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Review & Response

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NCASI Review of Carbon Implications of Proforestation

DECEMBER 2020

Overview

Scientific publications and popular press articles have suggested that delayed or avoided forest harvest benefits climate change mitigation through carbon sequestration and storage. Many of these articles are based on incomplete analyses of a baseline harvest scenario compared to a set of alternative reduced or no-harvest scenarios. Examples of incomplete analyses include: (1) considering carbon stocks in the forest without considering carbon stocks in harvested wood products; (2) reporting changes in total carbon stocks without considering costs of implementing those changes; (3) analyzing greenhouse gas emissions from harvesting forests without including emissions that result from not harvesting forests (using alternative products); and (4) acknowledging that carbon stored in wood products is not permanent without recognizing that natural disturbances can likewise result in impermanence of carbon stocks in unharvested forests. This review summarizes recent scientific literature regarding these factors and demonstrates their application using a case study on Douglas-fir (*Pseudotsuga menziesii*) forests in the US Pacific Northwest.

Introduction

Sustainable forest management is important for mitigating effects of climate change (Nabuurs et al. 2007). Forests remove CO_2 from the atmosphere and store it in live trees, dead wood, and harvested wood. Sustainable management of forests maintains this contribution while providing needed products and environmental co-benefits.

The value of carbon (C) storage in forests to mitigate climate change has received widespread attention, which has led some to conclude that the primary goal of forest management should be to increase stocks of forest C. For example, Moomaw et al. (2019) suggested that leaving forests "intact" (undisturbed) to reach their ecological potential (for storing C) is an "effective, immediate, and low-cost approach" to mitigating climate change. They coined the term "proforestation" to mean reducing or eliminating timber harvest to increase C storage in forests.

Similarly, numerous studies have suggested that climate change mitigation would be supported by lengthening rotations for commercial forest harvests, which necessarily involves reducing annual harvest quantity. For example, Diaz et al. (2018) simulated longer rotations for Douglas-fir forests in the US Pacific Northwest (PNW) and concluded that lengthening rotations from about 40 years to 75 years would be beneficial, albeit with a "non-trivial financial gap." Likewise, Graves et al. (2020) examined reductions in harvests of all species on private lands in Oregon and found that substantial harvest reductions over a period of 30 years would lower the state's greenhouse gas (GHG) emissions, but they did not attempt to estimate associated costs. In a comprehensive literature review, McKinley et al. (2011)

discussed decreasing forest harvest as a forest management strategy for increasing forest C and noted that harvesting high-biomass forests and replanting would reduce overall forest C stocks more than if stands were unharvested, even counting storage in wood products. Creutzburg et al. (2017) came to different conclusions in a simulation of coastal Oregon forests, suggesting that shifting away from current timber management practices could substantially decrease ecosystem C storage.

To help clarify considerations needed when discussing proforestation, this review will (1) summarize relevant scientific literature that provides insights to these claims; (2) list factors that must be considered in a comparison of forest management scenarios to mitigate climate change; and (3) demonstrate numerical results from such a comparison using a fairly simplistic model.

Literature Review

Increasing Stocks versus Increasing Sequestration

Numerous scientific sources (e.g., Allen et al. 2009; IPCC 2013; Miner et al. 2014; USGCRP 2017) have argued that the primary climate goal should be reducing net emissions of GHGs to the atmosphere. Net emissions are computed as total emissions of GHGs minus any removals (sequestration) of these gases from the atmosphere (e.g., by forests):

$$\text{Net Emissions} = \text{Emissions} - \text{Removals}$$

Net sequestration is simply the negative of net emissions (Removals – Emissions). Increasing net sequestration is equivalent to reducing net emissions. Thus, a primary climate objective is increasing net sequestration.

Increasing net sequestration may or may not be associated with increasing forest C stocks. Although reducing the level of harvesting may lead to an increase in forest C stocks, it may also lead to increased use of substitute products that are accompanied by much higher emissions from production and use. For example, Churkina et al. (2020) reported that cumulative emissions from manufacturing mass-timber materials are lower than those from mineral-based materials (such as concrete and steel).

Therefore, a focus on forest C stocks, without consideration of associated emissions, can lead to false conclusions about the relative merits of scenarios that increase forest C stocks.

Old Forests Grow More Slowly

In forestry terms, sequestration is equivalent to forest growth. Every ton of forest biomass contains 0.5 tons of elemental C, so growth rates (in tons per acre per year) are perfectly correlated to sequestration. Because net sequestration is a primary goal, a focus on the *rate* of sequestration is more relevant than a focus on accumulated forest C stocks. Therefore, while it is true that old forests store (contain) more C than younger forests, they also sequester C at a much slower rate. Gray et al. (2016) evaluated C stocks (quantities) and sequestration (rates) in forests of the PNW. They noted that because of increasing mortality in older forests, their rate of net sequestration (growth minus mortality) was not significantly different from zero in old-growth stands. It is widely acknowledged that forest growth trajectories show more rapid growth at young ages than at older ones, and maximum long-term growth is best achieved at harvest rotations near the culmination of mean annual increment (peak of average annual growth; Diaz et al. 2018).

