but the largest analysis, spanning the entire western U.S., found no effect (Hart et al. 2015). Later, after needles and twigs fall and quickly decay into soil, and after many snags have fallen, such areas have similar or lower fire intensity (Hart et al. 2015, Meigs et al. 2016).

Does Reducing Environmental Protections, and Increasing Logging, Curb Forest Fires? No, based on the largest analysis ever conducted, this approach increases fire intensity (Bradley et al. 2016). Logging reduces the cooling shade of the forest canopy, creating hotter and drier conditions, leaves behind kindling-like "slash" debris, and spreads combustible invasive weeds like cheatgrass.

Do "Thinning" Logging Operations Stop Wildland Fires? No. "Thinning" is used as a euphemism for intensive commercial logging projects that kill and remove many of the trees in a stand, often including mature and old-growth trees. With fewer trees, winds, and fire, can spread faster through the forest. In fact, extensive research shows that commercial logging, conducted under the guise of "thinning", often makes wildland fires spread faster, and in most cases also increases fire intensity, in terms of the percentage of trees killed (Cruz et al. 2008, 2014).

Did the Rim Fire Emit Carbon Equal to Over 2 Million Cars? No. Recent unpublished reports from the Forest Service, and the California Air Resources Board regarding wildfire carbon emissions are based on a flawed model (FOFEM) that has repeatedly been shown to exaggerate carbon emissions by nearly threefold (French et al. 2011). Further, the FOFEM model falsely assumes that no post-fire regrowth occurs to pull CO2 out of the atmosphere. Field studies of large fires find usually only about 11% of forest carbon is consumed, and only 3% of the carbon in trees (Campbell et al. 2007), and vigorous post-fire forest regrowth absorbs huge amounts of CO2 from the atmosphere, resulting in an overall net decrease in atmospheric carbon within a decade after fire (Meigs et al. 2009).[1]

Would Landscape-Scale Prescribed Burning Reduce Smoke? No, it's the opposite. Prescribed fires do not stop wildland fires when they occur (Stephens et al. 2009), though they can alter fire intensity. However, any short-term reduction in potential fire intensity following prescribed fire lasts only 10-20 years, so using prescribed fires ostensibly as a means to reduce the intensity of wildland fires would require burning a given area of forest every 10-20 years (Rhodes and Baker 2008). This would represent a tenfold increase, or more, over current rates of burning (Parks et al. 2015). High-intensity fire patches produce relatively lower particulate smoke emissions (due to high efficiency of flaming combustion in higher-intensity fire patches) while low-intensity prescribed fires produce high particulate smoke emissions, due to the inefficiency of smoldering combustion. Therefore, even though high-intensity fire patches consume about three times more biomass per acre than low-intensity fire (Campbell et al. 2007), low-intensity fires produce 3-4 times more particulate smoke than high-intensity fire, for an equal tonnage of biomass consumed (Ward and Hardy 1991, Reid et al. 2005). As a result, a landscape-level program of prescribed burning would cause at least a ten-fold increase in smoke emissions relative to current fire levels.

1. For example, Campbell et al. (2007) found that the Biscuit fire of 2002 emitted an average of 19 tons of carbon per hectare, and Campbell et al. (2016) found that decay of fire-killed trees in the Biscuit fire emitted an average of about 0.75 tons of carbon per hectare per year over the first 10 years post-fire (there were lower emissions from decay in subsequent decades). Therefore, for the first 10 years post-fire, the total carbon emissions from the Biscuit fire (carbon emissions from the fire itself, plus subsequent emissions from decay) were approximately 26 tons of carbon per hectare. Meigs et al. (2009) (Table 5) report that, by only five years after fire, regrowth was pulling 3.1 tons of carbon per hectare per year out of the atmosphere. Therefore, by 10 years post-fire, this equates to approximately 31 tons of carbon pulled out of the atmosphere by regrowth—i.e., an overall net increase in carbon of 5 tons per hectare relative to pre-fire levels.

Facts about California Forests, Wildfires, and Carbon

by Dominick A. DellaSala, PhD, Chief Scientist, Geos Institute

California's forests are nature's climate solutions, readily absorbing and storing massive amounts of carbon in trees, dense foliage, and productive soils over decades to centuries (Griscom et al. 2017). Protecting the carbon stored in forests from logging is key to a climate-safe future for California. However, recent policies proposed by the state are seeking to elevate logging levels while rolling back environmental protections in response to wildfires. These policies are sometimes portrayed as ways to sequester and store more carbon in forests and wood products. However, there is a better way to address pressing climate issues in California by using the best available science in forestry-climate policies as follows.

Do Forest Fires Emit Massive Amounts of Carbon Dioxide? At the forest stand level, most studies in the Pacific Northwest indicate that individual forest fires emit small amounts of emissions (Campbell et al. 2007; Meigs et al. 2009; Mitchell 2015). At the state level, total annual emissions from wildfires are much less (generally <10%) than total annual emissions from logging even during active fire seasons (Meigs et al. 2009; Campbell et al. 2012; Law et al. 2018; Oregon Global Warming Commission 2018). Trees killed by wildfires are not combusted (aside from twigs and leaves), and they decompose slowly over decades to centuries while logging releases carbon rapidly (the concept of carbon absorption being slow-in from forest growth over time and fast-out from rapid release by logging). About half the carbon produced in wildfires remains bound to the soils for nearly a century, while the other half is stored for millennia (Singh et al. 2012). After fires, growth of surviving trees and new vegetation sequester carbon, offsetting emissions within about 5-50 years (depending on site factors; Meigs et al. 2009, Mitchell 2015).

Does Logging Store or Release Carbon? Depending on logging intensity, forest type, and forest age class, up to 62% of carbon stored within a forest is released to the atmosphere as CO_2 pollution when forests are cut down due to decomposition (or burning) of logging slash, stumps, root wads, and soil carbon losses with additional emissions during transport and manufacturing of wood products, especially over large hauling distances (Oregon Global Warming Commission 2018, Law et al. 2018). The remaining 38% is temporarily embodied in wood product pools ranging from 1 year (paper) to decades (buildings) before decomposing and emitting CO_2 in landfills (Oregon Global Warming Commission 2018). This loss is not made up for by planting trees or substitution of wood for steel in buildings (Law et al. 2018). Thus, wood product pools have a much shorter carbon retention "life span" than the carbon stored in unlogged forests (Law et al. 2018). Based on recent studies in the Pacific Northwest, carbon stocks in forests can be doubled if forests are protected from logging on federal lands, timber harvest rotations extended from 35 to 70 years on private lands, and other forestry improvements (Law et al. 2018). Avoiding emissions from deforestation and forest degradation is also recommended by the Intergovernmental Panel on Climate Change as an effective means for preventing warming in excess of 1.5°C globally. According to NASA's Earth Observatory (2017), California already is pushing temperature increases dangerously close to unsafe levels.

Does “Thinning” Reduce Emissions from Wildfires? Studies of landscape-scale logging (“thinning”) to reduce the probability of crown fires show that this practice will not reduce carbon emissions under current or future climate scenarios and may in fact make matters much worse, especially if thinning residues are burned as biofuels (Meigs et al. 2009; Hudiburg et al. 2009, 2011; Campbell et al. 2012; Mitchell et al. 2012; Schulz et al. 2012; Law et al. 2013). This is because the amount of carbon removed by landscape-scale thinning and related activities to influence fire behavior is larger than that saved in a fire, and fire only occurs on a fraction of the areas thinned (Rhodes and Baker 2009, Campbell et al. 2012).

Conclusions

California’s forests have always benefited ecologically from periodic mixed-severity fires that create diverse wildlife habitat, stimulate plant growth and nutrient cycling, and carbon sequestration. Overall, they are not a major source of emissions currently as most of the carbon remains on site after disturbance and new vegetation offsets losses. Much bigger emissions are produced by logging and other industrial sectors. Thus, policies that advocate for increased logging are inconsistent with California’s otherwise ground-breaking climate change efforts, and the recommendations of the Intergovernmental Panel on Climate Change. Protecting forests from logging is a natural climate solution on par with global efforts to mitigate climate change impacts (Griscom et al. 2017). California has some of the most carbon dense forests on the planet and these forests should form the backbone of a comprehensive climate change strategy that includes avoiding and reducing emissions from all sectors while preparing for unavoidable consequences of rapidly advancing climate impacts.

Biomass Power is a False Solution

by Brian Nowicki, California Climate Policy Director, Center for Biological Diversity

Fire policies in California rely heavily on burning forest biomass for energy production paired with efforts to increase logging to alter forest fire behavior. Biomass power is often portrayed as being carbon neutral, but it is not. Instead, biomass facilities increase greenhouse gas emissions; undermine the transition to clean, renewable power; pose public health threats in already-disadvantaged communities; and distort policies for forest and fire management.

Biomass energy is more climate-polluting than coal.

Forest-sourced woody biomass energy generation emits about 50% more CO2 per megawatt-hour of electricity produced than coal-fired power and three times the CO2 of natural gas (Booth 2014). While the baseline emission rate for California's current electricity portfolio is about 500 lbs CO2 per MWh (CARB 2018), biomass can emit more than 3,000 lbs CO2 per MWh (Booth 2014), and smaller-scale facilities using gasification technology are similarly carbon-intensive (Ascent Environmental 2012).

Using forest biomass as a feedstock is a significant net negative impact to the climate.

In addition to smokestack emissions, an accurate accounting of the climate harms of biomass energy must include the carbon implications of the tree removals that generate the feedstock. Thinning operations tend to remove about three times as much carbon from the forest as would be avoided in wildfire emissions (Campbell et al. 2011), and the removal of live trees from the forest also results in a loss of future growth and carbon sequestration by those trees.

The climate damage of biomass can persist for decades to centuries.

Bioenergy converts stored carbon to CO2 instantaneously, while future resequestration or avoided decomposition may take years, decades, or even centuries to achieve atmospheric parity. Multiple studies have shown that it can several decades to discharge the "carbon debt" associated with bioenergy production, even where "waste" materials like timber harvest residuals are used for fuel (Manomet Center for Conservation Sciences 2010; Repo et al. 2010, McKechnie et al. 2011; Mitchell et al 2012; Schulze et al. 2012; Booth 2018). Where forests are harvested specifically for fuel, it can be decades to centuries, if ever, before the bioenergy system realizes a net carbon benefit (depending on harvest intensity, frequency, and forest characteristics) (Searchinger et al 2009; Hudiburg et al 2011; Campbell et al 2011; Mitchell et al. 2012). One study concluded that the resulting atmospheric emissions increase may even be permanent (Holtzmark 2012).

