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Accounting for the climate benefits of harvested wood utilization

Author
Cabiyo, Bodie Happy

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By

Bodie Cabiyo

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Professor Lara Kueppers, Co-Chair
Professor Matthew Potts

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Abstract

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Bodie Cabiyo

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Professor Daniel Sanchez, Co-Chair
Professor Lara Kueppers, Co-Chair

Adequately addressing climate change will require a diverse set of approaches to both mitigation and sequestration of carbon emissions. Recently, carbon dioxide removal (CDR) has become an increasingly important climate change mitigation strategy that depends on the ability to store carbon for long time periods. Forest carbon sequestration represents a key CDR opportunity that is regularly highlighted in academic literature and policy. But carbon stored in forests is inherently dynamic: both social and natural disturbances contribute to the flux of carbon from forests. In many forests, these disturbance events are expected to increase into the future, particularly with climate change. Forest management interventions like thinning, prescribed burning, and species selection can mitigate natural disturbance risks, but these strategies often release carbon in the short term and will have to evolve in the long term as climate change disrupts the resilience of forest ecosystems. Carbon stored in harvested wood products (HWPs) has the potential to be durable for decades to 100's of years, but the durability of carbon stored in HWPs is highly variable and depends on numerous factors that are difficult to accurately quantify. Forests play a key role across the spectrum of CDR opportunities, from reforestation to bioenergy with carbon capture and storage (BECCS). Throughout this dissertation, I work to understand the full carbon life cycle of some of these strategies and the implications for climate change mitigation efforts.

This dissertation focuses predominantly on questions of carbon storage, both in disturbance-prone forests and in various forms of harvested wood use. In the first project, I analyze wildfire risk, carbon storage in forests, and innovative uses of low-value wood residues. In the second project, I consider temporal accounting issues in HWPs. And in the third project, I consider the HWP life cycle at a global scale to identify opportunities for climate change mitigation. Although these projects cover a wide breadth of topical and regional emphasis, they are all linked in their shared motivation to understand carbon storage in HWPs as a critical component of forest carbon systems.

In Project 1, I consider the carbon implications of existing forest management goals and the potential impact of increased use of emerging wood products. Specifically, I assess the State of

California’s stated goal to treat 1 million acres of forest per year for fire hazard reduction, alongside its aggressive commitments to economy-wide carbon neutrality. Some research has suggested that forest treatment is at odds with climate goals. Treatments are often costly, and large amounts of low-value wood are often burnt or left to decay. Here, I assess climate and wildfire outcomes across several wood use and forest management scenarios. I find that with a suite of innovative wood uses, increased management and wood use could yield net climate benefits between 5.4-15.9 million tonnes of carbon dioxide equivalent (Mt CO_2e) per year when considering impacts from management, wildfire, carbon storage in products, and displacement of fossil-intensive alternatives. I find that products with durable carbon storage confer the greatest benefits, including traditional timber products and products with carbon capture and storage. Concurrently, I find that treatment could reduce wildfire hazard on 12.1M acres, 3.1M of which could experience stand-replacing effects without treatment. My results suggest a low-cost pathway to support California's climate adaptation and mitigation goals.

In Project 2, I consider the impact of simplified HWP accounting in a prominent forest carbon offset protocol. In Improved Forest Management (IFM) carbon offset projects registered under the California Air Resources Board Forest Carbon Protocol, carbon offset credits are generated against a baseline which represents counterfactual carbon storage in-forest and in wood products without offset revenue. Often, the chosen baseline for in-forest carbon stocks is well below the initial carbon stocks and, as a result, most projects generate a large proportion of lifetime credits in the first year. Further, the protocols produce baselines that are static through time. This is problematic because carbon in harvested wood changes over time as products go out of use and decay. To simplify the accounting, offset protocols take a single point in time as representative – the average of 100 years of decay. This simplification underestimates the carbon stored in harvested wood products in the counterfactual, resulting in project over-crediting. I find this simplified accounting yields 42 MtCO_2*e* of credits generated too early, nearly half of the credits in the study sample. Using a static baseline underestimates carbon stored in wood products initially and overestimates carbon stored at the end of the project. Functionally, this error delays the climate benefits of the program by offsetting fossil fuel emissions today with emissions reductions that won’t be realized for decades.

In Project 3, I expand the aperture of these questions to a global scale and consider the global life cycle of HWPs and the potential for climate change mitigation interventions in that life cycle. Multiple studies have investigated components of the global HWP life cycle, but none have considered the full life cycle of HWPs from gate to grave for all HWP categories or the potential role of emerging carbon capture technologies in the global HWP lifecycle. In this project, I model carbon emissions and carbon storage for all HWPs at a country scale, including production emissions, the product use phase, and product end-of-life. Following this, I model potential interventions to store carbon or reduce emissions in the HWP life cycle. I find that the global HWP sector is a net carbon sink in 2020 but, with implementation of CCS at mills could become a sink of 3.3 Gt CO_2/yr by 2050, not counting methane emissions which are uncertain but likely large. Approximately half of this sink would be attributed to baseline carbon storage in products and landfills, and the other half could be realized through CCS. In total, CCS of biogenic CO_2 could reach 1.2-1.8 Gt CO_2/yr by 2050, most of which would be at pulp and paper mills. I conclude that retrofitting existing mills with CCS is a potentially low-cost application of BECCS technologies that requires minimal new infrastructure and little additional biomass feedstock.

Dedication

I came to ERG to be a social scientist but realized that I'm compelled by all aspects of complex problems. For better or worse, ERG nurtured my tendency to be a jack of all trades. Six years later, I still feel like a master of none, but I have learned how to apply my dilettante nature in a meaningful way. Education has been a reliable ladder throughout my life, and I'm so grateful to ERG for providing the final rungs.

I've had many mentors over the last six years -- far more than I could thank here. Special thanks to: Dan Sanchez for saving me from a lifetime of working on clean cookstoves. Lara Kueppers and Matthew Potts for keeping me balanced between the worlds of ecology and management. Cathy Koshland for proving the importance of curiosity. Isha Ray for holding my hand while I pretended to be a social scientist. David Levine for making me think and write clearly. Kay Burns for showing me the power of empathy in professional settings. My many co-authors and collaborators who have made this possible and challenged me on my thinking and writing throughout; I acknowledge you in each chapter. My family for keeping me grounded. And Esther for being my Light of Eärendil along the way.

Like ecology, ERG is messy and beautiful. In this moment, I am overflowing with gratitude to this incredible community of a program. I am not only a better thinker but, unequivocally, a better person because of it.

The durability of stored forest carbon: an introduction

Bodie Cabiyo

Introduction

Adequately addressing climate change will require a diverse set of approaches to both mitigation and sequestration of carbon emissions. Recent work has emphasized the potential of forests to help meet climate goals in the near- and long-term (1–3). Still, tremendous uncertainties exist around aligning forest management and climate goals. Estimates of how forest management impacts net carbon emissions from forests vary substantially (4–6) and commercial forest management often promotes sub-optimal carbon and/or ecological outcomes (7). Balancing the diverse benefits we expect from forests is inherently complex, though, and necessitates tradeoffs across ecological and economic values (5, 8). The tradeoffs between timber supply and ecological value are often divisive, but the need for carbon sequestration and storage may offer more convergent management solutions.

In managed forests, more efficient use of harvested wood results in better forest carbon outcomes, but different wood products vary substantially depending on production emissions, substitution benefits, and end-of-life emissions (9–11). At the same time, in some situations the best carbon and economic outcome is to not harvest forests at all (12). Understanding the carbon tradeoffs between harvesting wood and retaining carbon storage in forests is critically important. Forest carbon management decisions will only increase in complexity as new wood products become viable.

Adding to this complexity, changing climate regimes threaten the ecological resilience of forests (13, 14). These threats are diverse, hard to predict, and may interact in unforeseen ways (14, 15). In some forests, climate-induced effects may shift the ecological regime indefinitely, often reducing carbon storage (13). At the forest stand level, ecosystem carbon stocks can fluctuate dramatically over decadal time scales as trees in that stand grow, die, and decay. At the landscape level, these localized fluctuations can coalesce into landscape-scale trends (16). In some cases, though, whole landscapes can be affected by catastrophic events like large wildfires or pest outbreaks. Different approaches to management and planning can significantly alter the resilience of forests (17). Management actions like thinning, fuels reduction, and controlled burning can reduce risks like wildfire. Planning actions like increasing species diversity and reducing tree planting density can mitigate future risks, although these actions may not always be aligned with maximizing short-term carbon storage (18).

Management, past and present, can interact with climate effects to exacerbate or ameliorate the resilience of forests to change. The ability of forests to adapt and continue to provide the benefits we expect from them will, in part, depend on a diverse set of policies and management decisions (19). As policies supporting climate mitigation in forests become more common, policymakers will need robust understanding of the risks and uncertainties around carbon sequestration and storage, both in the forest and in wood products.

Integrated approaches to management and production of wood products can lead to short-term carbon losses due to harvesting and long-term accumulation of carbon stored in long-lived wood products, landfills, and other engineered carbon storage. Like this, linking ecosystem carbon storage with engineered storage can produce multiple benefits, including forest resilience, increased revenue, and permanent carbon storage in wood products. This linkage also can address one of the key challenges of nature-based carbon removal: assuring permanent storage in a dynamic system.

Carbon storage in the biosphere in a changing world

Disturbance is a critical part of the ecology of forest systems. One of the key challenges of using forests for carbon sequestration is in reconciling the reality of disturbance with the expectation of constant carbon storage. This challenge is exacerbated by climate change. Broadly, climate change is expected to push forest systems towards younger, shorter, less carbon-dense forests (13). These future forests are expected to have higher rates of mortality due to climate-exacerbated disturbances, making the carbon they store less stable (13, 14). These disturbances are expected to increase in both frequency and severity. These risks are also exacerbated by forest stand structure: in some cases, forests with higher carbon density are less resistant to disturbances (18).

<u>Wildfire</u>

Wildfire risk is increasing worldwide as a result of climate change (20). In wildfire prone ecosystems, fires are expected to increase both in frequency and severity. However, fire severity is also a function of stand structure. Forest stands with dense tree cover and accumulated woody fuels can be at higher risk of catastrophic wildfire, particularly in dry ecosystems (21). In fact, historic wildfire frequencies in the Western US were much higher than present day but had relatively low severity because of their stand structure. Those fires burned at a much lower intensity resulting in little mortality and limited fuel accumulation (22). Fire hazard can also be increased through dead wood accumulation from other disturbances like drought and biotic invasion (23). Thus, effective fire hazard mitigation involves managing stand structure as well as mitigating interacting climate risks.

In part, managing for carbon storage and managing for low wildfire hazard are at odds. The projects with the highest amount of short-term carbon storage are likely to be dense, un-managed stands that are at high risk of catastrophic fire. These types of projects are, in the event of a wildfire, the most likely to have a “stand-replacing fire” outcome in which nearly all trees die and much of the carbon is combusted. Active forest management, like forest thinning or prescribed burning, may reduce wildfire hazard depending on the ecosystem and the goals of the manager (24). However, even the best management may not fully mitigate the risks of climate-enhanced fires. In fire-prone ecosystems, some periodic carbon release is unavoidable.

<u>Drought and water stress</u>

Drought and water stress are driven by irregular precipitation and increasing temperature, both of which are enhanced by climate change (13). Large-scale drought events can cause systemic forest loss. Prominent examples include recent droughts in California (~140 million trees lost) and Texas (~10% of forest cover lost) (14). Like wildfire, drought severity is a function of climate drivers as well as stand structure and species composition. Dense stands create water resource competition that can lead to mortality during drought events. Further, taller trees require more hydraulic pressure to transport water to leaves and thus are more susceptible to water stress-induced mortality (13).

Similar to wildfire, drought risk mitigation is partially at odds with managing for carbon storage. All else equal, the highest risk stands are those with densely packed, tall trees that hold large amounts of carbon. Managing forests for lower carbon density may make that carbon more resilient to reversal by drought (25). Species selection also plays a key role in mitigating risk. For example, evidence suggests that diverse forests may be more resilient to drought (26). Selection of drought-resistant species may significantly mitigate risk, including species that may be native to drier biomes (19).

<u>Biotic agents</u>

Biotic agents include risks like insect outbreaks or diseases that can affect tree growth and survival. Unlike drought and wildfire, biotic disturbances are highly localized and vary depending on climate, species, and geographic region. Still, biotic disturbances are generally expected to increase mortality globally in forests as climate change progresses (13). Biotic agents are typically species- or genus-specific, so homogenous forests, especially those of the same age-class and species, are at higher risk of catastrophic reversals (14). Homogenous stands support rapid growth and reproduction that can lead to biotic infestation of forests. For example, in Northeast US forests, eastern spruce budworm can kill whole stands of spruce and fir trees if they are homogenous but may have relatively little effect on stands where these species are mixed with other species (27). Further, outbreaks can occur on predictable cycles, like spruce budworm (40 years), or irregularly depending on environmental conditions.