Not All Forest Carbon Remains in the Forest

Forest C occurs in various “pools”. Forest C stocks can be estimated in pools for live trees, standing dead trees, fallen dead trees and branches (termed downed woody material or coarse woody debris), understory vegetation, forest floor (decomposing plant material above the mineral soil layers), soil organic C, and harvested wood.

When C is removed from the atmosphere by live trees, it is stored in the live tree C pool before transitioning into the standing dead pool (by tree mortality) or the harvested wood pool (by forest harvest). Standing dead trees eventually decompose and most of their C is emitted to the atmosphere. Similarly, some portion of harvested wood is lost to logging and manufacturing residuals and eventually decomposes, is combusted as waste, or is interred in landfills after use. The dead wood and harvested wood pools therefore have similarities: both represent eventual destinations for almost all live tree C; both result in decay or combustion that returns much of their C to the atmosphere; and both have widely varying timeframes in which these processes occur.

According to McKinley et al. (2011), about two-thirds of discarded wood is landfilled after use, where 77% will remain stored indefinitely (Skog 2008). For paper products, one-third of discarded paper is landfilled, with 44% not subject to subsequent decay. Smith et al. (2006) indicated that 40% of C in softwood lumber and plywood remains stored in products in use or in landfills 100 years after production. Therefore, it is critical that any analysis of C from managed forest where harvests take place must consider that a significant portion of that C may be stored offsite for decades to a century or more.

Permanence of Carbon Storage

Some studies discount the value of C storage in harvested wood products due to the impression that such storage is impermanent. However, it is important to note that C storage in forests also faces risks, leading to impermanence of forest C stocks. Anderegg et al. (2020) recommended that climate policies must fully account for these permanence risks because they could undermine effectiveness of forest-based climate solutions. The California forest C offset program acknowledges the risk of C loss from forests by requiring projects to submit 16% of credits to a buffer pool to offset expected losses due to risks such as fire, drought, insects and pathogens, and weather-related disturbances (hurricanes and wind damage, snow and ice events, and lightning). Many of these risks are expected to increase with climate change (Anderegg et al. 2020); climate-related risks contribute to more than half of those covered by the buffer pool. Assuming long-term undisturbed storage of C in forest ecosystems therefore overestimates the C outcomes of no-harvest scenarios.

Emissions from Harvesting and from Not Harvesting

To arrive at net sequestration, information is needed not only on removals of CO_2 from the atmosphere, but also emissions. When comparing scenarios, it is important to recognize that there are emissions from harvesting *and* emissions from not harvesting.

Emissions from harvesting may include fossil fuel emissions from harvesting and transportation equipment and manufacturing processes. As an example, Chen et al. (2018) estimated GHG emissions from harvesting and manufacturing of lumber in Canada to be 49 kg CO_2 equivalent (CO_2e) per cubic meter of wood harvested. In a life-cycle assessment for forest residue processing, Chen et al. (2017) computed emissions from a suite of logistical systems implemented in Oregon and Washington, arriving at emissions ranging from 38 to 57 CO_2e/bone dry ton of wood. Embodied emissions are often calculated using life-cycle analyses and reported as part of Environmental Product Declarations.

However, in a comparison of scenarios, it is important to also consider emissions that result from decreasing or halting timber harvest. Commercial timber harvest is conducted in response to market demand. If a scenario including harvest is compared to a scenario without (or with lower) harvest, it must be recognized that the demand for wood products does not simply vanish. A combination of two outcomes must occur: wood products will be acquired from some other locations (state, region, country); and/or some other product will be used in place of wood products. The first of these outcomes is known as "leakage" (Gan and McCarl 2007) and the second as "substitution" or "displacement" (NCASI 2020).

Leakage resulting from reduced timber harvest has been examined in forest economics literature. Wear and Murray (2004) studied timber harvest reductions on public lands in the PNW in the 1990s, when harvests declined 85% from 1988 levels. They found that 43% of the reduced public harvest was replaced by increased harvest on private timberlands, 15% by increased harvest elsewhere in the US, and 26% by increased harvest in Canada, for a leakage of 84%. Murray et al. (2004) applied econometric models to estimate leakage from forest C projects in the PNW and found that 16% of reductions in the PNW were offset by increases elsewhere. When leakage occurs, it means that reductions in harvest have not been fully effective at reducing emissions because some portion of those emissions are simply relocated by market activity. Recognizing this effect, the California Air Resources Board (CARB) requires improved forest management projects to incorporate a 20% leakage factor when they involve a reduction in harvesting compared to a baseline (CARB 2015).