The Trump Administration and Congress have directed federal agencies to disregard the science and assume biomass is carbon neutral.

The 2018 federal omnibus appropriations bill included a provision that ignored the recommendations of federal agencies and a scientific advisory board, and simply directed agencies to issue regulations that "reflect the carbon-neutrality of forest bioenergy." Similarly, in April 2018, EPA administrator Scott Pruitt disregarded science-based rulemaking and simply directed his agency to pursue policies that promote biomass.

California state policy ignores the carbon impacts of biomass as a component of forest policy.
California's greenhouse gas cap-and-trade program does not count the emissions from biomass combustion when calculating the level of carbon pollution for which electricity companies must obtain or purchase credits for smokestack emissions. Other California law requires that electricity suppliers collectively purchase 250 MW of biomass power annually, and California's Forest Carbon Action Plan and Vegetation Treatment Plan both prioritize biomass energy as a driver for forest thinning projects that remove live trees from the forest. Each of these policies includes a de facto assumption that biomass energy is carbon neutral, without explicitly stating that finding or providing any determination of the carbon impacts of biomass.

Policies that subsidize forest biomass divert funds from zero-carbon sources like solar and wind and impede the transition to renewable energy.
Biomass energy can be five times as expensive as wind and solar, costing $199/MWh compared to $40/MWh for wind and solar (PG&E 2017). Yet California requires that electricity suppliers collectively purchase 250 MW of biomass power annually.

Biomass results in significant emissions of air pollutants, often in California's most polluted communities.
In addition to producing large amounts of CO2, biomass generation can result in significant emissions of air pollutants that harm human health, including nitrogen oxides, carbon monoxide, particulate matter, and black carbon (Booth 2014). Biomass burning also emits large amount of hazardous air pollutants, including hydrochloric acid, dioxins, benzene, formaldehyde, arsenic, chromium, cadmium, lead, and mercury. Biomass emissions can exceed those of coal-fired power plants even after application of best available control technology.

The five most polluting biomass facilities in the San Joaquin Valley are located in the top four percent most disadvantaged census tracts in the state. For example, the Rio Bravo biomass plant in Fresno—which is expected to receive trees logged after the Rim Fire near Yosemite National Park, in a project promoted by the Sierra Nevada Conservancy—is located less than a half-mile from the Malaga Community Park, Malaga Elementary School and surrounding homes, in a neighborhood with a pollution burden score of 100 (Gale 2017).

Conclusion: Forest biomass energy is an expensive and highly polluting electricity source that is a false solution for the climate and for forest management.

Forest Fire Policies are Being Misapplied to Chaparral Ecosystems

by Richard Halsey, Executive Director, California Chaparral Institute

Chaparral is California's most extensive plant community. It is found in every county in the state. Characterized by drought-hardy shrubs, a Mediterranean-type climate, and infrequent, high-intensity fire, chaparral provides the habitat richness responsible for making California one of the most biodiverse regions on earth (Halsey and Keeley 2016). The chaparral's relationship to fire is dramatically different from that of California's forests. Actions that are often proposed for addressing fire in forest ecosystems are not appropriate in chaparral ecosystems and can lead to more flammable landscapes, destruction of critical habitat, and are an ineffective approach to protecting human communities built in these areas.

High-Intensity Fire Required

The natural fire regime for chaparral is characterized by large, high-intensity crown fires with a return interval of 30 – 150 years (Keeley and Fotheringham 2001; Lombardo et al. 2009; Safford et al. 2014). Research has demonstrated that the higher the intensity of the fire, the better the chaparral is able to recover (Keeley et al. 2005). Therefore, concerns over reducing fire intensity and severity are irrelevant to chaparral ecosystems; there's no such thing as a low-intensity chaparral fire except at the edges of fire perimeters or when localized conditions (e.g. boulders, wind shifts, moisture) reduce fire intensity. By the very nature of the physical structure of shrubs, high intensity fire is an inherent part of chaparral fires.

Long Fire Return Intervals are Required, and Too Much Fire Causes Loss of Chaparral

When compared to most forests, chaparral has comparatively long intervals between fires (30 – 150 years or more). Long fire return intervals are vital for the chaparral's ecological health. It can take up to thirty years for the native shrubs to build up enough seed in the soil to provide adequate germination rates post fire.

However, increases in fire frequency due to human-caused ignitions and the effects of climate change cause chaparral stands to become more open and are often invaded by nonnative grasses. Fire-return intervals fewer than 10 years have been shown to be highly detrimental to the persistence of chaparral species (Haidinger and Keeley 1993, Jacobsen et al. 2004). As grasses increase, the flammability of the chaparral ecosystem also increases. As a consequence, a positive feedback loop is created whereby more grass encourages frequent ignitions. Such frequent fires not only eliminate the native shrubs, but they facilitate the further spread of invasive weeds and grasses due to the fact that grass fires are less intense than shrubland fires. The type conversion process can ultimately lead to the complete replacement of native chaparral with nonnative grasses (Halsey and Syphard 2015).

Prescribed Burns and Vegetation Clearing are Destructive to Chaparral and Increase Fire

When fire management policies commonly used in forests—such as prescribed fire and vegetation clearing—are misapplied to chaparral, the results are destructive to the ecosystem and can actually increase fire. Since there is too much fire in chaparral plant communities due to human-caused ignitions, adding more through prescribed burns only increases the threat to the chaparral ecosystem's continued existence and conversion to invasive grasses that bring more frequent fires. Furthermore, prescribed burns are typically conducted in the late spring when the ecosystem is the most vulnerable to damage: the plants are growing, the soil is still moist, and many animal species are breeding. Therefore, prescribed burns can cause significant damage to plant growth tissues and destroy seeds in the soil due to soil moisture turning into steam, leading to chaparral type conversion.

Similarly, large-scale vegetation clearing projects ("fuelbreaks") also cause the loss of native chaparral and the spread of invasive grasses that leads to more frequent fires. Amid the increasing dangers to chaparral from the effects of climate change, it is imperative that land management agencies do not exacerbate the loss of chaparral through activities like prescribed burns and large-scale habitat clearance projects away from homes. Instead, fire management in chaparral should focus on reducing the unnaturally high level of fire ignitions that has accompanied human development in this ecosystem (Keeley et al. 2005b, Keeley 2006, Syphard et al. 2007).

Focus on Homes and Their Immediate Surroundings to Make Fire-Safe Communities

While fire's role in chaparral is different from in forests, the most effective way to keep homes from igniting during wildfires is the same in chaparral areas as in forest areas—focus on fire-safety features for homes and the zone right around them, rather than large-scale vegetation alteration in wildlands.

In a comprehensive study of the 2007 Witch Creek Fire in San Diego County, researchers found, "Wind-blown embers, which can travel one mile or more, were the biggest threat to homes in the Witch Creek Wildfire. There were few, if any, reports of homes burned as a result of direct contact with flames" from wildland fuels (IBHS 2008).

In a study examining 700,000 addresses in the Santa Monica Mountains and part of San Diego County researchers mapped the structures that had burned in those areas between 2001 and 2010, a time of devastating wildfires in the region (Syphard et al. 2012). Buildings on steep slopes, in Santa Ana wind corridors and in low-density developments intermingled with wild lands were the most likely to have burned. Nearby vegetation was not a big factor in home destruction. Looking at vegetation growing within roughly half a mile of structures, the authors concluded that the exotic grasses that often sprout in areas cleared of native habitat like chaparral could be more of a fire hazard than the shrubs. "We ironically found that homes that were surrounded mostly by grass actually ended up burning more than homes with higher fuel volumes like shrubs," Syphard said.

Working only on defensible space is not sufficient. Many homes with adequate defensible space have still burned to the ground because embers have entered through attic vents, ignited flammable materials around the home (litter in the gutter, wood stacks, wood fencing), or found their way under roofing materials (Maranghides and Mell 2009). The solution is to reduce the flammability of the home as much as possible: install ember resistant vents, Class A roofing, exterior sprinklers operated by an independent system, and remove flammable materials 100 feet from around the structure.

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Large Trees Dominate Carbon Storage in Forests East of the Cascade Crest in the United States Pacific Northwest

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Large-diameter trees store disproportionally massive amounts of carbon and are a major driver of carbon cycle dynamics in forests worldwide. In the temperate forests of the western United States, proposed changes to Forest Plans would significantly weaken protections for a large portion of trees greater than 53 cm (21 inches) in diameter (herein referred to as "large-diameter trees") across 11.5 million acres (∼4.7 million ha) of National Forest lands. This study is among the first to report how carbon storage in large trees and forest ecosystems would be affected by a proposed policy. We examined the proportion of large-diameter trees on National Forest lands east of the Cascade Mountains crest in Oregon and Washington, their contribution to overall aboveground carbon (AGC) storage, and the potential reduction in carbon stocks resulting from widespread harvest. We analyzed forest inventory data collected on 3,335 plots and found that large trees play a major role in the accumulated carbon stock of these forests. Tree AGC (kg) increases sharply with tree diameter at breast height (DBH; cm) among five dominant tree species. Large trees accounted for 2.0 to 3.7% of all stems (DBH $\geq$ 1" or 2.54 cm) among five tree species; but held 33 to 46% of the total AGC stored by each species. Pooled across the five dominant species, large trees accounted for 3% of the 636,520 trees occurring on the inventory plots but stored 42% of the total AGC. A recently proposed large-scale vegetation management project that involved widespread harvest of large trees, mostly grand fir, would have removed ∼44% of the AGC stored in these large-diameter trees, and released a large amount of carbon dioxide to the atmosphere. Given the urgency of keeping additional carbon out of the atmosphere and continuing carbon accumulation from the atmosphere to protect the climate system, it would be prudent to continue protecting ecosystems with large trees for their carbon stores, and also for their co-benefits of habitat for biodiversity, resilience to drought and fire, and microclimate buffering under future climate extremes.