<u>Other disturbances</u>

Storms represent another form of natural risk that may increase because of climate change. Wind and ice events, in particular, can have unpredictable and stand-replacing effects (15). These disturbance events are difficult to mitigate, are relatively random, and can affect entire landscapes. For example, a 1938 hurricane in New England uprooted roughly 1000 square miles of forest, redefining the landscape in a way that has persisted to present day.[1] Outside of the realm of natural disturbance, intentional or unintentional human-mediated disturbances may occur in areas with shifting regulations or uncertain land tenure (28). Although the causes for such disturbances are highly localized, such risks can be partially mitigated by clear land tenure laws and clear land management contracts.

<u>Risk mitigation</u>

In the face of increasing disturbance risks, carbon stored in forests should be understood as inherently dynamic. Although carbon storage in forests may reach dynamic equilibrium, climate change may also push previously resilient systems into alternative, lower-carbon stable states (29). Thus, if long-term carbon storage is a goal of forest management, actions may need to be taken to minimize risks of carbon storage reversal. Such actions might be grouped into three broad categories: (a) resistance-building actions that forestall the impacts of disturbances, (b) resilience-building actions that increase an ecosystems ability to return to the desired state after disturbance, and (c) response actions, that help facilitate a transition to a new equilibrium state (19). Simply increasing standing carbon density, which is the explicit objective of many carbon programs, may ultimately distract from management actions that encourage the long-term stability of forest ecosystems and the carbon they hold.

Storing carbon in the geosphere

Another key component of the forest carbon system is the durable storage of carbon in long-lived products, including traditional HWPs. Working forests actively export large amounts of carbon in the form of timber harvesting, a relatively small fraction of which is stored in long-lived HWPs. Even non-working forests may generate flows of harvested carbon in the form of thinning, although the residues from thinning are not always captured for use in products (30). But harvesting of any type can be contentious and the benefits of carbon stored in wood products is contested by academics (5, 9). These studies evince the need for rigorous accounting of the life cycle of HWPs to understand where carbon benefits truly accrue.

Traditional HWPs emit, reduce, and store carbon in various stages of their lifecycle. Carbon stored in biomass is emitted at the stage of harvesting as harvest residues are left to decay or burned, as well as at the mill in the form of masticated or combusted waste biomass. Fossil carbon is also emitted in the harvesting, transport, and production of HWPs in varying amounts depending on the product being made (31). Depending on the product, a small fraction of the carbon stored in a tree is stored in HWPs in use, and the amount of time those HWPs remain in use is highly variable (32). At a product's end-of-life, carbon can be recycled back into HWPs, can be emitted through combustion, or, can be stored long-term in landfills, eventually to be emitted at methane (33). Finally, HWPs also have the potential to substitute for carbon-intensive alternative products, like steel and cement in construction (34). Thus, traditional

[1] https://www.smithsonianmag.com/history/1938-hurricane-revived-new-englands-fall-colors-180964975/

HWPs follow a convoluted carbon journey with diverse carbon storage outcomes depending on production, use, and disposal pathways.

At the same time, numerous new classes of forest carbon uses are emerging in the context of climate change mitigation activity and emerging new markets for carbon removal (35). While traditional HWPs like paper and sawtimber products reasonably well-understood, new forms of carbon storage in products show promise to store carbon for long periods of time, sometimes at the expense of other benefits (e.g., shelter). Some emerging opportunities include the following classes of wood use.

- *Geologic CO2 storage*: CO2 capture and storage (CCS) is the process of injecting carbon dioxide (CO2), captured from an industrial (e.g., steel and cement production) or energy-related source (e.g., a power plant), into deep subsurface rock formations for long-term storage (36). When CO2 captured from biomass is used, the process can be used to remove CO2 nearly-permanently from the atmosphere while also transforming biomass energy into useful forms.
- *Biochar*: While biochar potentially represents a stable, long-term form of carbon storage in soil, physical characteristics of the feedstock and processing steps, as well as environmental factors such as precipitation and soil conditions, strongly influence this stability (37). As a result, there is a large degree of uncertainty in the durability of carbon sequestration in biochar. Further, the yield of biochar from feedstock is relatively low compared to many alternative uses for biomass (38). Still, biochar may represent an attractive low-tech, low-cost opportunity to convert large amounts of biomass carbon to a recalcitrant form for some settings.
- *Engineered structural wood products:* Various forms of merchantable wood, from dimensional lumber to pulpwood, can be used to produce durable construction materials, including cross-laminated timber and wood-fiber insulation boards (39). These products can substitute for conventional construction materials, such as concrete and steel in some architectural applications, displacing associated emissions and storing carbon for decades in buildings, and potentially longer in landfills or other uses (30).
- *Bioliquid injection:* The private sector is beginning to pursue deep geological disposal of bioliquids as an alternative to injection of captured CO2 (35). With the goal of bioliquid disposal, many additional potential conversion approaches can be considered (*e.g.*, direct liquefaction and maximizing production of black liquor). Processes that avoided production of bioliquids as waste can instead be optimized with deep disposal in mind.
- *Biofiber entombment:* Biofibers have been considered optional additions to cement and concrete as means of enhancing their performance, either for strength or durability (35). Addition of microfibers can reduce the total required amount of cement in concrete mixes for construction, with both economic and environmental benefits. Although still at an early stage, these composite materials could potentially store large volumes of carbon as biofiber composites.

- *Biomass burial:* As the value of carbon storage increases, solutions like biomass burial are emerging with the singular purpose of storing carbon. Biomass burial involves the direct burial of biomass, typically wood, in either anaerobic or hypersaline conditions so that the biomass decays more slowly than it would under natural conditions (40). Although biomass burial is often similar to the preservation of HWPs in engineered landfills, it differs both in intention and in the preparation of biomass for storage.

Understanding carbon storage over time in forests and wood products

The remainder of this dissertation focuses on questions of carbon storage, both in disturbance-prone forests and in various wood products. In the first project, I analyze wildfire risk, carbon storage in forests, and innovative uses of low-value wood residues. In the second project, I consider temporal accounting issues in HWPs. And in the third project, I consider the HWP life cycle at a global scale to identify opportunities for climate change mitigation. Although these projects cover a wide breadth of topical and regional emphasis, they are all linked in their shared motivation to understand carbon storage in HWPs as a critical component of forest carbon systems. The highlights from each project are described below.

<u>Project 1: Innovative wood use can enable carbon-beneficial forest management in California</u>

In Project 1, I consider the carbon implications of existing forest management goals and the potential impact of increased use of emerging wood products. Specifically, I assess the State of California's stated goal to treat 1 million acres of forest per year for fire hazard reduction (41), alongside its aggressive commitments to economy-wide carbon neutrality. Some research has suggested that forest treatment is at odds with climate goals (5). Treatments are often costly, and large amounts of low-value wood are often burnt or left to decay. Here, I assess climate and wildfire outcomes across several wood use and forest management scenarios. This analysis applies the FIA BioSum modeling framework (42) to understand management outcomes on California timberland. To understand the biogenic carbon balance of management, I develop a stochastic model to understand how potential wildfire outcomes would manifest under a fire regime consistent with contemporary and probable future fire activity. For each stand and simulated wildfire, I estimate sequestration and biogenic emissions associated with growth, fire, and decay. Finally, I rely on published values to model the cradle-to-grave carbon outcomes for harvested wood across four categories: harvest and transport emissions, production emissions, substitution of carbon-intensive products, and product end-of-life. I find that innovative use of low-value wood enables increased climate benefits and fire hazard reduction. Long-lived wood products have the greatest climate benefits. My results suggest a low-cost pathway to support California's climate adaptation and mitigation goals.

<u>Project 2: Temporal inconsistencies in California's forest carbon offsets protocol</u>

In Project 2, I consider the impact of simplified HWP accounting in a prominent forest carbon offset protocol. In Improved Forest Management (IFM) carbon offset projects registered under the California Air Resources Board Forest Carbon Protocol, carbon offset credits are generated against a baseline which represents counterfactual carbon storage in-forest and in wood products without offset revenue. In theory, this baseline represents "common forest practice" in the region. However, the protocols used to establish baselines are simplistic and likely produce over-crediting. Often, the chosen baseline for in-forest carbon stocks is well below the initial carbon stocks and, as a result, most projects generate a large proportion of lifetime credits in the first year (43). Further, the protocols produce baselines that are static through time. This is problematic because carbon in harvested wood can be estimated using a standard decay function that models the fraction of carbon in a wood product at any given point in time. To simplify the accounting, offset protocols take a single point in time as representative – the average of 100 years of decay. This simplification underestimates the carbon stored in harvested wood products in the counterfactual, resulting in project over-crediting. Using a static baseline underestimates carbon stored in wood products initially and overestimates carbon stored at the end of the project. The result is that the protocol overestimates the avoided carbon emissions resulting from IFM in the near term.

Project 3: Global carbon removal potential of wood products

In Project 3, I expand the aperture of these questions to a global scale and consider the global life cycle of HWPs and the potential for interventions in that life cycle to mitigate climate change. Multiple studies have investigated components of the global HWP life cycle, but none have considered the full life cycle of HWPs from gate to grave for all HWP categories or the potential to capture additional carbon in the global life cycle. In this project, I model carbon emissions and carbon storage for all HWPs at a country scale, including production emissions, the product use phase, and product end-of-life. Following this, I use scenario analysis to understand potential interventions to store carbon or reduce emissions in this life cycle. The most promising of these scenarios is one in which pulp and paper mills are retrofitted with CCS to capture the large flow of biogenic CO2 they already produce. This intervention represents a unique opportunity to capture and store billions of tons of CO2 annually with relatively little need for new biomass feedstock or new infrastructure development. I highlight that this low-cost carbon capture from an existing flow of biogenic CO2 could contribute significantly to IPCC-defined carbon removal goals while creating minimal demand for new biomass feedstock.

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Chapter 1: Innovative wood use can enable carbon-beneficial forest management in California

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Main Manuscript for

Innovative wood use can enable carbon-beneficial forest management in California

Bodie Cabiyo[1], Jeremy S. Fried[2], Brandon M. Collins[3,4], William Stewart[5], Jun Wong[5], Daniel L. Sanchez[5]

[1]Energy and Resources Group, University of California, Berkeley, Berkeley, CA, 94720

[2]Pacific Northwest Research Station, USDA Forest Service, Portland, OR, 97205

[3]Center for Fire Research and Outreach, University of California, Berkeley, Berkeley, California 94720

[4]Pacific Southwest Research Station, USDA Forest Service, Davis, CA, 95618

[5]Department of Environmental Science, Policy, and Management, University of California, Berkeley, Berkeley, CA, 94720

Author Contributions

B.C., B.M.C., and D.L.S. designed research; D.L.S. conceived the study; B.C. and J.W. performed research; J.S.F. and W.S. contributed foundational data and analysis; B.C. analyzed data; and B.C., J.S.F., and D.L.S. wrote the paper with contributions from all authors.

Abstract

Responsible stewardship of temperate forests can address key challenges posed by climate change through sequestering carbon, producing low-carbon products, and mitigating climate risks. Forest thinning and fuels reduction can mitigate climate-related risks like catastrophic wildfire. These treatments are often cost-prohibitive, though, in part because of low demand for low-value wood "residues". Where treatment occurs, this low-value wood is often burned or left to decay, releasing carbon. In this study, we demonstrate that innovative use of low-value wood, with improved potential revenues and carbon benefits, can support economical, carbon-beneficial forest management outcomes in California. With increased demand for wood residues, forest-health-oriented thinning could produce up to 7.3 million (M) oven-dry tonnes (ODT) of forest residues per year, an eight-fold increase over current levels. Increased management and wood use could yield net climate benefits between 6.4-16.9 million tonnes of carbon dioxide equivalent (M tCO_2e) per year when considering impacts from management, wildfire, carbon storage in products, and displacement of fossil carbon-intensive alternatives over a 40-year period. We find that products with durable carbon storage confer the greatest benefits, as well as products that reduce emissions in hard-to-decarbonize sectors like industrial heat. Concurrently, treatment could reduce wildfire hazard on 4.9M ha (12.1M acres), a quarter of which could experience stand-replacing effects without treatment. Our results suggest that innovative wood use can support widespread fire hazard mitigation and reduce net-CO_2 emissions in California.