A substitution or displacement effect occurs when one product is used in place of another with different levels of embodied emissions. A substitution factor quantifies the efficiency of using a wood-based product to reduce GHG emissions to the atmosphere compared to a non-wood alternative product (Sathre and O'Connor 2010; Leskinen et al. 2018). Substitution factors depend on assumptions about what product, and how much of it, is required to substitute for a wood product, and about emissions associated with producing both the wood and the non-wood products. A substitution factor reports the change in GHG emissions associated with using a non-wood substitute product per unit of C in the wood product (NCASI 2020). As examples, reported substitution factors for wood construction materials include 0.54 (Smyth et al. 2017), 1.3 (Leskinen et al. 2018), and 2.1 (Sathre and O'Connor 2010).

The practical effect of using substitution factors was noted by Fain et al. (2018) and Perez-Garcia et al. (2005), who reported that studies including substitution resulted in shorter C-optimal rotations and yielded more C benefits, but studies that did not consider substitution resulted in longer C-optimal rotations.

Cost of Changes in Carbon Stocks

While the focus should be on net C sequestration and not simply C stocks, changes in C stocks are a relevant indicator of C dynamics. Furthermore, forest C offset markets provide financial rewards for increasing forest C stocks. Offsets are paid for by emitters under cap-and-trade programs and sold by parties who can demonstrate C stock increases over a baseline. Offsets are traded in markets and measured in metric tons (MT) of CO_2e. A variety of mechanisms generate emissions reductions (and/or increase sequestration), and efficiency suggests that lower-cost options should be employed first.

Deferment or halting of forest harvest may incur a variety of costs: lost stumpage revenue to landowners; lost production at manufacturing facilities; reductions in employment; loss of timber-dependent businesses; lost tax revenue; and others. Perhaps the most immediate and easiest to quantify is the lost stumpage revenue. Sohngen and Brown (2008) evaluated opportunity costs for deferring harvest in Oregon and found costs of \$30 to \$50/t CO_2e (using stumpage prices of \$90 to \$120/MBF). Graves et al. (2020) suggested that reduced harvest provides climate benefits, but acknowledged that any financial benefit (e.g., C offsets) would not cover the cost (lost timber revenue). Diaz et al. (2018) also acknowledged that substantial incentives or price premiums must be available to landowners to engage in delayed harvest for C benefits but did not compute a price per ton in their analyses. Moomaw et al. (2019) simply claimed that leaving stands unharvested is a low-cost strategy but did not cite studies or conduct analyses to support this conclusion.

Regulatory approaches (rather than market approaches) to lengthening rotations may result in outcomes detrimental to climate change mitigation–loss of forest land. Numerous studies (Lubowski et al. 2008; Abt et al. 2010, 2014; Costanza et al. 2016; Dale et al. 2017; Birdsey et al. 2018; Kim et al. 2018) have concluded that when landowners have access to markets for wood products, they keep more land in forests and increase productivity of those forests. Therefore, removing financial returns to landowners by limiting access to such markets would lead to lower productivity and less forest area, with negative consequences for C stocks and attendant ecosystem services from forests.

Factors That Must Be Considered in Comparing Forest Carbon Scenarios

Summarizing the literature, there is compelling scientific support for including these factors in a comparison of scenarios involving forest management for C benefits:

1) Focus should be on net C sequestration, not simply C stocks.
2) Younger forests sequester C at a faster rate than older ones, which have higher cumulative stocks.
3) C is stored for years to decades in wood harvested from forests, and scenarios involving forest harvest must account for this storage.
4) C stored in forests is susceptible to a variety of natural and anthropogenic risks. Such risks of C loss should be reflected in analyses that evaluate long-term C storage in forests, as they are in estimates for storage in harvested wood products.
5) Emissions from both harvesting and not harvesting must be considered; the latter includes emissions from leakage and/or substitution.
6) Cost differences among scenarios (expressed per ton of CO_2e) are necessary to place results in context with other options for climate mitigation.

Case Study

To demonstrate a scenario comparison with all the above elements, forest C dynamics on privately-owned, planted Douglas-fir forests in Oregon and Washington were modeled. Forest area, current inventory, growth rates, mortality rates, and harvest rates by 10-year age class in a cohort model were estimated to compare four scenarios, summarizing C stocks and emissions by decade for a 100-year period.

Study Population

Privately-owned, planted Douglas-fir forests were chosen as a study population because:

- Douglas-fir forest can grow to very old ages, unlike some early-successional species such as southern yellow pines (*Pinus* spp.).
- The PNW landscape has been the focus of several papers examining extended rotations or decreased harvest (Hudiburg et al. 2009; Creutzburg et al. 2017; Diaz et al. 2018; Fain et al. 2018; Law et al. 2018; Graves et al. 2020).