Keywords: carbon, climate mitigation, eastside screens, forests, global change, large-diameter trees, 21-inch rule

INTRODUCTION

Forest carbon accumulation is crucial for mitigating ongoing climatic change, with individual large trees storing a substantial portion of the overall carbon in living trees. Globally, forests store about 862 Gt carbon in live and dead vegetation and soil, with 42% of it stored in live biomass (above- and belowground; Pan et al., 2011). Globally, forests removed the equivalent of about 30% of fossil fuel emissions annually from 2009 to 2018 (Friedlingstein et al., 2019), and 44% of that was by temperate forests. Temperate forests of the United States are the largest category of land sinks in the country, consistently offsetting about 14% of the Nation's CO_2 emissions (EPA, 2020). Projections indicate that ecological systems have significant additional climate mitigation potential, with forest carbon accumulation serving as a central component of a natural climate solutions framework (Griscom et al., 2017; Fargione et al., 2018; Moomaw et al., 2019; Cook-Patton et al., 2020).

Large-diameter trees constitute about half of the mature forest biomass worldwide and are key to the ability of forests to accumulate substantial amounts of carbon needed to mitigate climate change (Luyssaert et al., 2008; Lutz et al., 2018). Trees exceeding 60 cm (23.6 in) diameter at breast height (DBH) comprise ∼41% of the world's aboveground live tree biomass (Lutz et al., 2018). Furthermore, on average, 50% of the live tree biomass carbon in all types of forests globally is stored in the largest 1% of trees, but the value for the United States is lower, ∼30% in the largest 1% of trees due to widespread historical logging of large trees (Lutz et al., 2018). A single large tree can add the same amount of carbon to the forest within a year as is contained in a single mid-sized tree of the same species (Stephenson et al., 2014). The relationship between large-diameter trees and overall forest biomass suggests that forests cannot accumulate aboveground carbon (AGC) to their ecological potential without large trees (Lutz et al., 2018). Recognition of the importance of large-diameter trees in determining global atmospheric carbon stocks has led to management recommendations to conserve existing large-diameter trees and those that will soon reach large diameters (Lindenmayer et al., 2014; Lutz et al., 2018; Moomaw et al., 2019).

In addition to comprising a substantial portion of forest carbon storage and accumulation, large-diameter trees fulfill a variety of unique ecological roles such as increasing drought-tolerance, reducing flooding from intense precipitation events, altering fire behavior, redistributing soil water, and acting as focal centers of mycorrhizal communication and resource sharing networks (Bull et al., 1997; Brooks et al., 2002; Brown et al., 2004; Luyssaert et al., 2008; Beiler et al., 2015; Lindenmayer and Laurance, 2017). In the United States Pacific Northwest (PNW), carbon dense old growth forests buffer against increasing temperatures by creating microclimates that shelter understory species from rising temperatures (Frey et al., 2016; Davis et al., 2019a). Forests with large-diameter trees often have high tree species richness, and a high proportion of critical habitat for endangered vertebrate species, indicating a strong potential to support biodiversity into the future and promote ecosystem resilience to climate change (Lindenmayer et al., 2014; Buotte et al., 2020).

In the PNW region of the United States a 21-inch (∼53 cm) diameter criteria (USDA Forest Service, 1995) was enacted in 1994 to slow the loss of large, older trees and old forest patches in national forests east of the Cascade Mountains crest in Oregon and Washington. The forests under consideration shall be referred to as "eastside forests" to be consistent with United States Forest Service (USFS) terminology. In our study, we refer to trees that equal or exceed this value to be "large-trees" or "large-diameter trees." Extensive studies determined that the large tree component of old forest structure had decreased due to human uses, and that sensitive wildlife species associated with old growth forest such as the American Marten and the Northern Goshawk were also in decline (Interior Columbia Basin Ecosystem Management Project, 2000; Wisdom et al., 2000; Bull et al., 2005; Greenwald et al., 2005). Site-specific exceptions have allowed removal of some trees ≥21 in DBH. However, the 21-inch rule has prevented large-scale harvest of trees ≥21 in DBH. For example, the 2010 Snow Basin Vegetation Management Project proposed harvesting over 43,000 trees ≥21 in DBH but was stopped by litigation [League of Wilderness Defenders, et al., v. Connaughton, et al., No. 3:12-cv-02271-HZ (D. Or. Dec. 9, 2014)]. Recently restoration of these forests was claimed to be inhibited by the 21-inch rule that protects large trees from logging (Hessburg et al., 2020). The rationale for harvesting large trees is premised upon the use of historical baselines of stand structure and species composition as management targets, and assuming that by removing large shade-tolerant species like grand fir and Douglas-fir it will promote resilience to future drought and disturbance (Johnston et al., 2018; Merschel et al., 2019; Hessburg et al., 2020). However, ongoing climate change and many other anthropogenic stressors such as habitat fragmentation, invasive species, and declines in biodiversity, heighten concerns over use of historical conditions as management targets (Millar et al., 2007; IPCC, 2018; Ripple et al., 2020). Proposed amendments to the 21-inch diameter limit would allow widespread harvesting of the larger trees up to 30 in DBH (76.2 cm) across six National Forests of eastern Oregon and southwestern Washington with major implications for forest carbon dynamics and important environmental co-benefits (**Figure 1**).

Carbon storage is an increasingly important management objective for National Forest Lands in the United States (Depro et al., 2008; Dilling et al., 2013; Dugan et al., 2017). USFS Forest Inventory and Analysis (FIA) data have been used to develop baseline stocks and trends of forest carbon for every region and individual National Forest, including assessment of the main disturbance, management, and environmental factors that drive changes (Birdsey et al., 2019). Western United States forests show considerable potential to accumulate additional carbon over the coming century, especially forests within the PNW that are projected to have relatively low to moderate vulnerability to future drought and fire (**Figure 1**; Buotte et al., 2020). Strategies to mitigate climate change effects on forests require careful examination of the tradeoffs of proposed forestry practices on forest carbon stock accumulation, water cycling, and additional environmental co-benefits of forests, such as biodiversity and

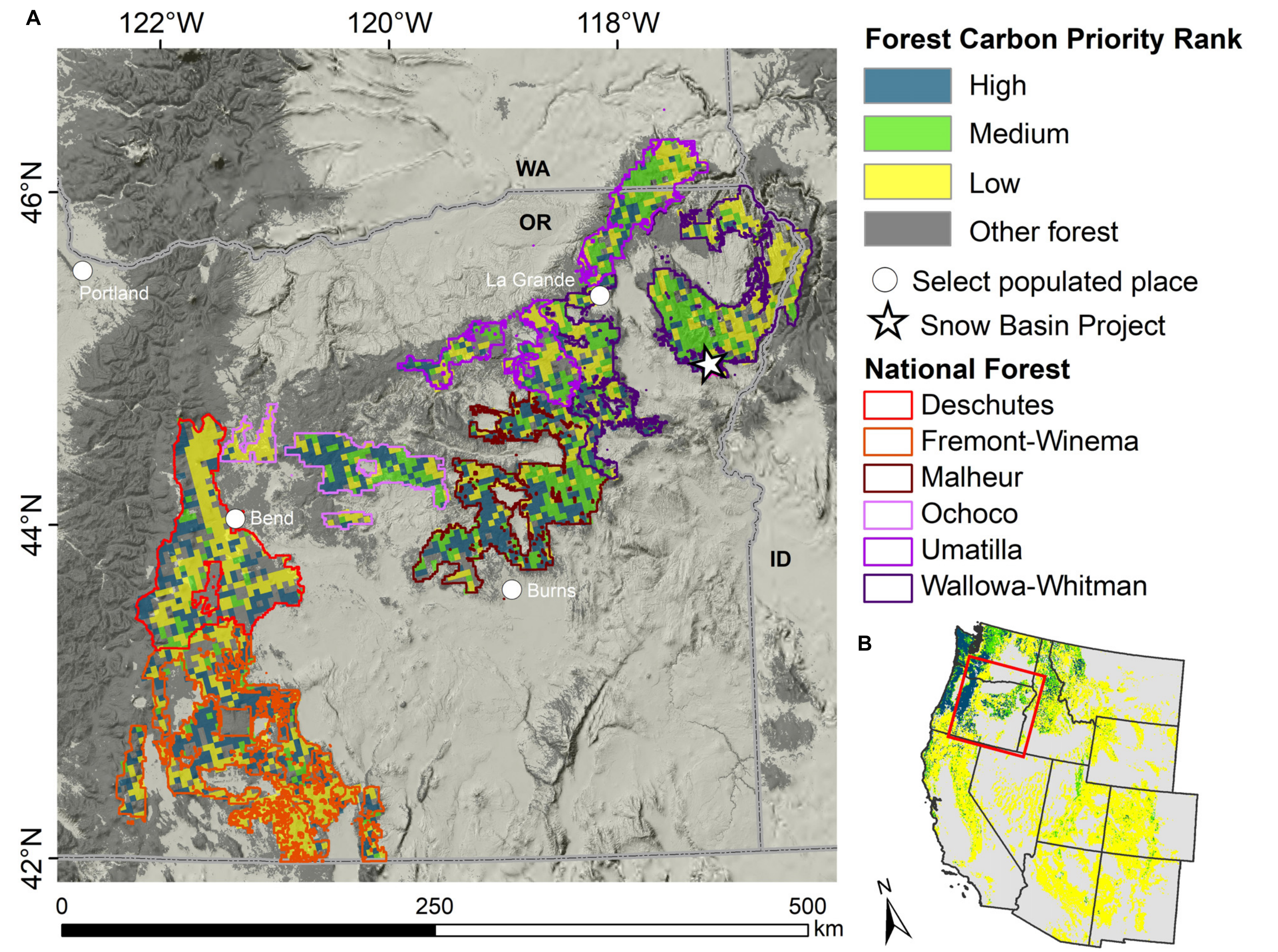

FIGURE 1 | Locations of the six national forests in eastern Oregon, southeastern Washington, and western Idaho, and the proposed Snow Basin Project area **(A)**. The forest carbon conservation priority rank is shown for the national forests and **(B)** for the broader western United States. The forest carbon conservation priority rank is from Buotte et al. (2020) and was derived from simulations of potential forest carbon sequestration and vulnerability to drought and fire over the 21st century. Forest extent is from Ruefenacht et al. (2008).

microclimatic buffering (McKinley et al., 2011; Law et al., 2018; Sheil, 2018; Buotte et al., 2020).