Significance Statement

Natural carbon sinks can help mitigate climate change, but climate risks – like increased wildfire – threaten forests' capacity to store carbon. California has recently set ambitious forest management goals to reduce these risks. However, management can incur carbon losses because wood residues are often burnt or left to decay. This study applies a systems approach to assess climate change mitigation potential and wildfire outcomes across forest management scenarios and several wood products. We find that innovative use of wood residues supports extensive wildfire hazard reduction and maximizes carbon benefits. Long-lived products that displace carbon-intensive alternatives have the greatest benefits, including wood building products. Our results suggest a low-cost pathway to reduce carbon emissions and support climate adaptation in temperate forests.

Main Text

Introduction

Climate change poses substantial challenges to managing temperate forests, particularly in California (1, 2). Due to extensive timber harvesting and fire exclusion in the 20th century, California forests are younger, denser, and more homogeneous than historical conditions (3, 4).

These changes have left California forests vulnerable to large-scale disturbances like drought, insects, disease, and wildfire. As in other temperate forests, California forests are at risk from increasing fire severity and frequency driven by climate change (5–7). Extreme wildfire events with large proportions of stand-replacing effects have become more common and pose an existential threat to forest ecosystems and their capacity to sequester carbon, particularly on federal lands (1, 8–13).

At the same time, recent work has emphasized the potential of forests to help meet climate goals in the near- and long-term (14–17). Still, tremendous uncertainties exist around aligning forest treatment and climate goals. Estimates of how forest treatment will impact net carbon emissions from temperate forests vary substantially (10, 18, 19). There is broad consensus that more efficient use of harvested wood can improve the carbon balance of management, but different wood products vary substantially depending on production emissions, substitution benefits, and end-of-life emissions (20–23).

In response to increasing wildfire risk, California's Forest Climate Action Team and the State of California have set a goal to reduce wildfire hazard on one million acres (0.4M ha) of public and private forest per year (24). These plans invoke fuel reduction treatments, timber harvest, and expanded use of harvested wood products. Active management – like prescribed fire and mechanical thinning – can mitigate wildfire impacts and provide many co-benefits (25, 26). However, these treatments are often costly even where the sale of larger harvested trees (sawtimber) is possible. Furthermore, fuel treatment effectiveness depends primarily on the removal of small trees, which comprise most of the "ladder" fuels in forests (27). Sale of small trees and residues (e.g. as biomass chips or pulpwood logs) could offset some treatment costs, but present market demand is limited. As a result, large amounts of low-value wood are left to decay or are burned after treatment, releasing stored carbon to the atmosphere. We propose that an alternative fate for this wood may enable expanded treatment and the flexibility to manage for multiple goals.

In this study, we investigate how a robust market for forest residues could affect the scale and impact of forest treatment in California. First, we model forest-health-oriented thinning treatments (Methods) and potential wildfire outcomes (Figure 4) on California's public and private timberland with Forest Inventory and Analysis (FIA) data. We consider three management scenarios: (1) Business as Usual with Limited Management (Low BAU), (2) Business as Usual with Expanded Management (High BAU), and (3) Innovative Wood Products (IWP). In the IWP scenario, the potential revenue generated from innovative wood products supports increased management over either BAU. Second, we examine the carbon benefits of several pathways for harvested wood using attributional lifecycle assessment, including production emissions, carbon storage, substitution of carbon intensive alternatives, and end-of-life emissions (Figure 3.b.). In Figure 3.a., we present the net carbon balance from expanded forest management and wood product markets in California.

Results

Baseline scenarios.

The Low BAU scenario represents a low-management future similar to, but not the same as, current practice in California (See Discussion). In Low BAU, we assume no thinning in both public and private (i.e. family-owned) forests. On corporate-owned land, we model thinning on the 0.8M ha (2M ac) where net revenue is >$2500/ha without revenue from forest residues. Under this management scenario, 1.6M ODT (4.1M m^3) per year of sawtimber are harvested over the next 40 years, on average. For comparison, California produced 3.8M m^3 per year on average over the past decade (13). The Low BAU scenario is characterized by both high fire hazard and high rates of carbon storage in un-treated forest. Accounting for wildfire occurrence and effects via stochastic simulation, this forest land will sequester 0.89±0.02 $tC \cdot ha^{-1}$ annually over the next 40 years (± indicates 95% confidence interval from Monte Carlo simulation). This value is close to previous estimates for temperate coniferous forests in the Western US (e.g. (28)). Direct emissions from fire are 0.40±0.01 $tC \cdot ha^{-1} \cdot y^{-1}$, but post-fire decay increases total emissions by 0.17±0.007 $tC \cdot ha^{-1} \cdot y^{-1}$over 40 years (SI Figure 4). We present alternative formulations of BAU in the SI, with similar results.

In the High BAU scenario, we consider the impact of maximizing the scale of management without subsidy (i.e. where net revenue is positive) and without revenue from forest residues. In this scenario, it is possible to manage 3.3M unique hectares over 40 years (8.1M ac), on both public and private land. The resultant flow of harvested sawtimber is nearly three times larger than in Low BAU, at 5.12M ODT (13M m^3) per year, comparable to historical production volumes (13). Most of this wood comes from trees smaller than 53 cm DBH (Figure 2). In addition, 4.4M ODT of forest residues are technically available in this scenario. Without a price on forest residues sufficient to recoup removal and transport, however, it is likely that this wood would be left to decay or burned in-forest, which we consider below. Where subsidies exist, forest residues may be sent to biopower facilities. Compared to Low BAU, increased management leads to a reduction in wildfire-related emissions: direct emissions from wildfire are 0.32±0.01 $tC \cdot ha^{-1} \cdot y^{-1}$, and decay adds 0.12±0.005 $tC \cdot ha^{-1} \cdot y^{-1}$ over 40 years. While the High BAU scenario reduces wildfire hazard on more hectares than Low BAU, it poses two key challenges: (a) management of stands dominated by small trees can be cost-prohibitive without subsidy and (b) the fate of low-value wood conflicts with climate goals.

Innovative Wood Products scenario.

In the “IWP” scenario, we examine how innovative uses of forest residues can enable better economic and carbon outcomes from management. We assess several products that are commercially and technically mature and have an estimated market size equivalent to >1M ODT wood/year in California (29). We estimate that low-carbon fuel and oriented strand board

(OSB) production can justify delivered forest residue prices in excess of \$100/ODT delivered (Figure S6), similar to other techno-economic analyses (30–32). In IWP, we assume a delivered price of up to \$100/ODT, which supports management beyond what is economically possible in High BAU. Most forest residues are available at lower prices, however (Figure 2).

With this additional revenue, 4.9M ha (12.1M ac) of forest can be managed over the next 40 years without subsidy. Some of this area is treated more than once, so on average ~0.2M ha (~0.5M ac) of forest can be treated each year. Most of this treatment is technically possible in the first two decades (SI Figure 1). We estimate that, at this price, thinning could produce 7.3M ODT of forest residues and 14.8M m^3 (5.7M ODT) of sawtimber annually over 40 years. This would represent a nearly eight-fold increase over current forest residue supply and a four-fold increase in sawtimber production (13, 33). Increased residue prices do not appreciably increase sawtimber harvest: while residue availability increases sharply by 62% with prices up to \$100/ODT, sawtimber availability only increases in the smallest merchantable diameter classes (Figure 2). Even a residue price of \$200/ODT would only increase sawtimber availability by 18% compared to no residue price. At \$100/ODT, a relatively small fraction (40 million ODT, 19%) of forest residues comes from small trees (10-20 cm DBH). Most residue is a byproduct of whole-tree harvest of larger trees and the entirety of trees of non-commercial species.

In IWP, it's possible to treat 1.3M ha (3.1M ac) that could experience stand-replacing wildfire effects (>95% mortality) without treatment, reducing potential basal area mortality by 28±1% on average in those stands (SI Figure 3). Of all area treated, 47% occurs on landscapes designated by CalFire as a high-priority (Zones 4 & 5) for reducing wildfire risk to ecosystem services (Figure 4). Mean annual combustion emissions from wildfire are 0.27±0.01 $tC{\cdot}ha^{-1}{\cdot}y^{-1}$, and post-fire decay adds 0.07±0.005 $tC{\cdot}ha^{-1}{\cdot}y^{-1}$. This represents a reduction in fire-driven emissions of 39% over Low BAU and 19% over High BAU.

Wood products LCA.

For the current mix of California sawtimber end-uses (13), we estimate a net substitution factor of 0.75 tC benefit per tC harvested (tC/tC), where "net" is the sum of production emissions and substitution of carbon-intensive alternatives. This value is slightly higher than estimated for Canada (34) because a larger fraction of timber products in California are used in buildings. It is lower than in similar studies, though, partially because building operational emissions are excluded (34). Wood that displaces steel and concrete has the largest carbon benefits of any use studied here. For this reason, we consider the effect of diverting all additional (vs. Low BAU) sawtimber produced in IWP to multi-family and multi-use buildings. This "IWP+Housing" scenario represents a future in which affordable, medium-density housing is prioritized. In IWP+Housing, the net substitution factor is 1.75 tC/tC because of increased steel and concrete substitution. This value is similar to net substitution factors found for other regions (34, 35), despite our optimistic wood use assumptions. When we include the end-of-life (modeled to 40 years), we find a net carbon benefit of 1.35 tC/tC for all sawtimber products. In the IWP+Housing scenario, it is 2.35 tC/tC.

For forest residue products, carbon benefits vary substantially (Figure 3.b.). Biopower, currently the most common use of forest residues in California, has a low carbon benefit (0.11 tC/tC) relative to more innovative technologies, primarily because of the absence of CO_2 storage and the displacement of relatively clean California grid electricity. Conversely, technologies with a large fraction of carbon storage have the greatest benefits. Biopower with carbon capture and storage has a comparatively high carbon benefit (0.81 tC/tC) because a large portion of the emitted CO_2 is captured and stored (CCS). Hydrogen, GluLam, and OSB have the highest carbon benefits (1.18-1.65 tC/tC) of any of the studied products, because of both high substitution benefits and carbon storage in wood products or via CCS. Further, these three products would all reduce emissions in "hard-to-abate" sectors like cement and industrial heat. While all these residue-based products are technically feasible, they rely on different forest residue components. OSB and GluLam, for example, require small-diameter (pulpwood) logs while hydrogen production can use mixed biomass that includes leaves and bark. In IWP we present an equal mix of only the products that both exceed a 0.5 tC/tC carbon benefit threshold and could use mixed biomass (fuels) or pulpwood logs (OSB, GluLam) at commercial scale (Figure 3.b.).

Net climate impacts.

We estimate the economy-wide net climate impact of management by combining in-forest carbon changes with harvested carbon benefits (Figure 3). Thus, the net carbon balance is a combination of sequestration, storage, emissions, and avoided emissions. In all three scenarios, the forest sector is a net carbon sink. In Low BAU and High BAU, we find similar net carbon benefits of 10.2M and 9.5M tCO_2e per year, respectively, over 40 years. The IWP scenario has a larger carbon benefit of 16.6M tCO_2e per year. In all three scenarios, traditional sawtimber products play an important role in supporting a positive net carbon balance of management. The IWP scenario, though, suggests clear benefits from innovative use of forest residues and sawtimber. In terms of climate goals, shifting from Low BAU to IWP confers a net climate benefit of 6.4M tCO_2e per year, primarily because of innovative forest residue use. Shifting from Low BAU to IWP+Housing yields a higher net benefit of 16.9M tCO_2e per year, largely due to substitution of steel and concrete with sawtimber. On a timescale relevant to California's immediate climate goals (2045), the IWP+Housing Scenario has the most pronounced, immediate benefits (SI Figure 5). In sum, innovative wood use may be critical to achieving California's dual goals of reducing both wildfire hazard and CO_2 emissions.

Discussion

Our results suggest that efficient wood use can play an important role in establishing California's forests as a resilient, long-term carbon sink. We find that innovative wood products would increase the scale of management and the carbon benefits from forest residues that would otherwise decay or be burned. These products can simultaneously advance existing forest management and climate goals in California. Below, we review our results in the context of forest management, innovative wood use technologies, and climate policy. We also highlight that, although this study integrates several critical elements of a complex system, there are important limitations. This analytical framework might serve as a template and a starting point to further investigate the complex interface between wood use and management in high-disturbance forests. Further, large-scale forest treatments like those discussed here may have unforeseen consequences. Investigating ecological outcomes not analyzed in this study, such as the comparative impacts of wildfire and expanded forest treatments on ecosystem services like biodiversity, would be a fruitful area of inquiry to extend this framework.