- Planted private forests represent an investment by landowners that expect a return, so opportunity costs are clearly quantifiable and relevant.
- Planted forests fit well within an even-aged cohort modeling approach such as the one used.

The study population consisted of 5.4 million acres, about 27% of all Douglas-fir forests in Oregon and Washington (Figure 1). Eighty-seven percent of private planted Douglas-fir forests were under 40 years old, and 99% were under 70 years old. Because of the high variability of estimates with small sample sizes, values were computed only for age classes in which there were at least ten remeasured inventory plots.

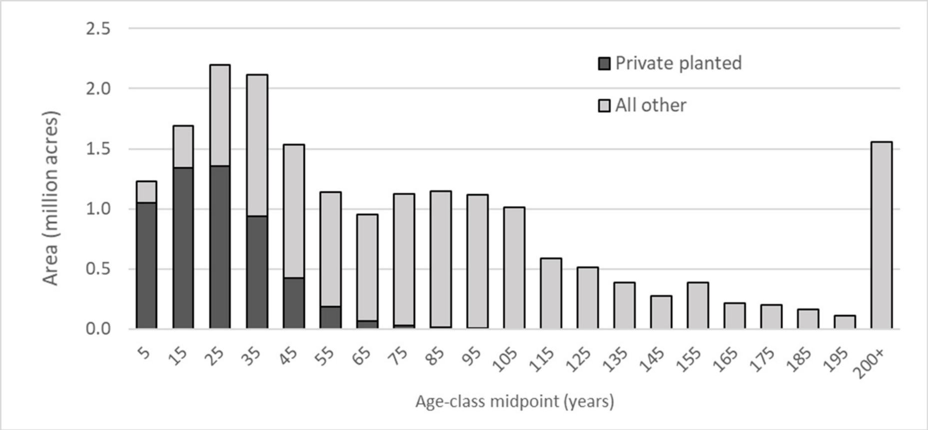

***Figure 1**. Area of Douglas-Fir Forests in Oregon and Washington by Age Class.*

Inventory Data

All data for this analysis were obtained from the Forest Inventory and Analysis (FIA) program of the US Forest Service (USDA 2020) using the most recent data posted: the 2019 evaluation group[1]. As a starting point, the current inventory of aboveground live tree biomass (tons) converted to tons of carbon (tC) was used. Aboveground live trees are the primary C pool that is measured and subject to management. Private planted forest accounted for only 16% of total Douglas-fir inventory in the two states (Figure 2).

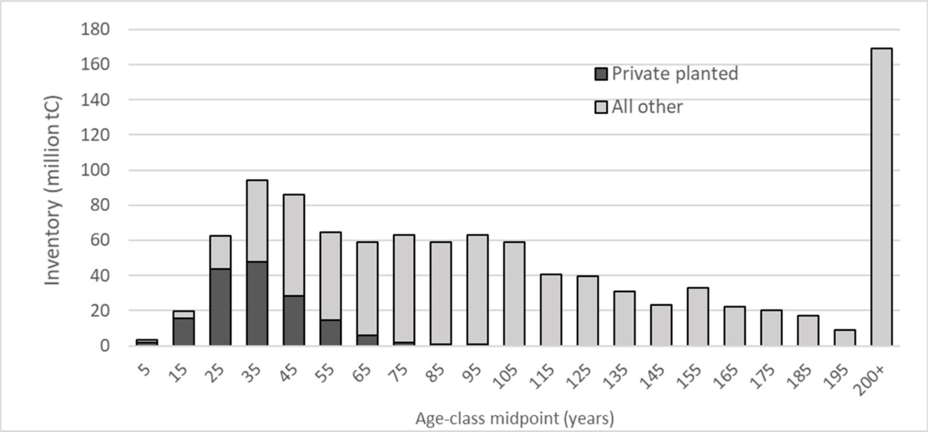

***Figure 2.** Aboveground Live Tree Carbon Inventory by Age Class.*

[1] An evaluation group is a complete set of inventory plots for a state as of a specified year. The 2019 evaluation group contains data from plots measured over the 10-year period up to and including 2019.

Next, estimates of “gross growth” as the annual C increment (tC/ac/yr) in aboveground live trees were attained, representing the quantity of C removed from the atmosphere (Figure 3). Private planted forests had higher growth rates than other forests at younger ages (<30) and comparable growth afterwards. It is also notable that growth rates for older forests (>70 years) were essentially flat at about 1 tC/ac/yr, and forests in the 10- to 30-year age range had growth rates more than double the older age classes.