The potential effects of changing the 21-inch rule on carbon storage of eastside forests have not been adequately considered. Proposed changes to the management of large trees should be carefully assessed prior to adoption of new rules given that forest carbon storage is critically important in the context of a warming climate, and that large trees disproportionately store more accumulated carbon, keeping it out of the atmosphere. The objective of this study is to evaluate the potential impact of removal of the 21-inch rule on forest carbon stocks and carbon stock accumulation in eastside forests (**Figure 1**). We examine the relationship between tree DBH and AGC storage at tree to population scales for selected dominant tree species in eastside forests, focusing on the following questions:

1. How does AGC storage change with tree diameter among the dominant eastside tree species?
2. How common are large trees across the eastside forests by species and what proportion of each species total AGC do they account for?
3. What are the potential consequences on carbon stores of widespread removal of large trees in eastside forest restoration projects?

MATERIALS AND METHODS

Study Area

Our study area included the six national forests in eastern Oregon, two of which extend slightly into southwestern Washington and western Idaho (**Figure 1**). Located within the East Cascades and Blue Mountains ecoregions, these national forests together cover approximately 11.5 million acres (4.7 million ha) of complex mountainous terrain, characterized by a broad range of environmental gradients in climate regimes and

TABLE 1 | Summary of forest inventory data and climate conditions for the six National Forests included in the analysis.

National forest	Forest inventory			Climate conditions			
	N. Plots	DBH (cm)	AGC (kg)	Annual precipitation (mm)	Temperature (°C)		
					Annual	Jan.	July
Deschutes	474	16 (187)	131 (11359)	817 (170)	6.1 (0.6)	−1.5 (1.9)	16.1 (1.8)
Fremont-Winema	740	17 (204)	145 (14269)	681 (156)	6.3 (0.6)	−1.4 (1.9)	16.6 (1.8)
Malheur	687	15 (145)	108 (8392)	571 (104)	5.9 (0.7)	−2.9 (2.0)	17.2 (2.1)
Ochoco	253	16 (188)	119 (13462)	484 (101)	7.1 (0.6)	−1.0 (1.9)	17.6 (1.9)
Umatilla	487	14 (139)	99 (8759)	802 (121)	6.8 (0.7)	−1.5 (1.8)	17.5 (2.0)
Wallowa-Whitman	694	13 (164)	91 (8301)	792 (119)	6.0 (0.7)	−3.0 (1.8)	17.3 (2.1)

Forest inventory data were from the USFS Forest Inventory and Analysis program for the period 2000 to 2018. The variables include number of inventory plots and the average (max) diameter at breast height (DBH) and aboveground carbon (AGC) of trees on inventory plots. Average (±1 SD) climate conditions were based on the 40-year period from 1981 to 2019 and derived from 4-km resolution gridded PRISM data (Daly et al., 2008).

associated forest types (Johnson and Clausnitzer, 1992; Peterson and Waring, 1994; Berner and Law, 2015). The Blue Mountains ecoregion functions ecologically and floristically as a transverse bridge between the Cascade Mountains to the west and the Rocky Mountains to the east (Kerns et al., 2017). This "mega-corridor" links together some of the most intact habitat remaining in the PNW region and is of great importance to regional-scale connectivity and carbon storage in response to a warming climate (McGuire et al., 2016; Buotte et al., 2020).

Climatic conditions from 1981 to 2019 were similar across National Forests with regard to January and July average temperatures but show substantial variation in annual precipitation (**Table 1**). The Deschutes and Fremont-Winema National Forests are in the eastern Cascade Mountains and thus receive relatively high annual precipitation (817 mm and 681 mm) due to orographic uplift. National Forests in the southwestern Blue Mountains receive less precipitation (Malheur, 571 mm; Ochoco, 484 mm) due to their location within the rainshadow of the Cascade Mountains. In the northeastern portion of our study area annual precipitation increases to 802 mm and 792 mm, respectively, for the Umatilla and Wallowa Whitman National Forests due to the intrusion of moisture-laden air masses through the Columbia River Gorge. The region's complex precipitation patterns are similarly reflected in the distribution of forest types and biophysical characteristics.

Forest Inventory and Analysis Dataset

We relied on forest inventory measurements from the USFS FIA program and our own selection of biomass equations to evaluate relationships between tree diameter and AGC storage, as well as assess potential effects of tree removal proposed as part of the 2010 Snow Basin project. The FIA conducts systematic forest inventories across the United States, with one field sampling plot for every ∼2,400 hectares (6,000 acres) of forest (O'Connell et al., 2017). In the western United States, the FIA surveys 10% of sampling plots each year, meaning that each sampling plot is revisited every 10 years. At each sampling plot, field crews collect data on tree species, tree size, and other forest attributes. These inventory data are available online through the FIA DataMart[1].

[1]https://apps.fs.usda.gov/fia/datamart/

We downloaded the latest FIA data (FIADB_1.8.0.02) for Oregon, Washington, and Idaho, and then identified inventory plots that occurred within the six national forests based on a spatial overlay with national forest boundaries (n = 3,973 plots). To maintain landowner privacy the FIA plot coordinates available to the public are slightly altered ("fuzzed"), thus our spatial overlay may include some plots just outside the national forests or miss some plots that occur around the edges of these forests. We selected the latest inventory measurements from each surveyed plot for the years 2010 to 2018 (the most recent year available). We filtered these data to include only live trees of the five tree species of interest. A total of 54,651 individual trees were measured on the microplots, subplots, and macroplots, which represented 636,520 trees after applying the expansion factors for each type of plot. Overall, we used data from 3,335 plots in this analysis. A summary of FIA plot conditions within our study area shows that relatively small variations in average DBH across National Forests, between 13 cm and 17 cm, translate into substantial differences in AGC (**Table 1**). The Fremont Winema National Forest had the largest average DBH (∼17 cm) and the largest average AGC stores (∼145 kg), whereas the Wallowa-Whitman had the smallest average DBH (∼13 cm) and the smallest average AGC stores (∼91 kg).

We performed the analysis using the statistical software R (v. 3.6.1; R Core Team, 2020) with add-on packages including data.table (Dawle and Srinivasan, 2019), ggplot2 (Wickham, 2016), maptools (Bivand and Lewin-Koh, 2019), raster (Hijmans, 2019), and rgdal (Bivand et al., 2019).

Estimating Tree Aboveground Carbon Storage

We estimated AGC storage (AGC; kg C) for each tree on the FIA inventory plots using tree biometric measurements from FIA along with species- and component-specific biomass allometric models and traits (**Supplementary Tables 1, 2**). Using species-specific allometric models that include height measurements are essential to capture climate and site effects on taper and height (volume), and reduce uncertainty compared to estimates that are diameter-based and generalized models. For example, Van Tuyl et al. (2005) found that generalized models underestimated biomass by 50% on the eastside. Specifically, for each tree we estimated AGC as the sum of carbon stored in stem, branch, bark, and foliage biomass. To estimate stem biomass, we first derived stem volume using species-specific allometric models that incorporated tree diameter and height (Cochran, 1985; **Supplementary Table 1**). We then converted stem volume to biomass using published information on the wood density of each species, which we found was important to reducing uncertainty in estimates (Van Tuyl et al., 2005; Ross, 2010; Berner and Law, 2015). We also estimated branch, bark, and foliage biomass using species-specific allometric models, substituting equations for similar species where necessary (Gholz et al., 1979; Means et al., 1994; Jenkins et al., 2004). We assumed dry stem, branch, and bark biomass was 47.6 to 52.5% carbon depending on species (Lamlom and Savidge, 2003) and that dry foliage biomass was 46.1 to 51.4% carbon depending on species (Berner and Law, 2016). For trees smaller than 10 cm DBH, total aboveground biomass was estimated from tree height and then converted to AGC assuming wood carbon content.

Tree-Level Analysis of Relationship Between Tree Diameter and Aboveground Carbon

To better understand how carbon storage varies with tree size in eastside forests, we examined relationships between tree diameter (DBH; cm) and aboveground carbon (AGC; kg) among trees sampled on the FIA plots. Specifically, we rounded the DBH of each tree to the nearest centimeter and then computed the average AGC of trees within each 1-cm size class. To account for sampling uncertainty, we repeatedly computed these metrics using a resampling approach where each draw ($n = 10^4$) utilized data from a random 25% of inventory plots sampled with replacement. We computed the median across these 10^4 draws as our best-estimate of these metrics and also derived 95% confidence intervals.

Population-Level Analysis of Tree Diameter and Aboveground Carbon

We evaluated how the cumulative percentage of tree stems and AGC varied with tree diameter across the six national forests of our study area using forest inventories. This involved quantifying the cumulative percentage of tree stems and total AGC by DBH for each tree species, as well as pooled across tree species. To account for sampling uncertainty, we again implement a resampling approach ($n = 10^4$ draws).

Snow Basin Case Study: Carbon Consequences of Proposed Large Tree Removal

To estimate the carbon consequences of large tree removal we utilized USFS National Environmental Policy Act (NEPA) documentation from the Snow Basin Vegetation Management Project located on the Wallowa-Whitman National Forest, Whitman Ranger District, Baker County, Oregon (**Figure 1**). The project area includes 10,721 ha (26,493 acres) of National Forest land and primarily encompasses two main subwatersheds (Paddy Creek-Eagle Creek and Little Eagle Creek). Elevations within the project area range from approximately 1,340 m (4,400 feet) on the southern boundary to approximately 1,980 m (6,500 feet) at the northern boundary near the border with the Eagle Cap Wilderness. The USFS's preferred alternative plan was to remove over 43,000 large trees (DBH $\geq$ 21 in or 53.3 cm) from the project area. The large tree removal was prevented by litigation, but the project nonetheless provides a realistic framework to evaluate the carbon cost of large tree removal associated with dry forest landscape-scale restoration projects.

The Snow Basin NEPA document estimated the number of large trees to be removed or retained per acre by biophysical environment (Snow Basin DEIS). The USFS's preferred alternative proposed removal of grand fir from cool/moist and warm/dry grand fir biophysical environments, as well as removal of ponderosa pine, Douglas-fir, and western larch from mixed ponderosa pine and Douglas-fir biophysical environments (**Table 2**). Based on this information we computed the total number of large trees proposed for removal vs. retention across the project area by biophysical environment. To evaluate the potential carbon consequences of large tree removal, it was necessary to know tree size class distribution. The FIA only surveyed 13 plots in the Snow Basin project area between 2010 and 2018, of which 12 plots included large trees. After an initial analysis, we deemed this inadequate for rigorous assessment of tree size class distribution and therefore used tree measurements from all FIA plots in the Wallowa-Whitman National Forest that included large trees for the species of interest (n = 217 plots). To estimate the size class distribution of large trees, we computed the fraction of large trees that occurred at 0.1 cm DBH intervals between 53.3 cm and the largest observed DBH for grand fir, and after combining ponderosa pine, Douglas-fir, and western larch into a mixed-species group. For each biophysical environment, we then distributed the large trees proposed for removal or retention among these size classes in proportion to the occurrence of trees in each size class. Lastly, we estimated the total large tree AGC that would be removed or retained by multiplying the number of trees in each size class by the mean AGC of trees in that size class and then summing AGC across size classes. To account for sampling uncertainty, we again computed these metrics using a resampling approach where each of 10^4 draws utilized data from a random 25% of inventory plots sampled with replacement. As before, we computed the median across these 10^4 draws as our best-estimate and derived 95% confidence intervals. For verification, we also performed this assessment using only FIA plots in the Snow Basin Project area rather than across the Wallowa-Whitman National Forest and present these findings in the **Supplementary Material**.