In this study, we emphasize thinning and surface fuel treatments aligned with California guidelines (Methods). These treatments promote multiple ecosystem benefits and a return to historical forest structure by reducing stand density and retaining the largest, most fire-resistant trees (3, 4, 36). Forest management plans are necessarily context-dependent and will depart, to varying extents, from those we assumed here. It's also likely that future management plans will require novel approaches to respond effectively to climate conditions without historical precedent (37, 38). Management planning may best be conceived as a proactive, adaptable process in order to meet multiple social and ecological goals under changing environmental conditions (37, 38). However, we find that across most timberland in California, carbon beneficial treatment is not feasible without including wood products. Innovative use of wood may be necessary to ensure that wildfire-motivated treatments yield climate benefits. This strategy complements others that emphasize reforestation or prolonged retention of larger trees to aid climate goals (14, 15, 17).

Innovative wood products have two primary value propositions in California: increasing revenues from harvested wood and improving the carbon balance of forest management. Two promising classes of products have emerged in recent reviews: low-carbon and carbon-negative fuels and engineered structural wood products (e.g. mass timber) (29, 32, 39). Low-carbon fuels derived from woody biomass show economic promise because of supportive State and Federal fuel policy, including the California Low-Carbon Fuel Standard. Multiple large-scale transportation fuel projects using California wood and biomass, but located in neighboring states, plan to commission plants in 2021. If additional facilities are instead sited in California, low-carbon and carbon-negative fuels can drive additional regional economic development benefits.

Mass timber products like cross-laminated timber (CLT) and glue-laminated timber (GluLam) are uncommon in the United States but have been widely adopted in European markets. Other engineered wood products like OSB, which can be made from pulpwood logs, are widely used but not produced in the Western US (40). Specific engineered wood products may have relatively higher substitution benefits (e.g., I-beams produced with OSB) or higher carbon storage density (e.g., CLT). These products often displace steel and concrete and would support our IWP+Housing scenario, which has the best net carbon balance of any scenario. The recent inclusion of CLT in California's building code may encourage widespread adoption and production. However, further research should verify the suitability of small-diameter wood, low-quality wood, and California tree species as feedstocks for these products.

In these cases, climate policies can play a critical role. California's Low-Carbon Fuel Standard, for example, provided a financial incentive between \$160-192/tCO_2-abated in 2018-2019, and was recently extended through 2030. Revenue from carbon payment programs like this enable the financial viability of innovative wood use. Similarly, the State has recently adopted other performance-based climate policies, such as Buy Clean California, that could drive use of wood building products. Investment mechanisms, like the new Climate Catalyst Fund, can also play an important role in defraying upfront costs, although these funds will need to grow to support facilities with higher capital costs (e.g. OSB). Those facilities may also require long-term supply contracts to ensure that capital costs will be recovered. Finally, workforce development initiatives could support the rapid scaling of forest treatments. Such policies may help achieve the central goal of the State's Forest Carbon Plan: to firmly establish California's forests as a more resilient and reliable long-term carbon sink (24).

Study limitations.

In our scenario analysis, we suggest Low BAU and High BAU as baseline scenarios. While neither of these scenarios are a perfect representation of reality, we expect that they bracket a range of likely futures without the influence of innovative wood use. We use Low BAU as the basis for comparison because it most closely approximates the current state of forest management in California, with high rates of active management under corporate ownership and much less on public- and family-owned forests. We also consider an alternative BAU that includes a more representative mix of public and private management but does not materially change the results presented here (SI Results). Alternatively, increased interest in wildfire hazard reduction and related policy changes may yield a future more similar to High BAU.

In this study, we have employed an attributional LCA approach, which includes the physical flows to and from a given system. However, it is unlikely that wood harvested in California will be exclusively used in California, and the consequences of an influx of wood products into the global market may have unforeseen outcomes. Localized policy, like California's Green Procurement Strategy, or policy that supports substituting wood products for carbon-intensive alternatives, may promote greater carbon benefits from wood harvested in the state without displacing wood products elsewhere. Although the LCA values used here represent current technology, the carbon benefits of these products may increase or decrease over the modeling period. Substitution benefits may change significantly as the mix and carbon intensity of displaced products evolves.

Predictions of future wildfire occurrence and outcomes are inherently uncertain (41). In our simulations, growth and wildfire emissions vary substantially depending on which forest plots burn and when they burn. This effect is most pronounced in Low BAU, where large amounts of carbon are stored in un-treated forest, but the stability of that carbon is highly uncertain. The values that parameterize decay and combustion have a large effect on wildfire emissions, as well. The parameters we use exclude non-CO_2 climate forcers and may underestimate actual wildfire emissions, limiting the carbon benefits of treatment. Further, we model forest growth in FVS, which is known to underestimate mortality and thus overestimate growth. We do not model the impact of non-fire climate effects like increased incidence of drought, insects, disease, or CO_2 fertilization. Nor do we consider persistent shifts in vegetation (e.g., from timberland to shrubland). In aggregate, we likely overestimate forest carbon stability and underestimate the carbon benefits of forest treatment (1).

Materials and Methods

A full documentation of the methods used to produce this work can be found in the SI Methods. Here, we present a brief summary of those methods.

Management

This analysis applies the FIA BioSum modeling framework (42) to understand management outcomes on California timberland. We rely on data collected from 5,404 field-sampled Forest Inventory and Analysis (FIA) plots between 2005 and 2016 that represent approximately 13.4M ha (33M ac) of California forest land. We refine this forest land sample to limit our analysis to forests that are classified as timberland and as one of four common California forest types: mixed conifer, Douglas-fir, true fir, and ponderosa pine. We consider the three ownership types that account for nearly all of California's timberland: corporate, non-corporate private ("family"), and National Forest System ("public"). We exclude land federally reserved from management and land administered by state- and local-government.

We model forest growth, management, and potential fire outcomes over 40 years with the Forest Vegetation Simulator (FVS) and the associated Fire and Fuels Extension (FFE). For each FIA plot, we simulate five forest treatments (SI Table 1) designed to represent forest restoration-motivated management compatible with the provisions of the Sierra-Nevada Forest Plan. The treatments differ with respect to thinning style (from below or across diameter classes), maximum size of trees allowed to be harvested, and treatment of surface fuels. Each treatment reduces basal area by a maximum of 33%. Treatments implement thinning with a whole-tree harvest system, using either a mechanical harvester (for DBH <53 cm) or manual felling. After thinning, surface fuels are treated with either prescribed fire or lop and scatter. We also simulate a "Grow Only" alternative to represent untreated forest. Subsequently, we evaluate costs and revenues for each treatment using BioSum. Sawtimber values are based on California Board of Equalization rates and, in the IWP scenario, residues have a maximum delivered value of \$100/ODT, although most can be delivered at lower prices (SI Methods). Residues include small trees (DBH <20cm), tops of larger trees, branches, and the entirety of non-commercial species of all sizes. We conduct a multi-criteria optimization in BioSum to choose a treatment that is net-revenue positive, reduces fire hazard, and maximizes live-tree carbon at the end of 40 years. Based on this optimization, BioSum calculates quantities of sawtimber and forest residues that could be delivered to an existing network of processing facilities.

Wildfire modeling

To understand the in-forest carbon balance of management, we stochastically simulate wildfire based on static potential fire outcomes predicted by FVS-FFE. These potential fire outcomes are modeled for each year independently and represent "what-if" fire hazard metrics. We develop a stochastic model to understand how these potential outcomes would manifest under a fire regime consistent with contemporary and probable future fire activity. We run 5,000 Monte

Carlo simulations for each management scenario to reflect the inherent spatial and temporal variability of wildfire. In each simulation, we randomize (a) how many plots burn, (b) which plots burn, and (c) when they burn. We assume a mean annual fire probability of 0.092%, which is slightly higher than historical conditions due to climate change (7, 12). We simulate both 90th and 97.5th percentile fire weather conditions, which are expected to increase in frequency during our modeling period (6, 43). These percentiles are associated with large wildfire occurrence in California forests, which account for an overwhelming majority of total burned area over recent decades (44). In this study, we present a mean of these two fire weather conditions. For each stand and simulated wildfire, we estimate sequestration and emissions associated with growth, fire, and decay. We predict post-fire growth with a scalar function derived from predicted wildfire mortality by basal area and FVS growth projections. We parameterize both direct combustion and post-fire decay with published values (45, 46). These parameters likely understate wildfire-induced emissions and, by extension, the carbon benefits of management (SI Methods).

Lifecycle analysis of wood products

We rely on published values to model the cradle-to-grave carbon benefits for harvested wood across four categories: harvest and transport emissions, production emissions, substitution of carbon-intensive products, and product end-of-life (Figure 1). We consider one tonne of harvested carbon as the primary unit of analysis. For forest residue-based products, we use data from LCA studies with feedstocks and system boundaries similar to what we model here. We only consider products that could use a portion or all the feedstock modeled here (e.g. topwood, but not mixed biomass, for OSB; SI Methods). Where possible, we rely on data from studies using the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model (47). We normalize harvest and transport emissions across all products to be consistent with values used in GREET. Across all product pathways, we assume an average California grid carbon intensity of 225 g CO_2e/kWh (48). The methods and assumptions for each product pathway are described in the SI Methods.

For sawtimber, we adapt the methodology used by (34) to the California market context. This approach yields an economy-wide displacement factor for sawtimber products including emissions from extraction, transportation, and production of a representative suite of building materials. In this study, we retain all values in (34) except for product end uses, which are economy specific. We use historical California-specific end use data instead (13). In the IWP+Housing Scenario, 100% of increased sawtimber supply (over Low BAU) is assumed to displace steel- and concrete-intensive multi-unit buildings, resulting in a larger net substitution factor (SI Table 4). We conservatively assume that 24% of sawtimber is used for biopower and 75% is used in durable wood products, despite more carbon-beneficial uses for sawmill residues (49). We calculate a category-weighted mean half-life for all primary wood products of 38 years (SI Table 4) (50). After primary use, we assume that 65% of retired wood products are sent to landfills, 25% to biopower facilities, and 10% are not collected (49). In the landfill, 90% of wood carbon is assumed to be permanently inert (51, 52), although this assumption has a limited effect over our 40-year modeling period (Figure 3.a.).

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Figures and Tables

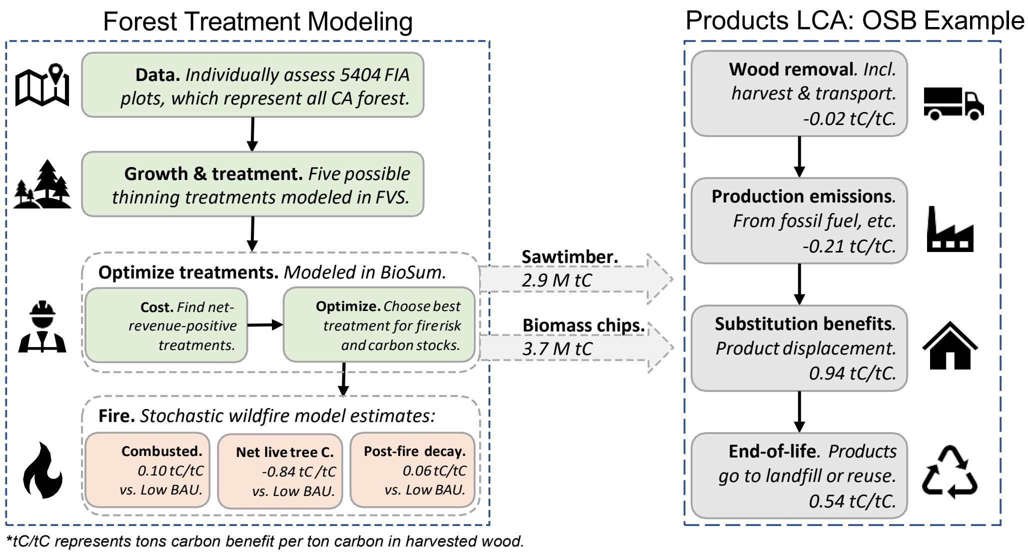

Figure 1. Modeling framework, system boundaries, and example results for one product, oriented strand board (OSB). Product carbon benefits (right side) are specific to OSB, while in-forest carbon fluxes (left side) are common to all products in IWP. Carbon benefit values presented are cumulative over 40 years. See SI Methods for a complete description of all steps above. FIA is Forest Inventory and Analysis; FVS is Forest Vegetation Simulator; BAU is Business As Usual.