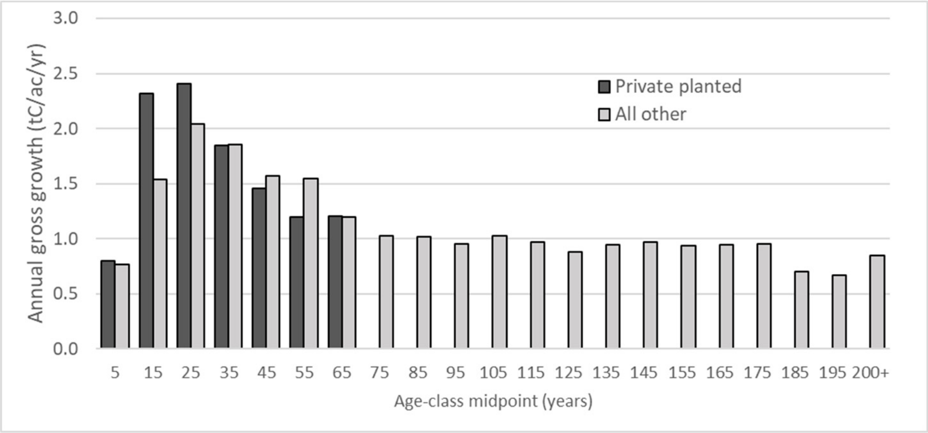

Figure 3. *Annual Gross Growth (sequestration) by Age Class.*

Annual mortality was measured as a percentage of inventory at midpoint of an age class (Figure 4). Much of the tree mortality is density dependent at young ages, which is why thinning is often used as a silvicultural intervention. Expressing mortality as a percent of inventory captures the effect of decreased mortality when stand density is reduced through thinning. Annual mortality dropped rapidly over the first few decades and then stabilized around 0.5%.

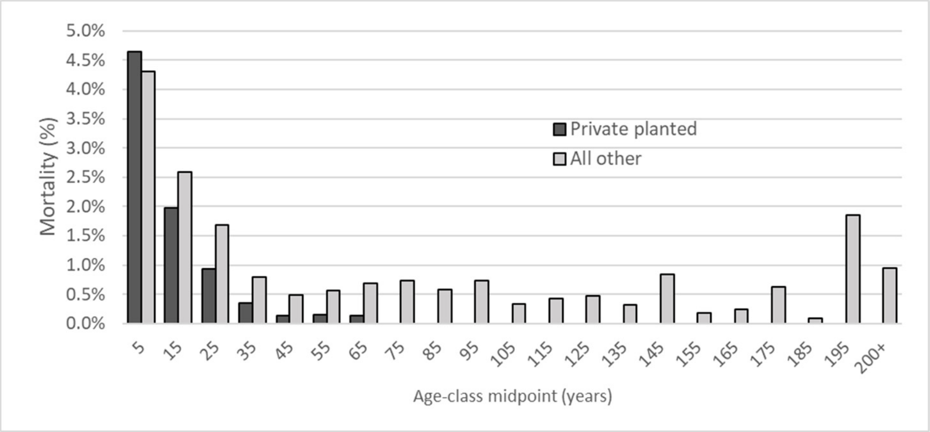

Figure 4. *Annual Mortality as a Percent of Inventory by Age Class.*

Finally, biomass harvested from forests was also estimated using FIA data (Figure 5). About 70% of total harvest from Douglas-fir forest in these states came from privately-owned planted lands. The PNW FIA database contained information on type of harvest and revealed that 94.5% of harvest came from the regeneration harvest at the end of a rotation, with just 5.5% coming from thinning. Slightly more than half of the volume removed by thinning came from the 21- to 30-year age class, with the rest fairly evenly distributed across the next older three age classes.