RESULTS

Relationships Between Tree Diameter and Aboveground Carbon

Average tree aboveground carbon (AGC; kg) rapidly increased with tree diameter at breast height (DBH; cm) among dominant tree species measured on USFS inventory plots located in the

TABLE 2 | The total number of trees ≥21 in DBH proposed for removal and retention across the Snow Basin project area as reported in the NEPA plan prepared by the USFS (Table 53 of Snow Basin Vegetation Management Project DEIS, 2011).

Biophysical environment	Acres treated	Species removed	Mean number of large trees per acre		Total number of large trees	
			Removed	Retained	Removed	Retained
Cool/moist grand fir	973	Grand fir	5	9	4,865	8,757
Warm/dry grand fir	5,262	Grand fir	5	5	26,310	26,310
Warm/dry Douglas-fir and ponderosa pine Warm/moist Douglas-fir	6,136	Douglas-fir, ponderosa pine, western larch	2	3	12,272	18,408
Grand total	12,371	–	–	–	43,447	53,475

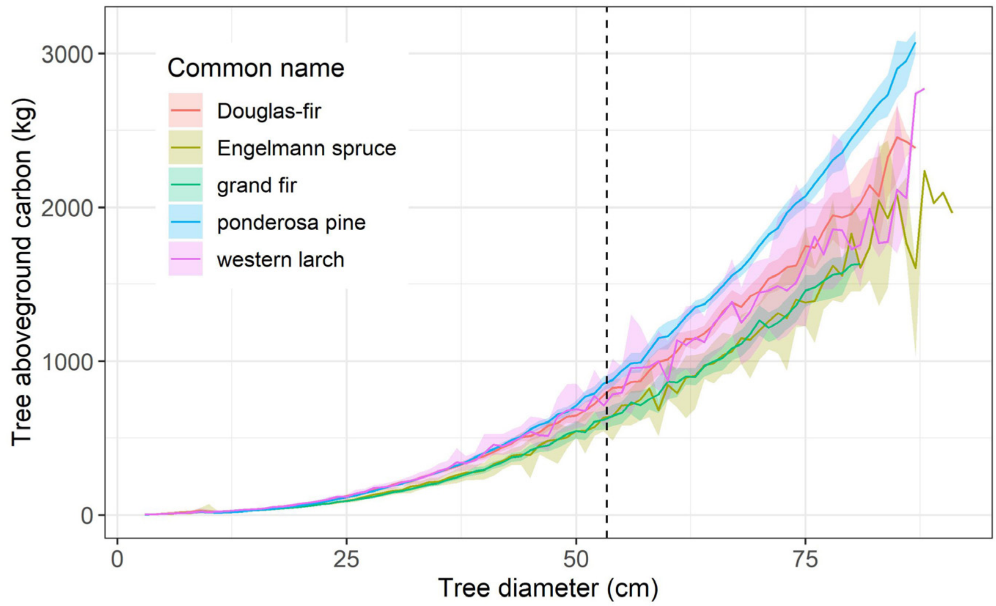

FIGURE 2 | Average tree aboveground carbon (AGC; kg) increases sharply with tree diameter at breast height (DBH; cm) for five dominant tree species measured on USFS inventory plots located in the six eastside national forests. The 21-inch (53.3 cm) DBH limit of the Eastside Screens (vertical dash line) is shown for reference. Tree AGC (≥10 cm DBH) was estimated from tree diameter and height using regional species-specific allometric equations, wood density, and carbon content. Similarly, sapling AGC (DBH = 3–9.9 cm) was estimated from sapling height using regional species-specific allometry and wood carbon content. These estimates and 95% confidence intervals were derived by resampling ($n = 10^4$) measurements from inventory plots (n = 3,335). For visual clarity, the tree diameter limit for each species excludes the largest 0.5% of trees, generally those larger than 80 to 88 cm DBH.

six eastside national forests (**Figure 2**). For instance, an average 25 cm (∼10”) diameter tree stored 90–121 kg of AGC depending on tree species, while a 50 cm (∼20”) diameter tree stored 541–683 kg of AGC. Thus, doubling tree diameter over this range led to a 5.3–6.2-fold increase in AGC. Similarly, tripling tree diameter from 25 cm to 75 cm led to a 13.8–18.2-fold increase in AGC, with the largest increase observed for ponderosa pine. For any given diameter, ponderosa pine, Douglas-fir and western larch tended to store more AGC than an Engelmann spruce or grand fir.

These results clearly showed that for large trees, a small increase in diameter corresponds to a massive increase in additional carbon storage relative to a small tree increasing by the same diameter increment. Overall, as trees grow larger, each additional centimeter of stem diameter corresponds with a progressively larger increase in tree carbon storage.

Tree Diameter and Aboveground Carbon Storage Within Tree Populations

Large trees (DBH ≥ 21 in or 53.3 cm) accounted for a small percentage of each species' tree stems, but a substantial percentage of the total AGC stored by each species on FIA inventory plots located in the six eastside national forests (**Figure 3** and **Table 3**). Specifically, large trees accounted for ∼2.0 to ∼3.7% of all stems (DBH ≥ 1” or 2.54 cm) among the five dominant tree species; however, these trees held ∼33.3 to ∼45.8% of the total AGC stored by each species (**Table 3**). Grand

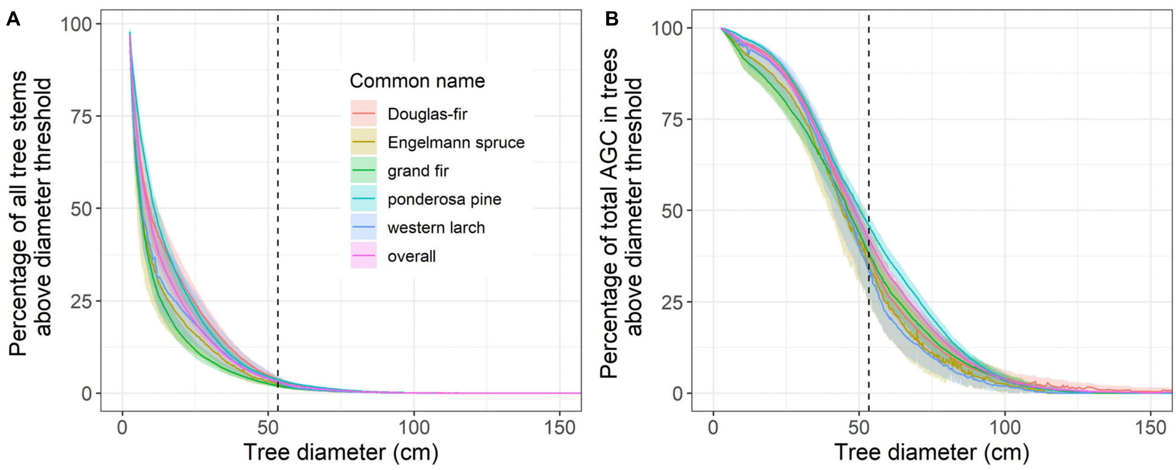

FIGURE 3 | Percentage of **(A)** all tree stems and **(B)** total aboveground carbon that occur in trees above a given diameter threshold by species based on measurements from USFS inventory plots located in the six eastside national forests. The dashed vertical lines represent a 21 in (53.3 cm) tree DBH. These estimates and 95% confidence intervals were derived by resampling ($n = 10^4$) measurements from inventory plots (n = 3,335). For example, trees with ≥21 in diameter represent 3.7 [3.2, 4.2]% of all ponderosa pine and account for 45.8 [42.8, 48.8]% of all aboveground carbon stored by this species on these inventory plots. Only ∼0.002% of trees exceeded 150 cm diameter, so for visual clarity we limited the x-axis to 150 cm.

TABLE 3 | Percentage of all trees stems and total aboveground carbon (AGC) occurring in trees above and below the 21 in DBH threshold based on measurements from USFS inventory plots located in the six eastside national forests.

Common name	Number of trees in the population	% of total species stems in trees. . .		% of total species AGC in trees. . .	
		<21 in	≥21 in	<21 in	≥21 in
Douglas-fir	100021	96.3 [95.4, 97.1]	3.7 [2.9, 4.6]	62.5 [58.2, 66.7]	37.5 [33.3, 41.8]
Engelmann spruce	31375	97.6 [96.0, 98.6]	2.4 [1.4, 4.0]	65.3 [54.6, 75.5]	34.7 [24.5, 45.4]
Grand fir	187445	98.0 [97.5, 98.5]	2.0 [1.5, 2.5]	61.6 [57.0, 66.2]	38.4 [33.8, 43.0]
Ponderosa pine	286970	96.3 [95.8, 96.8]	3.7 [3.2, 4.2]	54.2 [51.2, 57.2]	45.8 [42.8, 48.8]
Western larch	30708	97.2 [95.5, 98.4]	2.8 [1.6, 4.5]	66.7 [58.4, 74.8]	33.3 [25.2, 41.6]
Overall	636520	96.9 [96.6, 97.3]	3.1 [2.7, 3.4]	57.8 [55.7, 60.0]	42.2 [40.0, 44.3]

Live trees were measured on 3,335 plots surveyed between 2010 and 2018. The estimates and 95% confidence intervals below were derived by resampling bootstrap sampling. Each bootstrap sample ($n = 10^4$) utilized measurements from 25% of inventory plots selected at random.

fir had the lowest percentage of stems ≥21 in DBH (∼2.0%), while ponderosa pine and Douglas-fir had the highest percentage of stems exceeding this threshold (both ∼3.7%). However, large grand fir trees accounted for 38.4% of total species AGC, whereas large Douglas-fir accounted for 37.5% of total species AGC. Western larch had the lowest percentage of total species AGC in trees ≥21 in (∼33.3%), whereas ponderosa pine had the highest percentage (∼45.8%).