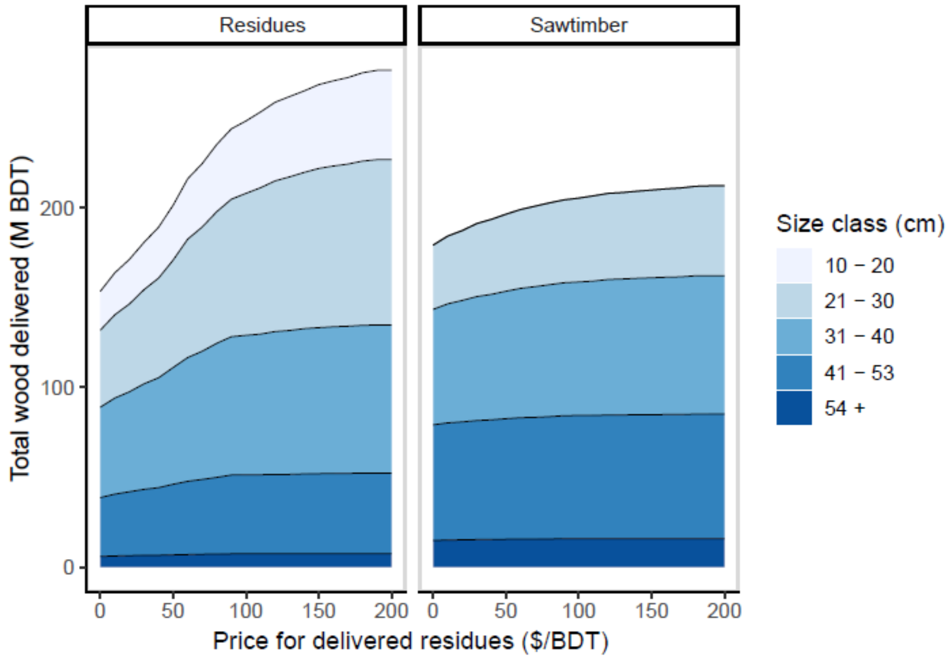

Figure 2. Wood availability at increasing delivered forest residue prices, by diameter at breast height (DBH) class. Residues include tops and branches from larger trees harvested for sawtimber, as well as entire trees of non-commercial species.

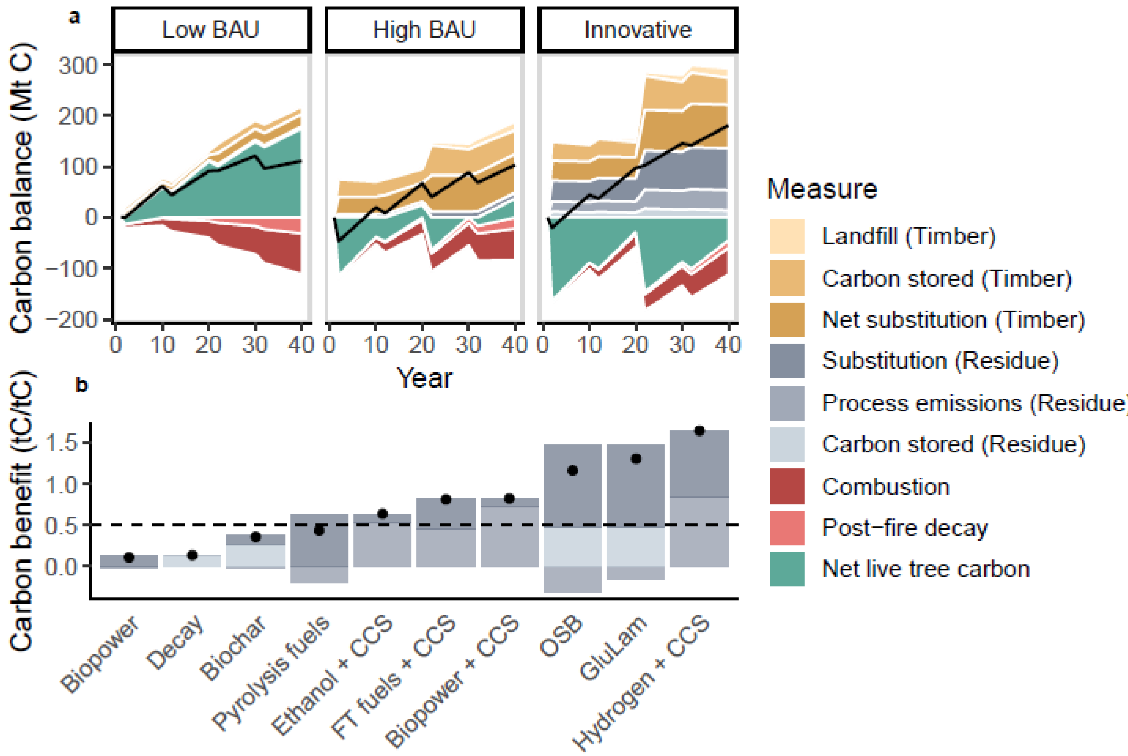

Figure 3. Lifecycle forest carbon balance across (a) three scenarios and (b) several technology pathways. Net carbon values are represented by dots in (b) and black lines in (a). In (b), the dotted line represents the threshold used to select the suite of technologies in IWP. Net live tree carbon values are relative to carbon stocks in year zero, and large decreases are associated with harvest events. In Low BAU, we model management only on corporate land, where potentially profitable (net revenue >$2500/ha). In High BAU, we model management wherever it is net revenue positive with a delivered residue price of $0. In Innovative (IWP), we model management wherever it is net revenue positive with a delivered residue price of up to $100/ODT. Treatment area under IWP defines the study area for High and Low BAU, which include untreated forest.

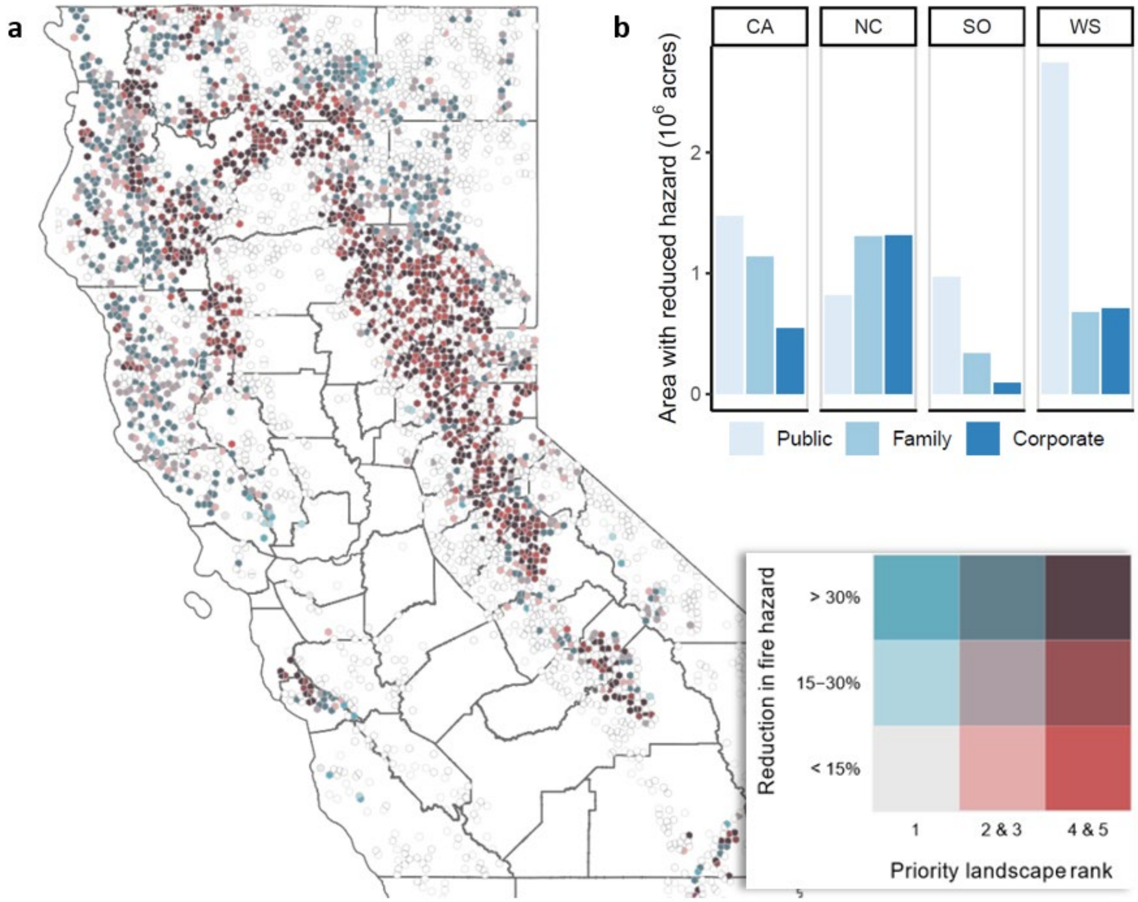

Figure 4. Fire hazard reduction under IWP in (a) CalFire Fire Priority Zones and (b) summed across the study area. Reduction in fire hazard is defined as the basal area mortality fraction with treatment minus the mortality fraction without treatment in the event of a wildfire with severe fire weather. In (b), each hexagon represents a single FIA plot, which is statistically representative of a larger area of forest (usually, ~2000-2500 ha). Empty hexagons represent untreated plots and county boundaries are shown in the background. In (b), values are grouped by FVS Variants, where CA is Central California, NC is North Coast, SO is Northeast California, and WS is Western Sierra. Colors represent ownership groups, where 'Family' is non-corporate private land.

Chapter 2: Inconsistent temporal accounting in California's carbon offsets protocol

The following chapter was completed in collaboration with three other researchers who have given their consent that this material be re-published here. It is intended to be submitted to Nature Climate Change as a Brief Communication. The chapter is a copy of the main manuscript.

Inconsistent temporal accounting in California's carbon offsets protocol

Bodie Cabiyo[1], Daniel L. Sanchez[2,3], Matthew D. Potts[2,3], Barbara Haya[4,5]*

[1]Energy and Resources Group, University of California, Berkeley, CA 94720;
[2]Department of Environmental Science, Policy, and Management, University of California, Berkeley, CA 94720;
[3]Carbon Direct, Inc., New York, NY 10004;
[4]Center for Environmental Public Policy, University of California, Berkeley, CA 94720;
[5]California Institute for Energy and Environment, University of California, Berkeley, CA 94720;
**Author to whom correspondence should be addressed.*

Abstract

Averages taken over time are commonly used to simplify carbon accounting but can introduce significant temporal errors. Here, we analyze two simplifications in the way California's precedent-setting forest carbon offset protocol accounts for wood products, delaying the climate benefits of the program by decades. We find that these simplifications have resulted in the miscrediting of 42.2Mt CO_2e—nearly half of the credits we analyzed—worth $578M at recent market rates.

Main text

Carbon offsets play a central role in climate policy and private-sector climate action but have been widely critiqued[1–6]. The demand for carbon offsets is rapidly growing, particularly in voluntary markets, and multilateral policy agreements like Article 6 of the Paris Accord are likely to spur growth[7,8]. The California Air Resources Board (CARB) compliance carbon market is widely regarded as a model for carbon offsets implementation (e.g., in Washington)[2,9,10]. To date, 196M tons (CO_2e) of carbon offsets have been registered under CARB's offset program, 68% of which are Improved Forest Management (IFM) projects registered under the Forest Projects offset protocol. Offset protocols like this often rely on simplified metrics, like temporal averages, for estimating emissions reductions, in part because they can reduce barriers to entry. However, temporal averages can distort climate benefits in the near-term while being accurate in the long-term.

IFM projects generate offset credits by increasing or maintaining carbon stocks in managed forests—most often through reduced harvesting. In the CARB IFM protocol, credits are generated relative to a "baseline" counterfactual scenario. In most cases, the baseline implies a large initial harvest event (Figure 1.a.). This results in the generation of a large, up-front tranche of credits from "avoided emissions" of onsite forest carbon, minus two deductions: (a) carbon that would have been stored in harvested wood products (HWPs) and (b) market leakage, or harvest activity shifting from the project to other land. The protocol generates credits as if all harvesting to reach the baseline would occur in the first year, but the HWP and market leakage deductions are taken as if harvesting is spread over 100 years. This mismatch creates a distortion between when carbon benefits are credited and when they are realized.

To investigate the impact of this inconsistency, we re-calculate credit generation in the first reporting period (RP_1; typically one year) of CARB IFM projects for which data were available (n = 53). Instead of taking 100-year averages, we apply temporally explicit accounting of baseline harvesting rates (as in ref. 5) and carbon in HWPs. For each CARB IFM project, project developers must model a counterfactual scenario for forest growth and harvesting over a 100-year period. The model outputs for this counterfactual scenario are then averaged over that period to yield two annualized values: (a) carbon in trees standing

on-site ("on-site carbon stocks"), and (b) carbon in trees harvested "Prior to Delivery to a Mill" (PDM). The PDM value is the basis for calculating both market leakage and counterfactual carbon stored in HWPs. The baseline on-site carbon is the value against which offset credits are generated in the first year.