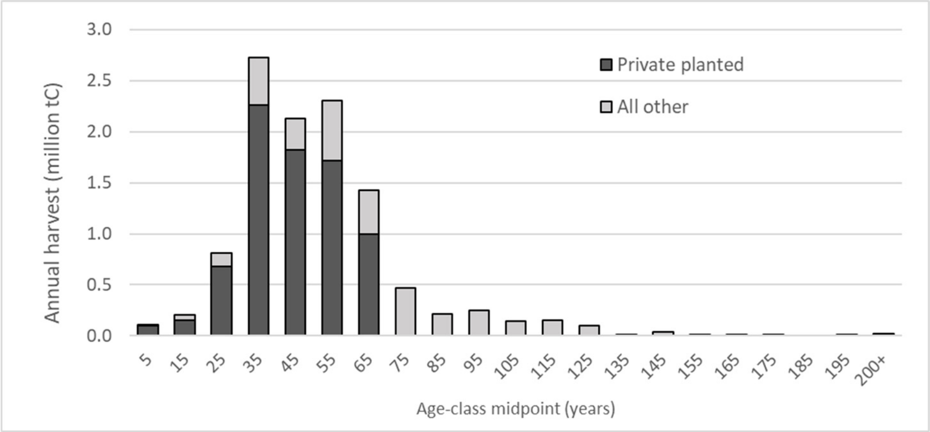

Figure 5. *Annual Harvest Removals by Age Class.*

Model Framework

In the age-class cohort model, the initial age-class distribution (and inventory amounts) moved decade by decade into older age classes, adding growth and subtracting mortality and harvests at each step. Carbon from mortality moved into the dead wood pool where it decomposed according to published rates (Harmon and Hua 1991). Harvest by thinning was removed according to patterns observed in the PNW FIA database (5.5% from thinning, distributed across relevant age classes). Rotation harvests were assumed to begin with the oldest age classes, working down the age-class distribution until harvest targets were met. Harvested C was distributed to logging residues (2.9%; Simmons et al. 2016) that entered the dead wood pool, and product distributions (hardwood and softwood, pulpwood, and sawtimber) with associated decay rates following Smith et al. (2006). Acres subjected to rotation harvest moved into the 0- to 10-year age class and were assumed to be replanted, with one-year delay for site preparation and planting.

Forest C stocks in aboveground live trees, aboveground dead wood, and harvested wood products were tracked by decade. Emissions from decay of dead wood and combustion/decay of harvested wood products were also recorded. For scenarios involving a change in harvest level compared to the baseline, substitution factors reflected the increase or decrease in emissions from substitutes for wood energy and softwood sawtimber. This analysis used the lowest (most conservative) substitution factor for softwood sawtimber, 0.54 kgC/kgC (Smyth et al. 2017).

Scenarios

Four scenarios were used to illustrate differences in C dynamics from various harvest levels and resulting rotation lengths (Table 1). The Baseline scenario used current annual harvest levels reported by FIA and resulted in an average age at harvest of 52.8 years at the end of the 100-year simulation. The Increased and Decreased harvest scenarios were based on harvest levels 10% above and below baseline, respectively, and resulted in 40- and 65-year rotations. The Proforestation scenario assumed all stands were left intact (no harvest) for C storage.

Table 1. *Summary of Scenarios.*

	Baseline	Increased	Decreased	Proforestation
Harvest (tC/yr)	8,282,976	9,111,274	7,454,678	0
Average stand age @ year 100	29.1	22.9	35.9	113.9
Average harvest age @ year 100	52.8	40.3	65.0	NA

Results

Carbon Stocks

The Proforestation scenario resulted in the highest levels of forest C stocks after 100 years, with about 800 million tC stored in aboveground live trees, standing dead trees, and down woody material (Figure 6). The other three scenarios included harvests that resulted in C storage in harvested wood products and in forest stands. Harvests in the Increased and Decreased scenarios resulted in 2% less and 1.9% more C stored after 100 years, respectively.

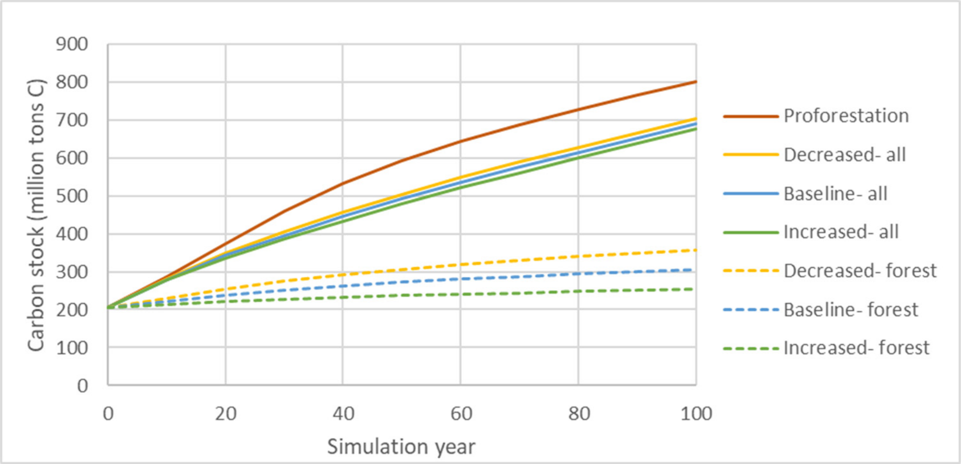

Figure 6. *Carbon Stocks for Four Scenarios.*
[“forest” indicates stocks remaining in forest (live tree, dead tree, down dead wood), “all” includes harvested wood product stocks as well; the difference between dashed “forest” stock lines and solid “all” lines represent the contribution of harvested wood products to total carbon storage; there are no harvested wood stocks in the Proforestation scenario]

Emissions and Sequestration

Gross sequestration (growth) and emissions from dead wood decay, harvested wood products, and substitution are summarized for each scenario in Figure 7. Emissions include decay of dead wood in the forest, combustion or decay of harvested wood, and emissions from production and use of substitute products and energy.