Pooling across the five dominant species, large trees accounted for ∼3.1% of the 636,520 trees occurring on the inventory plots, but stored ∼42.2% of the total AGC. Similarly, based on the FIA's CRM method, the five tree species together accounted for 44.9% of total AGC, with species-specific AGC stores ranging from ∼36.5 to ∼49.3% of the total stored by each species (**Supplementary Table 3** and **Supplementary Figure 1**).

A similar examination for trees >30 in DBH (76.2 cm) showed that among the five species, trees larger than 30 in DBH accounted for 0.4–0.9% of stems, but held ∼9.0 to ∼19.0% of each species AGC (**Table 4**). Ponderosa pine had the highest percentage of stems >30 in DBH (∼0.9%) and accounted for the highest percentage of AGC (∼19.4%) among the five species. Douglas-fir had ∼0.6% of stems >30 in DBH, and these stems held ∼12.9% of total species AGC. Engelmann Spruce, grand fir, and western larch each had ∼0.4% stems >30 in DBH, and accounted for ∼10.3, ∼14.0, and ∼9.2% of each species total AGC, respectively. Overall, trees >30 in DBH represent 0.6% of stems on inventory plots, but stored 16.6% of the total AGC.

Snow Basin Case Study: Carbon Consequences of Proposed Large Tree Removal

As per the USFS NEPA documentation, the Snow Basin project preferred alternative would have removed ∼43,447 large trees (DBH ≥ 21 in or 53.3 cm) across 12,371 acres, while retaining ∼53,475 large trees over this area (**Table 2**). We estimate this would translate into removing ∼131.3 Gg AGC stored in large trees, while retaining ∼164.1 Gg AGC in large trees (1 Gg = 10^9 g; **Table 5**). Grand fir from cool-moist and warm-dry environments would together comprise ∼67% (∼87.3 Gg C) of large tree AGC

TABLE 4 | Percentage of all tree stems and total aboveground carbon (AGC) occurring in trees above and below the 30 in diameter threshold based on measurements from USFS inventory plots located in the six eastside national forests.

Common name	Number of trees in the population	% of total species stems in trees...		% of total species AGC in trees...	
		<30 in	>30 in	<30 in	>30 in
Douglas-fir	100021	99.4 [99.2, 99.5]	0.6 [0.5, 0.8]	87.1 [83.7, 90.2]	12.9 [9.8, 16.3]
Engelmann spruce	31375	99.6 [99.2, 99.9]	0.4 [0.1, 0.8]	89.7 [82.7, 95.7]	10.3 [4.3, 17.3]
Grand fir	187445	99.6 [99.5, 99.7]	0.4 [0.3, 0.5]	86.0 [82.3, 89.3]	14.0 [10.7, 17.7]
Ponderosa pine	286970	99.1 [99.0, 99.3]	0.9 [0.7, 1.0]	80.6 [78.3, 82.7]	19.4 [17.3, 21.7]
Western larch	30708	99.6 [99.3, 99.8]	0.4 [0.2, 0.7]	90.8 [86.2, 94.8]	9.2 [5.2, 13.8]
Overall	636520	99.4 [99.3, 99.4]	0.6 [0.6, 0.7]	83.4 [81.8, 85.0]	16.6 [15.0, 18.2]

Tree AGC was derived using the FIA's Component Ratio Method. Live trees were measured on 3,335 plots surveyed between 2010 and 2018. The estimates and 95% confidence intervals below were derived by resampling bootstrap sampling. Each bootstrap sample (n = 10^4) utilized measurements from 25% of inventory plots selected at random.

TABLE 5 | Removal and retention of aboveground carbon (AGC) in large trees (DBH ≥ 21 in) within the proposed Snow Basin project area.

Common name	Env.	Total AGC in large trees (Gg C)...		% of large tree AGC...	
		Removed	Retained	Removed	Retained
Grand fir	Cool moist	13.6 [12.0, 15.3]	24.5 [21.6, 27.5]	35.7 [35.7, 35.7]	64.3 [64.3, 64.3]
Grand fir	Warm dry	73.7 [65.0, 82.5]	73.7 [65.0, 82.5]	50.0 [50.0, 50.0]	50.0 [50.0, 50.0]
Douglas-fir + ponderosa pine + western larch	Warm dry or moist	43.8 [40.5, 47.3]	65.7 [60.8, 71.0]	40.0 [40.0, 40.0]	60.0 [60.0, 60.0]
Overall	–	131.3 [120.8, 141.5]	164.1 [151.8, 176.2]	44.4 [44.2, 44.7]	55.6 [55.3, 55.8]

Estimates of total AGC are given in gigagrams of carbon (1 Gg = 10^9 g). Values in brackets denote 95% confidence intervals derived by resampling (n = 10^4) inventory plots used to determine tree size class distributions.

removed by the project, while ponderosa pine, Douglas-fir, and western larch from warm-dry and warm-moist environments would comprise the remaining ∼33% (∼43.8 Gg C). The project would have the largest relative and absolute impact on grand fir occurring in warm-dry environments, where 50% (∼73.7 Gg C) of large tree AGC would be removed. We obtained similar results albeit with substantially wider confidence intervals using only FIA inventory plots occurring in the Snow Basin project area (**Supplementary Table 4**).

DISCUSSION

Our study reveals the large carbon stocks associated with large-diameter trees in the region, and the potential for significant losses in AGC with large tree logging. The 21-inch rule was initially conceived to protect remaining late successional and old-growth forest and the native species that depend on these unique ecosystems for survival in forests east of the Cascades Crest in Oregon and Washington (Henjum et al., 1994). The carbon storage associated with the 21-inch rule on the six national forests is a significant co-benefit of this protective measure. Large trees (DBH ≥ 21 in or 53.3 cm) constitute ∼3% of the total stems, but store ∼42% (∼45% with CRM; **Supplementary Table 3**) of the AGC across the six eastside forests (**Table 3**). This finding highlights the important role of large trees in storing carbon in eastside forest ecosystems, and is consistent with previous findings on the disproportionately important role of large trees in the forest carbon cycle (Hudiburg et al., 2009; Lutz et al., 2012, 2018; Stephenson et al., 2014). The sharp increase in carbon storage with increasing tree diameter (**Figure 2**) speaks to the importance of preserving mature and old large trees to keep this carbon stored in the forest ecosystem where it remains for centuries (Law et al., 2018; Lutz et al., 2018). Once trees attain large stature, each additional DBH increment results in a significant addition to the tree's total carbon stores, whereas small-diameter trees must effectively ramp up to size before the relationship between DBH and AGC results in significant carbon gains. Harvest of large-diameter trees, even focused on a specific species (e.g., grand fir) can remove upward of 50% of the large tree AGC from these ecosystems (**Table 5**).

There are substantial differences in wood density among the species examined that contributes to the observed differences in AGC at a given DBH. Wood density tends to be higher for ponderosa pine (∼0.45 g cm^3), Douglas-fir (∼0.52 g cm^3), and western larch (∼0.56 g cm^3) than for Engelmann spruce (∼0.37 g cm^3) or grand fir (∼0.40 g cm^3; **Supplementary Table 2**). An evaluation of the relationship between tree height and diameter suggests the observed differences in AGC are not driven by ponderosa pine, Douglas-fir and western larch tending to be taller at a given DBH than Engelmann spruce or grand fir. Although stem decay is common in mature and old grand fir, a synthesis shows no evidence of carbon consequences of heart rot in grand fir (Harmon et al., 2008). While we could not estimate heart rot loss in grand fir due to lack of sufficient data, heart rot respiration has been estimated for another species and it had a scant contribution to ecosystem respiration (Harmon et al., 2004).

Forestry practices exert significant controls on stand structure and forest carbon dynamics, and alterations of harvest practices can substantially alter carbon storage and accumulation (Masek et al., 2011; Turner et al., 2011; Krankina et al., 2012; Kauppi et al., 2015; Law et al., 2018). Generally, there is a negative relationship between harvest intensity and forest carbon stocks whereby as harvest intensity increases, forest carbon stocks decrease while emissions increase (Hudiburg et al., 2009; Mitchell et al., 2009; Simard et al., 2020). It can take centuries to reaccumulate forest carbon stocks reduced by harvest (Birdsey et al., 2006; McKinley et al., 2011). It would take 180 to 310 years to reach maximum biomass in the Blue Mountains and East Cascades, respectively, after harvest of the large trees (Hudiburg et al., 2009), too long to help reach climate mitigation targets in the next few decades. The amount of harvested carbon that remains stored in wood products is insufficient to offset the loss of carbon stored in the forest. If harvested, life cycle assessment shows that 65% of the wood harvested in Oregon over the past 115 years has been emitted to the atmosphere, 16% is in landfills and only 19% remains in wood products (Hudiburg et al., 2019). Thus, harvesting the large trees will increase, not decrease emissions and end centuries of long-term carbon storage in the forests.

The 21-inch rule has preserved the trees that store and accumulate a disproportionately large amount of carbon in these forests (**Figure 3** and **Table 3**). Of the ~3% of tree stems over 21 in DBH in the study area, about 81% of these are between 21 in and 30 in DBH and account for ~61% of the AGC in all large trees (**Supplementary Figure 2**). Trees over 30 inches DBH account for ~19% of the large tree stems and hold ~39% of AGC in large trees. These findings are similar to those reported by Stephenson et al. (2014) and emphasize the relative importance of the sub-30 in DBH large trees in the study area, and the value in allowing these trees to continue growing and replenish the stock of trees over 30 inches DBH that are rare on the landscape. This proforestation strategy is among the most rapid means for accumulating additional quantities of carbon in forests and out of the atmosphere (Moomaw et al., 2019).

The importance of forest carbon storage is now greatly amplified by a warming climate that must urgently be addressed with reductions in greenhouse gasses and natural climate solutions (IPCC, 2018; Ripple et al., 2020). The preponderance of forests in our study area have medium to high carbon accumulation potential and low future climatic vulnerability (**Figure 1**), which reinforces the value of protecting large trees to help abate our current trajectory toward massive global change (Fargione et al., 2018; Buotte et al., 2020). Rather than holding ecosystems to an idealized conception of the past using historical conditions as management targets, a good understanding of the environmental co-benefits associated with large tree protection is needed to inform management strategies that contribute toward solving humanity's most pressing Earth system challenges (Millar et al., 2007; Rockström et al., 2009; Barnosky et al., 2017; Ripple et al., 2020). Replacing large diameter trees with seedlings will create a major carbon loss to the atmosphere during harvest (Harris et al., 2016) and not achieve storage of comparable atmospheric carbon for the indefinite future.