In most projects, the baseline on-site carbon is much lower than the initial carbon stocks (ICS) on the project (37±14% lower, where ± indicates one standard deviation). As a result, these projects generate a large tranche of credits in RP_1 roughly equivalent to the difference between initial carbon stocks and the baseline on-site carbon. This first tranche of credits represents the avoidance of an implied harvest event (Fig. 1.A.). After RP_1, credits are generated as forest carbon stocks grow. Thus, the counterfactual scenario for on-site carbon stocks is functionally distilled into one large initial harvest event, after which forest growth and harvesting are assumed to be in steady state for the remainder of the project lifetime (Fig. 1.A.).

In contradiction, on-site carbon lost through harvesting (PDM) is calculated as an annualized rate, smoothing over the large initial implied harvest event in RP_1 (Fig. 1.B.). As a result, the average PDM value used represents a harvesting rate that is only 6.1±4.8% of the implied harvest in RP_1. This PDM value is then used to calculate market leakage and counterfactual carbon stored in HWPs, which are deducted from the credits generated in each reporting period.

Counterfactual carbon in HWPs is also annualized across a 100-year decay function, which further underestimates the HWP deduction in RP_1 (Fig. 1.C.). This means that each year, carbon flowing into the HWP pool is immediately discounted to its 100-year average value. In other words, when carbon goes into an HWP, a large portion is assumed to instantaneously decay. Carbon stored in HWPs is underestimated for the first ~30-40 years. This leads to over-crediting since most projects claim to reduce production of HWPs.

These simplifications amount to two unique timing issues within the current protocol. The first—Timing Issue 1—corresponds to the annualization of PDM values (Fig. 1.B.). The second—

Timing Issue 2—corresponds the annualization of HWP decay rates (Fig. 1.C.). Both issues lead to overestimating carbon benefits in the beginning of most CARB IFM projects.

Timing Issue 1—averaging counterfactual harvests—results in the largest overestimation of carbon benefits (Fig. 2). Recalculating credit generation for RP_1 without this timing issue resulted in a reduction in offset credits of 33.6 Mt CO_2e in our sample. This represents 37±5% of all IFM credits in our sample, including credits from other RP's. Of these, 17.7 Mt CO_2e credits are associated with the leakage deduction and 15.9 Mt CO_2e with the HWP deduction.

Timing Issue 2—averaging decay of HWPs—is fundamentally dependent on Issue 1. That is, fixing HWP decay assumptions will be inconsequential if the starting HWP deduction is very small. In isolation, Timing Issue 2 only accounts for 0.32 Mt CO_2e in our sample. But calculated together, fixing Timing Issues 1 and 2 reduces credits issued by 42.2 Mt CO_2e. This implies a rate of 46±4% over-crediting in the IFM projects we analyzed (all RP's). Thus, fixing Timing Issue 2 would significantly reduce credits generated in RP_1, but only in concert with Issue 1. Timing Issue 2 would also reduce initial crediting in all other RPs.

Both issues introduce temporal inconsistency in accounting rather than absolute error. Timing Issue 1 would theoretically be paid off over the 100-year lifetime of the projects if carbon continues to be sequestered by the forest. Timing Issue 2 would be paid off after ~30-40 years of counterfactual HWP decay. After this, Timing Issue 2 begins to underestimate carbon benefits as HWPs decay past the mean value (Fig. 1.C.). But timing matters because carbon credits are used to offset fossil fuel emissions that have immediate climate impact. Each year that carbon credits are miscounted represents a year of unmitigated climate impacts from fossil emissions. Temporal miscrediting in the CARB offset program implicitly trades climate impacts today for benefits that accrue decades from now.

Fixing these two issues would substantially reduce the number of credits issued into the California compliance offset market. IFM credits make up 68% of all CARB credits generated to date, and 92% of these have been generated in RP_1. Given the systematic nature of these timing issues and the relative homogeneity of CARB IFM projects, it is possible that a large portion of credits in the CARB compliance market are subject to the issues highlighted here. In our sample alone, these issues account for $578 M of over-crediting at current market prices.

This analysis shows how averaging climate impacts over a 100-year period can create perverse carbon accounting outcomes. Simplification of accounting methodologies has important benefits – namely in making carbon accounting more accessible to practitioners. But where simplifying assumptions are made, carbon accounting methodologies should work to make assumptions conservative. The CARB IFM protocol is one example of how simplifying assumptions can produce meaningful systematic errors, which in turn distort carbon markets, potentially disincentivizing projects with greater near-term climate benefits.

This challenge is not unique to carbon offsets. For example, the widely used Global Warming Potential (GWP) compares the impacts of climate forcers over a standardized time horizon like

100 years. This creates simplified metrics but can obscure and under-value the near-term impacts of short-lived pollutants like methane, hampering decisionmakers' understanding of temporal tradeoffs[11–13]. Alternative metrics that are temporally integrated can provide more transparency, despite some added complexity[11,14]. Similarly, in bioenergy carbon accounting, the choice of using instantaneous versus cumulative metrics can produce conflicting results for the same system[15]. Bioenergy systems often incur upfront carbon debts—due to immediate oxidation of biomass—that are obscured by common cumulative metrics. As with the CARB IFM protocol, these issues highlight the need for careful attention to the temporal aspects of carbon accounting.

The issue analyzed here represents one of several recently highlighted challenges for the CARB IFM protocol. Most projects model a baseline for on-site carbon that converges with the minimum value allowed by the protocol. The way these minimum values are set leads to systematic over-crediting[4]. Further, averaging this baseline over time introduces timing inconsistencies but is impossible to assess with available data. The buffer pool contributions, which are designed to insure against carbon storage reversals, are likely too small to adequately address increasing disturbance risks associated with climate change[16,17]. Finally, uncertain market leakage represents a substantial, unresolved challenge to accurate offset crediting. Currently, market leakage associated with reducing timber harvest is poorly constrained in the literature, with figures ranging from 16.2–88% in the United States, whereas the CARB IFM protocol uses a fixed 20% leakage rate[5,18,19]. Future work should strive to develop robust tools to address these challenges in carbon offsets protocols. Taken together, these issues highlight the critical need for reassessment in a precedent-setting carbon offsets protocol.

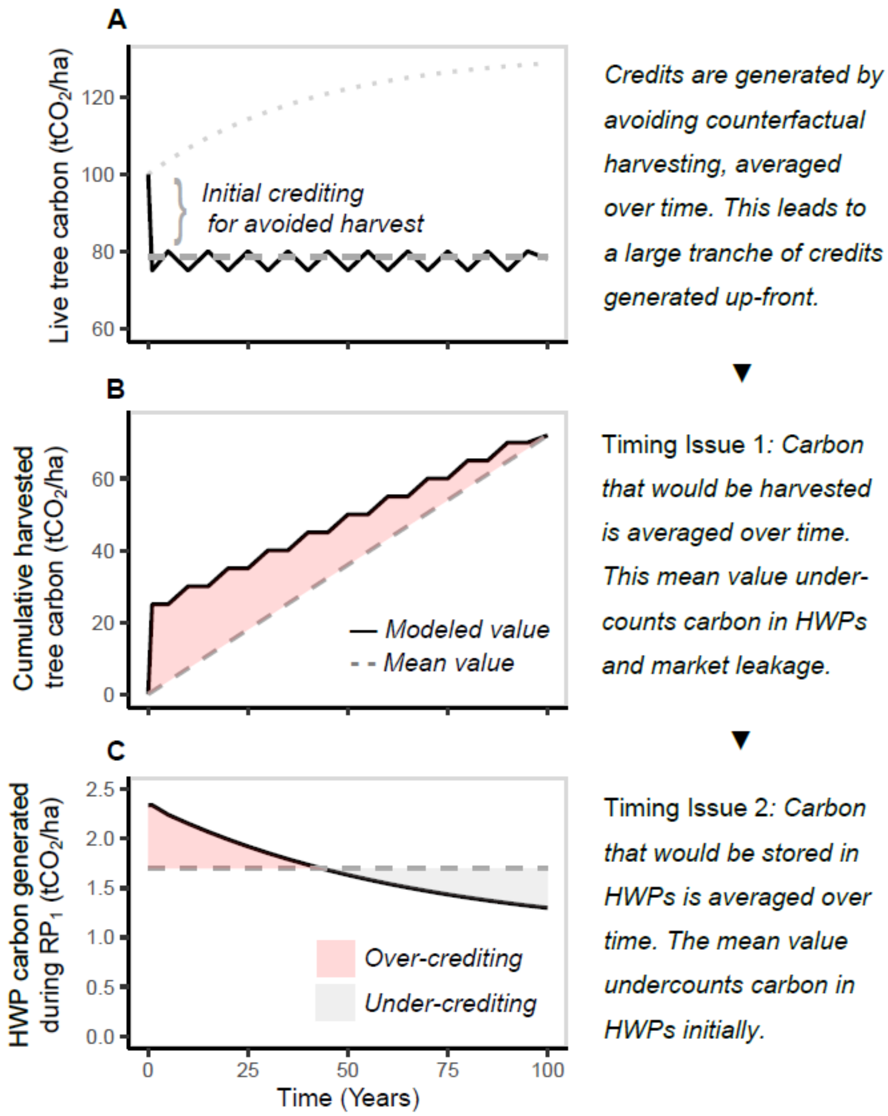

Figure 1. *Differences between modeled counterfactuals (dark lines) and the mean of those models (dashed lines) in a stylized IFM offset project, for on-site carbon stocks (A), cumulative carbon loss from harvesting (PDM) (B), and HWPs (C). Red shading represents the cumulative overestimate of carbon benefits (i.e., over-crediting), which is greatest in RP_1 and shrinks to zero by RP_{100} in (B) and RP_{40} in (C). Grey shading represents a cumulative underestimate. The dotted grey line in (A) represents actual, anticipated forest growth over time without harvesting.*

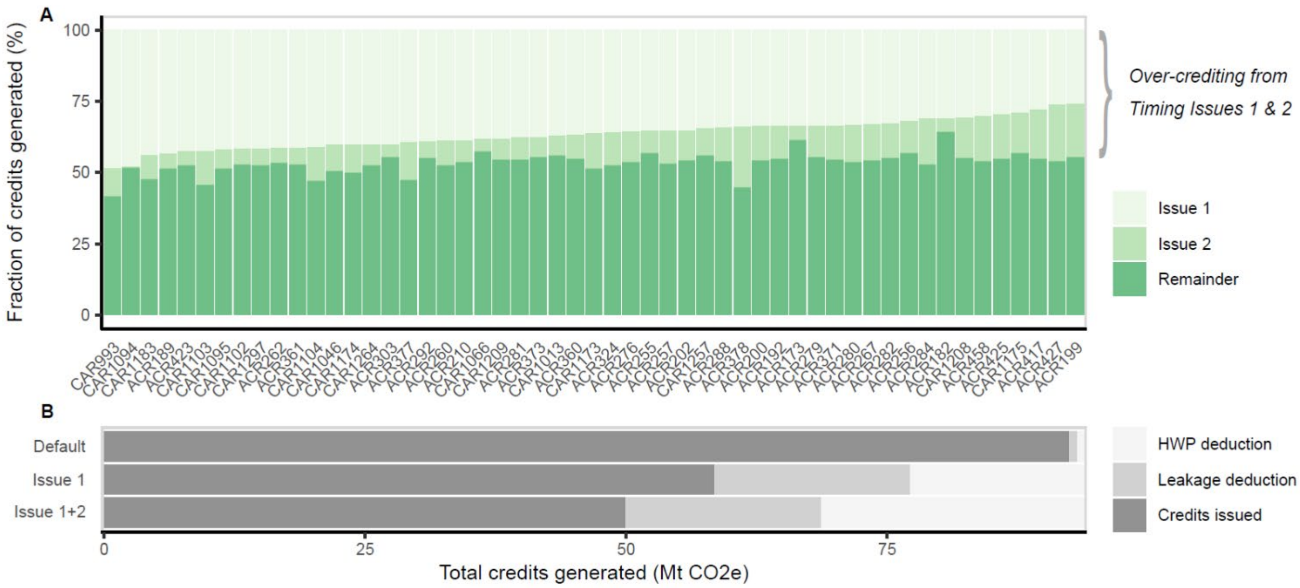

Figure 2. *Credits generated in RP_1 on a relative, project specific (A) and absolute (B) basis under three scenarios: actual credit issuances without adjustments ("Default") and credit generation adjusted to fix Timing Issues 1 and 2. In (A), individual projects are identified by their CARB project code. Timing Issue 1 fixes temporal inconsistencies associated with annualizing harvest rates. Timing Issue 2 fixes the averaging of HWP decay over 100 years. In this figure, Timing Issue 2 is additive to Issue 1.*

Materials and Methods

Data & Validation: We base this analysis on published data (ref. 4) which includes all IFM projects enrolled under the CARB Forest Projects offset protocol for which sufficient public data are available. We supplement this dataset with digitized data collected from official project documents for carbon in trees harvested prior to delivery to a mill (PDM), carbon in trees delivered to a mill (DM), and mill efficiencies. These values are not consistently reported in the project documents, despite being essential for the calculation of market leakage and carbon in HWPs. Where PDM values were not available, we back-calculated PDM from reported market leakage where possible. Similarly, where DM values were not available, we estimated the DM value by applying the average ratio of DM/PDM for all projects for which data were available (0.54±0.08). Where mill efficiency values were not available, we used the mean value from all projects with observations (0.65±0.03). Product fractions were unavailable for most projects, so where projects had multiple mill efficiency values, we used the unweighted mean of all values for the project. To validate these data, we recalculate credits generated using Equation 5.1 in the 2015 CARB US Forest Projects protocol[20]. Our recalculations agree with credits reported by CARB, with minor deviations already reported elsewhere[4].