Because growth is a function of age, it is highest in the Increased scenario where average stand age is lowest and lowest in the Proforestation scenario. Emissions are also lowest in the Proforestation scenario, which has substantial substitution emissions but no emissions from harvested wood products. In the Increased scenario, harvest is greater than baseline and the substitution effect results in lower (negative) emissions from substitution, serving as an emissions reduction (hatched area in Figure 7).

Net sequestration is the difference between gross sequestration (growth) and total emissions, represented by the distance between the dashed line and the total of emissions in Figure 7. Emissions exceeded growth after 60 years in the Proforestation scenario, so net sequestration became negative. Net sequestration is summarized for all scenarios in Figure 8 and Table 2. The average annual net sequestration of the Proforestation scenario was less than half that reported for any other scenario.

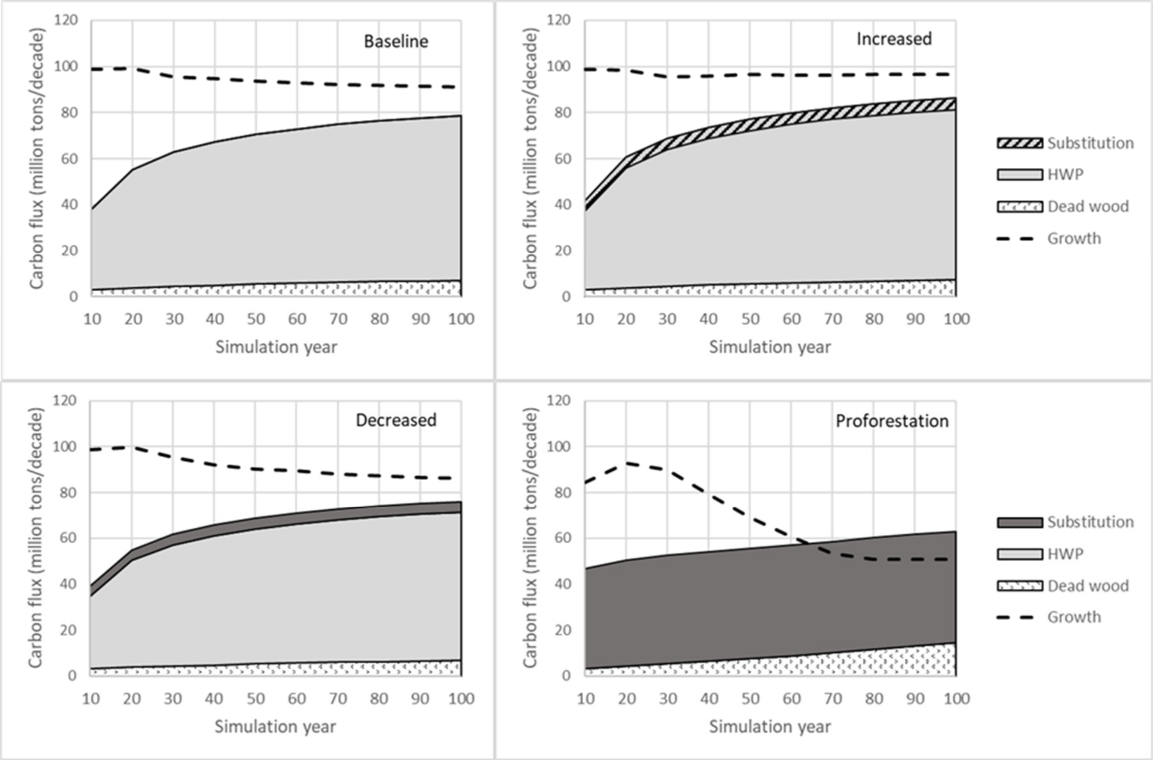

Figure 7. *Gross Sequestration (growth) and Emissions for the Four Scenarios (in million tC per decade).*
[emissions include decay of dead wood, combustion/decay of harvested wood, and emissions from substitute products; net sequestration is the difference between growth and total emissions]