Large grand fir trees (DBH $\geq$ 21 in or 53.3 cm) accounted for the lowest proportion of tree stems for the five species we evaluated (2%), possibly reflecting the ingrowth of young, small grand fir within portions of our study area (Merschel et al., 2014; Johnston et al., 2016). One of the main premises for the proposed removals of large-diameter grand fir trees is that they have increased over historic levels, especially in drier biophysical environments. However, our data does not indicate an uncharacteristic abundance of large grand fir trees across these forests. Given the recent history of high-grade logging that focused on large and old trees (Henjum et al., 1994; Rainville et al., 2008), historical abundances of large trees were much greater than today (Wales et al., 2007; Hagmann et al., 2013; Kauppi et al., 2015), and thus would have represented a larger fraction of aboveground biomass than currently found on these forests.

Interestingly, with respect to the overall representation of species abundance based on inventory plots (**Table 3**), the population of grand fir with 187,445 stems is a distant second to ponderosa pine with 286,970 stems, followed by Douglas-fir (100,021 stems), Engelmann spruce (31,375 stems), and western larch (30,708 stems). To the extent that current forest stand structure is skewed toward smaller diameter classes, ponderosa pine, grand fir, and Douglas-fir trees are notable for the number of trees on inventory plots.

It is important to note that a diameter limit that emphasized protection of carbon stores would ideally protect trees starting at a lower DBH limit (~12–15 inches). The 21-inch rule has provided significant leeway for fuels reduction and ecological restoration toward historical baselines to proceed in eastside forests. Ecological restoration treatments generally recommend giving protection to large and old trees, while reducing surface and ladder fuels, and accompanied by understory thinning treatments where appropriate and reintroduction of low-intensity fire at intervals (Allen et al., 2002; Brown et al., 2004; Agee and Skinner, 2005; Noss et al., 2006). From a forest carbon perspective, it would be prudent to restore an appropriate number of the approximately 97% of tree stems <21 inches DBH to replenish the stock of large trees over time (Lutz et al., 2018; Moomaw et al., 2019), while continuing to protect large trees for their carbon stores and the myriad other benefits that they provide. This approach would achieve the benefits of proforestation in the larger, most fire-resistant trees and reduction of fuel loads and stem density in the smaller diameter tree stems.

The Snow Basin project provides a case study for how amending the 21-inch rule to allow logging large trees could change management across the six eastside National Forests. Overall the Snow Basin project would have removed 44% or 131,000 metric tons of carbon stored in large trees. This is equivalent to ~0.75% of annual Oregon statewide carbon emissions, and this does not include carbon in tree roots or the AGC losses due to the removal of trees <21 in DBH. Removing nearly half of the carbon content from the large trees over 12,000 acres would create a carbon deficit in the live, dead, and soil carbon pools that will persist for many decades to the end of the century. However, left standing, these 43,445 large trees continue to grow, sequestering more carbon into long-term stores (Stephenson et al., 2014; Law et al., 2018; Domke et al., 2020). Older trees (∼100 years) are the next generation of old growth and already possess qualities associated with large, old trees, such as large canopies, deep root systems, and thick, fire-resistant bark.

Co-Benefits of Carbon, Habitat, Biodiversity, Water Availability, Resilience to Climate Extremes

High carbon conservation-priority forests support important components of biodiversity and are associated with increased water availability (McKinley et al., 2011; Perry and Jones, 2016; Berner et al., 2017; Law et al., 2018; Buotte et al., 2020). Large-diameter snags persist as standing snags for many years, providing valuable wildlife habitat, and account for a relatively high proportion of total snag biomass in temperate forests (Lutz et al., 2012). In PNW forests, large hollow trees, both alive and dead, are the most valuable for denning, shelter, roosting, and hunting by a wide range of animals (Bull et al., 2000; Rose et al., 2001). In the Interior Columbia River Basin, grand fir and western larch form the best hollow trees for wildlife uses (Rose et al., 2001). Downed hollow logs continue to serve as important hiding, denning, and foraging habitat on the forest floor (Bull et al., 1997; Bull et al., 2000). Large decaying wood influences basic ecosystem processes such as soil development and productivity, nutrient immobilization and mineralization, and nitrogen fixation (Harmon et al., 1986). Continuing to protect large trees in eastside forests provides the greatest benefit for carbon, habitat, and biodiversity.

Water availability and microclimatic buffering are also disproportionately affected by large trees and intact forests (Frey et al., 2016; Buotte et al., 2020). Forest canopies of the PNW buffer extremes of maximum temperature and vapor pressure deficit, with biologically beneficial consequences (Davis et al., 2019a). Removal of large trees quickly leads to a large increase in soil and canopy heating, which increases enough to impact photosynthesis (Kim et al., 2016), seedling survival, and regeneration (Kolb and Robberecht, 1996; Davis et al., 2019b). The climatic changes toward warmer and drier conditions expected in the next decades will likely increase forest stress and mortality (Allen et al., 2015). Eastside forests experienced hotter and drier conditions from 2003 to 2012 concentrated in the months of August and September, especially in drier forest type groups (i.e., ponderosa pine, juniper), whereas spring months (April–June) showed trends toward cooler and wetter conditions (Mildrexler et al., 2016). Projections suggest that proportionally, the largest changes in microclimatic buffering capacity will occur in lower elevation or dry forests, which currently have more limited buffering capacity (Davis et al., 2019a). In these drier regions, microclimatic buffering by forest canopies may create important microsites and refugia in a moisture-limited system (Meigs and Krawchuk, 2018). In an old growth ponderosa pine stand in eastern Oregon, ∼35% of the total daily water used from the upper 2 m was replaced by hydraulic redistribution from deep soil by deep-rooted larger trees in summer (Brooks et al., 2002). The bigger trees rarely reach 80% loss of hydraulic conductivity, and both mature pine and mesic Douglas-fir were better buffered from the effects of drought on photosynthesis compared with young pine (∼20-year old) due to full root development and larger stem capacitance in older trees (Kwon et al., 2018). Redistribution of deep soil water can increase seedling survival during summer drought when young trees lack the root development to reach deep soil water (Brooks et al., 2002). While large tree composition may have shifted today relative to European settlement times, these large trees nonetheless continue to perform important functional attributes related to water and climate such as carbon storage, hydraulic redistribution, shielding the understory from direct solar radiation, and providing wildlife habitat. These functional attributes of large trees, irrespective of species, characterize ecosystems through thousands to millions of years (Barnosky et al., 2017), and are not quickly replaced.

In mesic forest environments, microclimatic buffering and transpirational cooling are amplified because sites with higher moisture availability are better able to shift energy to latent as opposed to sensible heat fluxes (Dai et al., 1999; Mildrexler et al., 2011). During midday in full sun the surface temperature of a closed canopy moist forest is warmer than the temperature beneath the canopy which is protected from direct solar radiation (Thomas, 2011). Microclimates in moist forests are strongly linked to their closed-canopy structure (Chen et al., 1999; Aussenac, 2000). Removal of the overstory creates canopy openings that increase solar radiation penetration resulting in increased drying of the understory vegetation and the forest floor, and a thermal response of rising land surface temperatures (Chen et al., 1993, 1999). This alteration in the subcanopy thermal regime changes atmospheric mixing between the ground, subcanopy, and canopy, which in turn modifies the microclimate condition of the affected stand. Microclimate modifications associated with forest harvesting are expected to be greatest in moist forests and may affect resilience to climate change and increase the risk of occurrence and severity of wildfires (Lindenmayer et al., 2009). Maintaining mesic microclimates may give undisturbed moist forests and the species they support some inherent resilience to climate change. Moreover, an evaluation of the effects of water limitations on forest carbon cycling in the eastern Cascade Mountains found that grand fir radial growth was not strongly associated with variability in temperature or water variability (Berner and Law, 2015). A lengthening of the growing season may increase productivity in high-elevation grand fir stands. The microclimatic buffering, current and future potential carbon stores, and intact nature of previously unlogged grand fir and Douglas-fir forest types are co-benefits of protecting large trees that need to be considered in future management decisions.

Potential Solutions

The consequences of reducing protection of large trees are significant reduction in forest carbon stores and their climate mitigation, impacts on habitat for animals including birds, and resilience to a changing climate for decades to centuries to come. Given the rarity of large trees across the landscape, and their outsized role in *storing carbon removed from the atmosphere*, our findings call into question the value of removing large trees for forest modification in eastside forests.

If the 21-inch rule were retained on these lands, continued protection of the existing carbon stock would prevent large quantities of harvest related biogenic carbon from being released to the atmosphere. It is also essential to let a sufficient number of sub-21 inch trees remain to become additional large, effective carbon stores, and assure that carbon accumulation continues in these forests. Rather than weakening the 21-inch rule, we suggest strengthening this important measure and expanding large tree protections to other western United States public lands that have been adversely affected by a similar history of large-tree logging. Protecting and growing more large trees is the most effective near-term option for accumulating more carbon out of the atmosphere, and will benefit other ecosystem services as well.

Some public lands (local, state, and federal) could become part of a *designated reserve system* that includes intact forest landscapes, and carbon rich forests, that hold most of these large, older trees. This is an appropriate use of public lands because the services they provide including biodiversity, water retention, carbon accumulation and storage, and regional cooling by evapotranspiration serve the public interest, and promote sustainable economies that benefit from land protection and restoration.

The critical need to adapt to more wildfire in the west is congruent with protecting large-trees in fire-prone forests. Older forests experience lower fire-severity compared with younger, intensively managed forests, even during extreme weather conditions (Zald and Dunn, 2018). A shift in policy and management from restoring ecosystems based on historical baselines to adapting to changing fire regimes and from unsustainable defense of the wildland–urban interface to developing fire-adapted communities is needed (Schoennagel et al., 2017). Improved fire and forest management is part of the solution, but the most effective changes in terms of protection of people and property, will be near homes and on private property. Prioritizing federal fuel treatments around communities and creating better mechanisms for reducing fuels on private land can help reduce home loss and better protect communities (Moritz et al., 2014; Schoennagel et al., 2017). Given the natural role of fire in the West, managing more wild and prescribed fires with a range of severities will help reduce future wildfire threats and increase ecological benefits in many systems (Schoennagel et al., 2017).