Estimate of Over-crediting: We build on methods developed in ref. 5. We assume that the functional PDM for RP_1 is equivalent to the implied harvest event, that is, $PDM_1 = ICS - OSCS$ where OCSC is the average baseline on-site carbon stock. We calculate carbon stored in HWP's in RP_1 (HWP_1) in two steps. First, we calculate HWP_1 using the new PDM value with the equation given by CARB: $HWP_1 = (DM_1 - DM_{actual}) * (1 - LR) * ME * F_{mean}$, where LR is the leakage rate, ME is the mill efficiency, and F_{mean} is the 100-year mean carbon stored in HWP's in-use and in-landfill accounting for decay. We estimate DM_1 as $PDM_1 * \frac{DM_0}{PDM_0}$, where DM_0 and PDM_0 are the values assumed in the project. This function assumes that the modeled relationship between DM and PDM is consistent over time. Second, we calculate HWP_1 using the temporally explicit decay functions on a one-year timestep cited in the CARB protocol, rather than taking the 100-year mean[20,21]. We calculate market leakage using a modified form of the formula given by the CARB protocol: $(PDM_1 - PDM_{actual}) * LR$, where PDM_{actual} is the actual harvested carbon in RP_1 (typically 0). To estimate the value of over-crediting, we use weighted mean compliance market prices reported by CARB from 2013-2020 (\$16.81/tCO2e)[22]. For all calculations of actual issuances, we use formulas given in the CARB protocol[20].

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Author contributions:
B.C. designed the project and conducted the analysis. All authors analyzed results and wrote the paper.

Competing interests:
All authors have been or continue to be compensated for performing consultant advisory services for Carbon Direct, Inc., which will generally benefit from a well-functioning carbon market. The authors do not stand to gain financially from any specific outcome of this paper.

Chapter 3: Global carbon removal potential of wood products and mills

The following chapter is intended for eventual publication with multiple contributors, pending potential analytical revisions and additions. This material has not been submitted to any academic journals or published as a preprint as of the time of submission in this dissertation.

The global carbon removal potential of wood products and mills

Abstract

A large portion of carbon sequestered by working forests is harvested and converted into harvested wood products (HWPs) like paper and lumber. Some of this carbon is stored in products or landfills, but much of it is released, alongside fossil fuel carbon, during the production process at the mill. Here, we quantify the lifecycle carbon impacts associated with HWPs globally and investigate the potential for paper recycling and CO_2 capture and storage (CCS) at mills. We find that paper recycling reduces emissions but also reduces carbon stored in landfills. So recycling, on net, only yields climate benefits from reducing landfill methane emissions. We find that the global HWP sector is a net carbon sink in 2020 but, with implementation of CCS at mills could become a sink of 3.3 Gt CO_2/yr by 2050, not counting methane emissions which are uncertain but likely large. Approximately half of this sink would be attributed to baseline carbon storage in products and landfills, and the other half could be realized through CCS. In total, CCS of biogenic CO_2 could reach 1.2-1.8 Gt CO_2/yr by 2050, most of which would be at pulp and paper mills. We conclude that CCS retrofitting of existing HWP mills has significant potential to remove atmospheric carbon at low capital and environmental cost.

Introduction

Forests are increasingly being leveraged for their ability to remove and store carbon from the atmosphere. Estimates suggest that reforestation could likely enable gigatons of CO_2 removal (1, 2). Working, or managed, forests also have the potential to store large amounts of additional carbon, although estimates vary significantly depending on the intervention (3–5). In working forests, though, the flow of carbon harvested annually is a meaningful fraction of the total stock – in some cases, greater than 10% of the total stock is harvested sustainably each year (6). Thus, the flow of carbon from working forests may be just as important as the potential stocks those forests can contain. A fraction of the harvested carbon flow is stored in harvested wood products (HWPs). Correspondingly, the production of HWPs, especially pulp and paper, also creates one of the largest existing streams of point-source biogenic CO_2 that could be captured and stored to augment the carbon removal capacity of working forests. Point-source biogenic CO_2, a concentrated stream of oxidized plant carbon, is central to the implementation of bioenergy with CO_2 capture and storage (BECCS).

Large-scale BECCS may be necessary to reach the most stringent targets set out in the IPCC Sixth Assessment Report (7). BECCS couples biomass feedstocks with engineered CO_2 storage to permanently remove CO_2 from the atmosphere and active terrestrial carbon cycle. Many BECCS technologies are commercially viable today and have the potential to scale rapidly. Numerous projects have either been planned or are actively being developed globally, mostly where there are robust incentives for sequestering carbon (8, 9). One of the most common critiques of

BECCS, however, is that large-scale BECCS deployment under most scenarios would require dedicated energy crops, potentially displacing food crops (10–12). Further, siting and building new BECCS facilities is slow and costly and can pose new environmental and pollution challenges (13). Installing CCS on existing bioenergy facilities may reduce these challenges. For example, CCS retrofits to existing bio-ethanol refineries represents a commercially viable BECCS strategy that would require minimal additional bioenergy feedstock and infrastructure build-out (14). This type of CCS-retrofit solution to capture an existing flow of biogenic CO_2 is also possible at many wood mills globally, which typically provide a large portion of their own heat and power (15–17).

Several recent studies have quantified aspects of the climate mitigation potential of HWPs and mills. Localized studies have estimated the potential for CCS on pulp and paper mills, focusing on the technoeconomic feasibility and potential scale (15–17). In the United States, the annual technoeconomic potential for CCS on pulp and paper mills was estimated to be 150 Mt, of which 77% is biogenic; in Europe, the estimated technical potential is 69 Mt/yr, of which 90% is biogenic (15, 18). This concept has been tested at pilot scale, as well: the Wallula Basin Pilot Project, an early pilot of a pulp mill CCS retrofit, was shown to be economically viable at modest carbon prices (e.g., \$15/t CO_2) and capable of permanently sequestering 1 Mt of biogenic CO_2 per year (19, 20).

To our knowledge, there have been no studies of the global potential for CCS on existing pulp and paper mills and no studies have quantified the potential of CCS on sawmills at any scale. Multiple recent studies have, however, conducted lifecycle carbon accounting of the paper industry at a globally aggregated scale, including emissions from production, use, and end-of-life (21–23). Some studies have investigated the global life cycle of sawtimber products and identified a modest carbon sink in in-use products (24), but none to our knowledge quantify production emissions or the end-of-life of the product lifecycle (e.g., storage in landfills), both of which are significant.

In aggregate, these studies suggest meaningful opportunities for and contributions to climate mitigation in the HWP lifecycle, but do not give a complete picture of the global carbon removal potential of HWPs. Here, we combine multiple components of the global HWP lifecycle to understand the process emissions associated with HWPs, the carbon stored in HWPs, and the total potential for capturing point-source CO_2 from mills. Further, we disaggregate estimates of these measures across multiple energy feedstock inputs and product types at the country scale. This system-level, global assessment of these factors allows for robust comparison between climate change mitigation opportunities in the HWP lifecycle. We assess scenarios with two key opportunities that have been identified in the literature: increased recycling of paper products and CCS at HWP mills. Both opportunities have been noted to potentially shift the carbon balance of the HWP life cycle (15, 17, 21).

Results

Our analysis considers the global gate-to-grave climate impact of HWPs, including those potentially produced with CCS, from 2015 through 2065, including both sawtimber and paper products, across multiple carbon management scenarios. We find that sawtimber products store the largest amount of carbon annually, but paper products have the greatest climate mitigation potential through the capture of emitted CO_2 at pulp mills. This is partly because a large fraction of the CO_2 emitted from pulp mills is biogenic, particularly for products sourced from chemical pulping. We identify installation of CCS on existing mills as a promising carbon-removal pathway that requires minimal new feedstock and infrastructure.

Production emissions

We estimate the gate-to-gate process emissions for producing HWPs across several product categories (Figure 1). Total process emissions from paper production in 2020 is estimated to be 1243 Mt CO_2. Biogenic CO_2 comprises 54% (659 Mt CO_2) of this and is the largest CO_2 source in the paper lifecycle. This fraction is lower than reported for the US, but this difference likely reflects a higher proportion of mechanical and recycled pulping processes globally than in the US (15). Electricity is the second largest emission source at 295 MtCO_2/yr (24%). The highest-emitting product category is "Other paper", which is broadly defined by FAOSTAT and predominantly includes packaging papers. In 2020, "Other paper" produced in China represents the largest single product group reported by FAO.

Total process emissions for sawtimber products in 2020 was 536 Mt CO_2, of which 112 Mt (21%) are biogenic (Figure 1). Electricity accounts for 60% (312 Mt CO_2), the largest fraction of emissions from sawtimber products. A disproportionate amount of total process emissions are from fibreboard – particularly high- and medium-density fibreboard – due to the large amount of energy required per unit of product. In contrast, sawnwood makes up the largest portion of all sawtimber products globally (55%) but only accounts for a third of the total emissions because of relatively low energy required.

By 2050 under SSP2, total production emissions associated with pulp and paper are expected to climb to 1704 Mt CO_2/yr, assuming no changes to production processes or energy inputs (Figure 2). Similarly, production emissions for sawtimber products may climb to 814 Mt CO_2/yr. While there are several opportunities to reduce the climate impact of producing HWPs, the large fraction of emissions from electricity use for both sawtimber (60%) and paper (25%) suggests that the carbon footprint of producing HWPs can decrease significantly if the carbon intensity of electricity production declines, as has occurred in the US and the European Union (25).

A large portion of the emissions from producing HWPs are associated with two high-producing countries: China and the US. China accounts for the largest portion of production emissions for both sawtimber (257 Mt CO_2/yr) and paper (495 Mt CO_2/yr) products. China is also the largest producer of HWPs, but high production emissions are also a product of a relatively carbon intensive grid (550 gCO_2/kWh). The US accounts for the next largest fraction of global pulp and paper emissions, producing 147 Mt CO_2/yr, similar to figures reported elsewhere (15). (The US only emits 51 Mt CO_2/yr from the production of sawtimber products.) By 2065, under SSP2, Germany and India may displace the US as the second- and third-largest pulp and paper emitters after China.

Carbon storage in products

We consider the full use phase and end-of-life for each HWP category separately. Because each phase is defined by a temporal function, the fraction of carbon stored or emitted in each phase varies over time. For simplicity, we present values from 20 years post-production of HWPs as an approximate mid-point value across product categories (Figure 3).

Total storage in in-use products is modeled as a function of the half-life of IPCC-defined product classes (paper, sawnwood, and panels). Paper only has a half-life of 2 years, so most paper quickly transitions to the end-of-life phase. Only 1 Mt CO_2/yr is stored for at least twenty years in in-use paper products globally, although this simple half-life model may underestimate the lifetime of some paper products (e.g., printing paper). Sawnwood and panels both have substantially longer half-lives (Table 4), so a larger fraction of the carbon in these HWPs remains stored over decadal timescales. Carbon storage in in-use sawtimber products in 2020 is 693 Mt CO_2/yr, far outweighing the emissions from producing those products.

The end-of-life phase for HWPs is more complex and less well-understood in the academic literature. Broadly, this phase involves two critical steps: allocating the fate of each HWP after its useful life and modeling decay in landfills, including the evolution of methane due to anaerobic conditions (Methods). IPCC methods are poorly aligned with recent literature on decay of HWPs in landfills, particularly with regard to the amount of carbon expected to remain fully inert (26, 27). This inert fraction, called the Carbon Storage Factor (CSF), has been studied in lab settings and the field (26, 28, 29). Excluding CSFs in our model significantly reduces carbon stored in paper products but not sawtimber products (Figure 3). For paper products, relatively little carbon would be stored in landfills after 20 years without a CSF. With a CSF, 527 Mt CO_2 is stored in landfills in 2020 in the form of inert paper carbon. For sawtimber products, the CSF has relatively little impact on storage of carbon in the first 20 years, because they have relatively long use half-lives and decay slowly relative to paper.