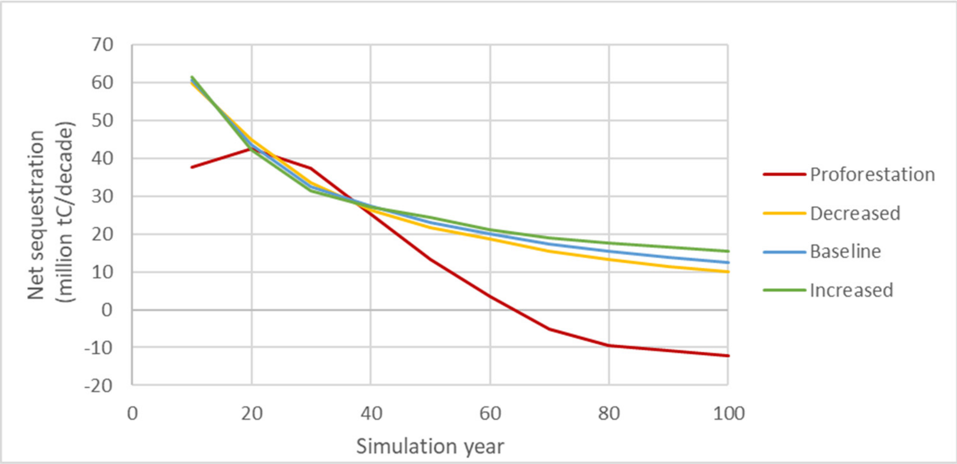

Figure 8. *Net Sequestration for the Four Scenarios (in million tC per decade).*

__Table 2.__ Summary of 100-Year Average Annual Sequestration, Emissions, and Net Sequestration by Scenario. (all values in million MT CO_2e/yr)

	Baseline	Increased	Decreased	Proforestation
Annual sequestration (growth)	31.3	32.2	30.4	22.7
Annual emissions from dead wood	1.8	1.8	1.8	2.84
Annual emissions from HWP	20.7	22.7	18.6	0.0
Annual emissions from substitution	0.0	-1.6	1.6	15.8
Annual net sequestration	8.9	9.2	8.5	4.1

Cost of Carbon Storage

The change in C storage between the Baseline and other scenarios is the distance between their respective lines in Figure 6. The cost for this change in storage was computed as the softwood sawtimber stumpage revenue lost or gained in each scenario compared to the baseline, using a nominal stumpage price of $250/MBF and converting to C units (Table 3). For the Increased scenario, annual stumpage income was greater by $138 million but stocks declined by an average of 131 thousand tons per year. This resulted in a parity price for C of $1055/tC ($317/MT CO_2e) that could be paid to offset C losses. The Decreased harvest scenario increased stocks over the baseline by an average of 130.5 thousand tons per year at a cost of $139 million per year, for a C cost of $1060/tC ($319/MT CO_2e). Similarly, the Proforestation scenario resulted in C stock increases at a cost of $373/MT CO_2e.

__Table 3.__ Summary Values for Estimating Cost of Carbon Storage.

	Baseline	Increased	Decreased	Proforestation
Harvest (tC/yr)	8,282,976	9,111,274	7,454,678	0
Annual stumpage value (million $)	$1384	$1522	$1245	$0
Average annual stock change over 100 years (million tC/yr)	4.845	4.714	4.975	5.959
Cost per tC	$0	$1055	$1060	$1241
Cost per MT CO_2e	$0	$317	$319	$373

Computing the cost per ton of increased C storage allows comparison of reduced harvest levels or proforestation with other mechanisms to reduce emissions or increase terrestrial C stocks. For example, market prices for forest C offsets have recently traded at about $13 to $15/MT CO_2e (Dezember 2020), or less than 4% of the proforestation cost. Carbon capture and storage, sometimes viewed as prohibitively expensive, has been included in California's Air Resources Program Low Carbon Fuel Standard program with credits trading at about $186/t$CO_2$e (Beck 2020), about half of the proforestation cost.

Risks of Impermanence

When forest C projections are made using growth and yield equations or models, additional steps may be required to incorporate the risk of C losses due to natural disturbances. However, when projections are made based on forest inventory data (as in this case study), the effect of current levels of natural disturbance is embedded within the data. For example, the mortality estimates in Figure 4 include C lost annually from the live tree pool through mortality due to weather (e.g., ice damage, wind; 183.7 ktC/yr), disease (170.3 ktC/yr), insects (30.5 ktC/yr), animal damage (12.2 ktC/yr), and fire (7.6 ktC/yr).

If natural disturbance is not included in the data/models used for projection or if future disturbance risks are expected to exceed current levels (Anderegg et al. 2020), a risk discount factor (CARB 2015) could be applied. As an example, a 10% risk discount applied to the Proforestation scenario would result in 10% less credit claimed for C stock changes (row 3 of Table 3) at the same cost, leading to a cost of \$415/MT CO_2e.

Conclusions from Case Study

This case study on privately-owned, planted Douglas-fir forests in Oregon and Washington demonstrates that reducing or eliminating harvest from a study area can increase stocks of C stored in both forests and harvested wood products, but does so at a very high cost, even when only one cost component is included (lost stumpage revenue to landowners). This suggests that far less expensive options are available to increase terrestrial C sequestration to mitigate climate change. Furthermore, reducing or eliminating harvest results in (1) a much older forest with slower growth and lower annual sequestration; and (2) substantial emissions generated in producing and using substitute products. The combination of these consequences resulted in the Proforestation scenario having a negative C balance (emissions exceeding sequestration) after 60 years and overall average net sequestration less than half that of any other scenario.

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