Eastside forests are surrounded predominantly by rural communities. For production forests, lengthening the forest cycle will keep a larger amount of carbon in trees and soils rather than in the atmosphere (Law et al., 2018). Rather than coupling funding of forest restoration or community payments to logging large trees and disturbing older forests, new policies could be enacted that compensate rural communities for protecting large trees in older forests and some of the younger trees that will become large with their associated carbon stores. To implement such policies, the amount to be paid to a community needs to be marginally greater than the revenue earned from cutting these large trees and the older forests in which they are located. Policies that provide compensation for setting aside reserves and individual trees with microhabitats are already in place in Europe (European Commission, 2015). Because this service benefits society as a whole and is an irreplaceable part of the natural climate solutions framework urgently needed for climate stabilization, the payment funds should come from the treasury since all citizens benefit from carbon accumulation by these trees and forests. With over half of Oregon's forest east of the Cascade Mountains crest, these forests are key to the State's climate mitigation and biodiversity conservation goals.

CONCLUSION

Conducting a quantitative assessment using empirical data has determined the large carbon stock that would be lost and the resulting climate consequences if these large trees are harvested. Research indicates that 2021 begins a pivotal decade for humanity to transition off of fossil-fuels (IPCC, 2018) and move "to achieve ... stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system." (UNFCCC, 1992, Article 2). To meet net-zero carbon goals by 2050, it is estimated that reductions in net carbon emissions must be 7.6% per year over the decade of the 2020s (UNEP, 2019). This is most readily accomplished by reducing fossil fuel, bioenergy and industrial carbon dioxide emissions while simultaneously accumulating more carbon dioxide by protecting existing older forests that contain the largest share of carbon, and by allowing more forests to continue to accumulate carbon through proforestation (IUCN, 2020). Proforestation allows existing forests to continue growing without harvest or other management practices so that more trees can reach the large tree size that accumulates more carbon in the near and long term than do reforestation and afforestation (Moomaw et al., 2019). No additional land is required as is the case with afforestation, and proforestation is the lowest cost opportunity for reaching the zero net carbon goal by 2050. In fire-prone forests such as in our study area, a diameter limit strikes the balance between protecting the most fire-resistant trees that store the most carbon and allowing fuels reduction with reintroduction of fire in dry biophysical environments. Intact mesic forests are ideal locations for proforestation. Harvesting large trees will add very large amounts of biogenic carbon to the atmosphere (Harris et al., 2016), and make the net zero carbon goal difficult or impossible for Oregon to achieve. The young trees will never be able to recover and accumulate the amount of carbon that is in the growing and older forests during these next critical decades, and will only equal current levels a century or more from now.

Protecting large trees to help stabilize climate is critically important for managing forest ecosystems as social-ecological systems. Knowledge of the disproportionately large amount of carbon stored in a small fraction of trees creates an opportunity to engage the public, decision makers, and forest managers in their importance as an integral part of the climate solution.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study and are available through the FIA DataMart: https://apps.fs.usda.gov/fia/datamart/CSV/datamart_csv.html.

AUTHOR CONTRIBUTIONS

DM and LB conceived the ideas and designed the study with critical suggestions from BL and RB. DM and LB collected the data. LB processed the data and carried out the modeling. DM and LB analyzed the data and led the writing of the manuscript. BL, RB, and WM assisted with writing, and contributed critically to the drafts. All authors gave final approval for publication.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/ffgc.2020.594274/full#supplementary-material

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Open Letter to Decision Makers Concerning Wildfires in the West

As scientists with backgrounds in ecological sciences and natural resources management, we are greatly concerned about proposals to speed up and expand logging on public lands in response to recent increases in wildfires in the West – proposals such as the House version of the 2018 Farm Bill. There are pragmatic, science-based solutions that can maintain biologically diverse fire-dependent ecosystems while reducing risks to communities and firefighters facing some of the most active fire seasons in recent memory. Unfortunately, such solutions are getting lost in the endless rhetoric and blaming that has characterized wildfires in the media, Congress, and the Trump administration. We the undersigned are calling on decision makers to facilitate a civil dialogue and careful consideration of the science to ensure that any policy changes will result in communities being protected while safeguarding essential ecosystem processes.

Why Is the West Burning and Is This Unnatural?

Wildfires have shaped the ecology of western ecosystems for millennia, whether lit by lightning or managed by American Indian tribes for cultural benefits. Wildfires vary in intensity and occurrence, across regions and vegetation types, elevation and climatic gradients, so there is no one-size-fits all strategy. The West has always burned and will always burn, and it needs to in order to maintain ecosystems and the myriad services they provide to the public in the form of carbon sequestration, clean water, abundant wildlife, and outdoor amenities. Attempting to suppress fires that are not a risk to communities is impractical, costly, risky to firefighters, and ecologically damaging. Also, forests are not the majority of the area burned annually on average in the United States; grasslands and shrublands are a large component of area burned annually that is unaffected by any forest management.

What is different today about wildfires is they are now burning over larger landscapes (more acres) since the 1980s, although overall fewer acres are burning today compared to that estimated in early decades and historical timelines.[1] Wildfire season in the West recently has lengthened from an average of five to seven months, and the number of large wildfires (>1,000 acres) has increased from 140 to 250 per year.[2] This is occurring as average annual temperature in the West has risen by nearly 2 degrees Fahrenheit since 1970s and winter snow pack has declined.[3] Increases in acres burning can now be attributed, in part, to climate change[4] and the

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[4]Abatzoglou, J.T., and A.P. Williams. 2017. Impact of anthropogenic climate change on wildfire across western US forests. PNAS 113:11770-11775.

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increase is expected to continue in many areas with additional warming, leading to even greater suppression costs and loss of life.[5]

In addition to climate change, more than 80 percent of fires nationwide have been caused by people,[6] and millions of homes are now in harm's way,[7] resulting in skyrocketing costs. Putting more money into fire suppression will not reduce homeowner losses as long as homes continue to be built next to fire-adapted ecosystems, lack defensible space[8] and/or fire-proofing, and measures are not taken to reduce human-caused wildfire ignitions.[6]

What Is Active Management and Does It Work to Reduce Fire Activity?

Active management has many forms and needs to be clearly defined in order to understand whether it is effective at influencing fire behavior. Management can either increase or decrease flammable vegetation, is effective or ineffective in dampening fire effects depending on many factors, especially fire weather, and has significant limitations and substantial ecological tradeoffs.

Thinning Is Ineffective in Extreme Fire Weather – Thinning is most often proposed to reduce fire risk and lower fire intensity. When fire weather is not extreme,[9] thinning-from-below of small diameter trees followed by prescribed fire, and in some cases prescribed fire alone,[10] can reduce fire severity in certain forest types for a limited period of time[11]. However, as the climate changes, most of our fires will occur during extreme fire-weather (high winds and temperatures, low humidity, low vegetation moisture). These fires, like the ones burning in the West this summer, will affect large landscapes, regardless of thinning, and, in some cases, burn hundreds or thousands of acres in just a few days.[12] Thinning large trees, including overstory trees in a stand, can increase the rate of fire spread by opening up the forest to increased wind velocity, damage soils, introduce invasive species that increase flammable understory vegetation, and impact wildlife habitat.[9] Thinning also requires an extensive and expensive roads network that degrades water quality by altering hydrological functions, including chronic sediment loads.

Post-disturbance Salvage Logging Reduces Forest Resilience and Can Raise Fire Hazards – Commonly practiced after natural disturbances (such as fire or beetle activity), post-disturbance clearcut logging hinders forest resilience by compacting soils, killing natural regeneration of

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[5]Schoennagel, T., et al. 2017. Adapt to more wildfire in western North American forests as climate changes. PNAS 114:4582-4590.
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[12]Stephens, S.L., et al. 2015. Large wildfires in forests: what can be done? Action Bioscience April 15

conifer seedlings and shrubs associated with forest renewal, increases fine fuels from slash left on the ground that aids the spread of fire, removes the most fire-resistant large live and dead trees, and degrades fish and wildlife habitat.[13] Roads, even "temporary ones," trigger widespread water quality problems from sediment loading. Forests that have received this type of active management typically burn more severely in forest fires.[13]

Wilderness and Other Protected Areas Are Not Especially Fire Prone – Proposals to remove environmental protections to increase logging for wildfire concerns are misinformed. For instance, scientists[14] recently examined the severity of 1,500 forest fires affecting over 23 million acres during the past four decades in 11 western states. They found fires burned more severely in previously logged areas, while fires burned in natural fire mosaic patterns of low, moderate and high severity, in wilderness, parks, and roadless areas, thereby, maintaining resilient forests. Consequently, there is no legitimate reason for weakening environmental safeguards to curtail fires nor will such measures protect communities.

Closing Remarks and Need for Science-based Solutions

The recent increase in wildfire acres burning is due to a complex interplay involving human-caused climate change coupled with expansion of homes and roads into fire-adapted ecosystems and decades of industrial-scale logging practices. Policies should be examined that discourage continued residential growth in ecosystems that evolved with fire. The most effective way to protect existing homes is to ensure that they are as insusceptible to burning as possible (e.g., fire resistant building materials, spark arresting vents and rain-gutter guards) and to create defensible space within a 100-foot radius of a structure. Wildland fire policy should fund defensible space, home retrofitting measures and ensure ample personnel are available to discourage and prevent human-caused wildfire ignitions. Ultimately, in order to stabilize and ideally slow global temperature rise, which will increasingly affect how wildfires burn in the future, we also need a comprehensive response to climate change that is based on clean renewable energy and storing more carbon in ecosystems.

Public lands were established for the public good and include most of the nation's remaining examples of intact ecosystems that provide clean water for millions of Americans, essential wildlife habitat, recreation and economic benefits to rural communities, as well as sequestering vast quantities of carbon. When a fire burns down a home it is tragic; when fire burns in a forest it is natural and essential to the integrity of the ecosystem, while also providing the most cost-effective means of reducing fuels over large areas. Though it may seem to laypersons that a post-fire landscape is a catastrophe, numerous studies tell us that even in the patches where fires burn most intensely, the resulting wildlife habitats are among the most biologically diverse in the West.[15] For these reasons, we urge you to reject misplaced logging proposals that will damage

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our environment, hinder climate mitigation goals and will fail to protect communities from wildfire.

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