Methane emissions from landfills represent a potentially large but poorly defined emissions category in the HWP life cycle (Discussion). Reported as an annual flow in year 20 post-production, methane emissions account for 73.5 Mt CO_2e/yr, similar to other studies (21). Most of this methane (96%) is associated with decay of paper products. Though annual flow of methane is a commonly used metric, it is difficult to compare to other emissions in the HWP lifecycle because each given unit of HWP will generate annual methane emissions for decades, whereas process emissions occur only at the beginning of the life cycle. Thus, cumulative methane emissions is likely a more comparable measure to the other emissions in the HWP life cycle. By 2040, we estimate that HWPs produced in 2020 will have generated 1.5 Gt CO_2e from methane evolution alone (uncertainty range: 1.1-1.8 Gt CO_2e). We do not present methane emissions in aggregated results because of the discrepancy in temporal accounting, but they constitute a significant opportunity to reduce emissions from the HWP life cycle.

Scenario analysis

Baseline scenarios

Baseline carbon removal potential is limited to carbon stored long-term in in-use products or in landfills. In 2050, the total carbon stored in HWPs on an annual basis is 1.6 Gt CO_2/yr, two-thirds of which is sawtimber products (Figure 4). Not counting biogenic emissions and methane, the total carbon footprint in the Baseline is 1.4 Gt CO_2/yr. However, this carbon sink is offset by large cumulative methane emissions: by 2070, HWPs produced in 2050 will have emitted 2.0 Gt CO_2e, significantly more than any other stage of the HWP life cycle (Figure 5).

The Recycling scenario envisions paper waste recycling at the theoretical maximum (Methods). Recycled pulping is more efficient than chemical pulping, resulting in lower emissions overall. At the same time, paper is diverted from landfills, resulting in less carbon storage. In 2050, carbon stored in landfills is reduced by 0.45 Gt CO_2/yr, diminishing the HWP sink. Slightly offsetting this effect, fossil emissions from paper production are reduced by 0.1 Gt CO_2/yr over the Baseline scenario (Figure 4). The Recycling scenario does substantially reduce methane emissions from landfills, in alignment with the reduced carbon storage in landfills. After 20 years, HWPs produced in 2050 will have emitted methane equivalent to 0.6 Gt CO_2e, 1.4 Gt CO_2e less than in the Base scenario (Figure 5). This represents largest carbon benefit of recycling within the production, use, and end-of-life stages of the pulp and paper life cycle.

CCS scenarios

Despite substantial differences in material flows and industrial processes between the two BECCS scenarios (Recycling and Baseline), both yield similar climate mitigation potentials in 2050, although carbon is stored in different reservoirs in each scenario. We report 2050 results for scenarios because of both the time required for industrial transitions and the centrality of 2050 in global climate negotiations.

In the Baseline + CCS scenario, we model CCS that captures 90% of CO_2 emitted from all mills globally. In this scenario, the total amount of carbon storage would be 3.3 Gt CO_2/yr, and about half (1.6 Gt CO_2/yr) is carbon stored in in-use HWPs and landfills and half is captured through CCS (1.7 Gt CO_2/yr). Biogenic CCS accounts for most of the CCS portion at 1.2 Gt CO_2/yr, with over 80% coming from pulp and paper mills. At 2020 levels of production, it would be possible to capture 1.2 Gt CO_2/yr (0.8 Gt biogenic), also with over 80% coming from pulp and paper mills. Across all time periods, China and the United States have the largest carbon removal potential, but Germany, India, and Japan also have significant potential in 2050 (Figure 6).

In the most aggressive industry-change scenario, Recycling + CCS, 3.4 Gt CO_2/yr would be sequestered in 2050, of which 1.8 Gt CO_2/yr would be capture of biogenic CO_2. Of this, 0.92 Gt CO_2/yr would come from feedstock freed by reduced demand for virgin pulpwood. These benefits are augmented by a large reduction in methane emitted from landfills (above). Although both CCS scenarios have similar net climate outcomes, they rely on different forms of carbon storage. The Baseline + CCS scenario relies on both geologic CO_2 storage and landfilled HWP carbon, although the longevity of the latter is poorly understood (e.g., see CSF discussion above). The Recycling + CCS scenario relies heavily on geologic storage of captured CO_2, including a large portion of combusted virgin feedstock. It is also possible that this feedstock could have uses with greater carbon benefits, such as for mass timber products or low-carbon biofuels like green hydrogen (30).

Discussion

Our analysis of the global HWP life cycle reveals meaningful existing and potential contributions to mitigating climate change. In the baseline scenario, we find a larger annual sink of carbon physically stored in HWPs than previously reported. We model two dramatic shifts in material use in the HWP system and find that, while increased recycling brings relatively limited climate benefits, installation of CCS on existing mills represents a large and little-acknowledged carbon removal opportunity. We find that the global HWP system could account for up to 1.7 Gt CO_2/yr of CCS of biogenic carbon, constituting additional net carbon removal. This is augmented by CCS of non-biogenic CO2, carbon stored in HWPs, and potential landfill methane capture. These values suggest that carbon removal opportunities in the HWP system can significantly augment opportunities to increase carbon stocks in working forests, which are frequently the focus of academic research (3, 31).

The greatest opportunity we identify is CCS retrofitting to capture CO_2 from existing pulp and paper mills. Several studies have verified the technical and economic viability of such an approach and quantified its local potential (15, 17, 18, 32, 33). There are at least two unique benefits to this approach. First, it greatly reduces the need for new bioenergy feedstock required to generate energy and CO_2 for CCS. Second, it can reduce the capital and environmental costs associated with building bioenergy facilities from the ground up. Both of these benefits ameliorate significant concerns raised about new BECCS development (12).

The viability of such a strategy depends on several technical and economic factors. Critically, the economics of CCS directly depends on a market for carbon removal. Multiple studies have estimated the marginal cost of CCS on Kraft pulp mills in the range of ~$20-100 / t$CO_2$ captured and stored, depending on the capture technology and mill configuration, like available heat (32). While prices in this range are increasingly viable in various voluntary and compliance carbon markets, access to these markets may be uneven and location dependent. Important factors that can affect prices include access to waste heat and additional biomass, purity and availability of biogenic CO_2 for capture, and access to CO_2 storage infrastructure. CCS retrofitting on existing mills will often be cheaper than building new BECCS facilities from the ground up, but retrofitting costs may be prohibitively expensive in some cases where new boilers, new biomass feedstock, or extensive new CO_2 transport infrastructure are required (32).

Kraft pulp and paper mills (which employ chemical pulping) have been emphasized in the CCS literature to date because of a high concentration of biogenic CO_2 produced from combustion of black liquor in the recovery boiler. We find that the large majority of capturable biogenic CO_2 is associated with chemical pulping. Conversely, CCS on recycled paper mills and sawmills is less well-understood and smaller in total magnitude but may have significant local opportunities where mills produce enough biogenic CO_2 to justify CCS retrofitting. In all types of mills, CCS opportunities are likely limited by access to CO_2 storage infrastructure, which can as much as double storage costs (16). In some regions, like northern Europe and the southeast US, CCS hub infrastructure is already in development (8, 9). Such infrastructure is also a function of the distance to geologic storage reservoirs. Although we considered absolute storage opportunities at the country level in this study, access to such storage depends on distance to storage, the economics of CO_2 transport, and the availability of infrastructure. Future research should consider these questions at a more granular spatial scale.

This analysis substantially extends two recent studies on the global lifecycle carbon impacts of HWPs. The first study used IPCC-defined methods to estimate the amount of carbon stored in in-use HWPs reported by FAOSTAT, including both sawtimber and paper products (24). The authors find similar (335 vs. 492 $MtCO_2$/yr) values to those in this study. They do not, however, include any process emissions estimates nor do they consider the end-of-life of HWPs, both of which are critical components of the HWP lifecycle. The second study relied on globally aggregated data to model the gate-to-grave carbon impacts of paper products under multiple scenarios (21). The authors found similar results to those reported in this analysis regarding process emissions and the limited climate benefits associated with a radical transition to paper recycling. For example, we find 594 $MtCO_2$/yr for non-biogenic emissions while van Ewijk et al. (2020) find 648 Mt CO_2/yr. This difference is partly explained by our use of country-scale grid intensities rather than a single global grid CI. Van Ewijk et al. did not consider sawtimber products or CCS potentials, but the detailed energy and material process modeling underlying their study substantially informed the simplified pulp and paper model developed here (22, 23).

There are multiple important limitations to this work. Wherever possible, we have used country- and product-specific data but, in many cases, data do not exist at that level of granularity. For example, we use a single half-life value to describe the use phase for multiple types of wood panels across 140 countries because better data do not exist. This lack of data obscures important nuances, like the difference between the use of fibreboard in short-lived furniture vs. plywood used to sheath long-lived buildings. Similar coarse data had to be used to represent methane evolution and capture, post-use product trajectories, best-available technology SEC values, and others. Further, the FAOSTAT and IEA global databases we relied on have missing and erroneous observations, despite being the only reliable global, country-level statistics on HWP production and energy consumption (see Methods). This gate-to-grave study also excludes in-forest dynamics, which is a critical part of the HWP system. Unfortunately, global data on working forests is scarce, inconsistent, and difficult to link to HWP datasets like FAOSTAT. Future extensions of this work will leverage emerging global forest data to include spatial analysis of the in-forest dynamics linked with the global HWP life cycle.

Methods

General approach

We model the global carbon lifecycle of several classes of paper and sawtimber HWPs, including the production, use, and end-of-life stages of each product, country, and year from 2016-2065 (projected). This analysis is based on HWP production data from the Food and Agriculture Organization of the United Nations (FAOSTAT) and the International Energy Agency (IEA), the only reliable HWP data available on a global, country-specific scale. FAOSTAT reports HWP trade statistics annually for each country across several HWP categories. The IEA World Energy Balances reports country-level total energy consumption by energy source across numerous industries, including sawtimber products and paper products. We integrate these global data sets and augment them with product- and lifecycle stage-specific variables from peer-reviewed studies. Finally, we consider four prospective carbon use scenarios: baseline HWP production and use, maximized recycling of paper products, and capture and storage (CCS) of point-source CO_2 in both the baseline and recycling case.

Demand projections

To represent future production of HWPs, we use published HWP category-wide projections derived from the Global Wood Products Model through 2065 under the three SSP scenarios (SSP2, SSP4, SSP5) that capture the widest range of variability in HWP production (34). These data are the only projections available that are specific to both country and HWP category (i.e., panels, paper, sawtimber). Some granularity is lost by grouping product types by category, particularly for paper products that have significantly different demand trajectories (21). However, aggregated projections for paper demand align well between studies, even if product-specific trajectories are erased. We map category-level demand projections to specific products modeled here by recalculating demand as a fraction of reference-year consumption (2015). For some countries with low levels of HWP production, the year 2015 has zero reported production. Where production for 2015 is reported as zero, we use 2020 as a reference year. We map these values to the mean reference period of 2015-2020 of FAO data for each product and country.

Process emissions

Broadly, process emissions are calculated as the product of the total units of HWP production, the specific energy consumption (SEC) per unit of production, and the carbon intensity of energy used. For sawtimber products, the emissions for a given product (E_j) are calculated by summing emissions across each energy source, i:

$$E_j = \sum_i P_j * SEC_j * REF_i * CI_i$$

Where production (P_j) is reported by FAOSTAT each country and year and REF_i, the relative emission factor for each energy source, is calculated from gross energy consumption reported by IEA. CI_i is the carbon intensity for energy source i. SEC values for each sawtimber product are calculated from (35) and are given in Table 1.

Table 1. Specific energy consumption for several categories of sawtimber product. Values are mean values where multiple studies are reported in (35)*. In all cases, SD values for fibreboard are high due to one high outlying value. MDF/HDF is medium- / high-density fibreboard. Other fibreboard includes the observations for all fibreboard types.*

Product	SEC (GJ/t)	SD (GJ/t)	Observations
Sawnwood, non-coniferous	6.1	0.3	3
Sawnwood, coniferous	3.3	0.1	4
Plywood	5.8	1.8	4
Particle board	10.6	5.9	3
MDF/HDF	12.1	7.3	4
OSB	5.0	1.9	4
Hardboard	13.5	10.0	3
Other fibreboard	10.0	7.5	14

We use a similar calculation for pulp and paper. However, calculation of process emissions from pulp and paper production is complicated by the presence of multiple pulping processes that contribute to each product category. To incorporate this variation, we calculate fractional product allocations for each observation into each pulping class based on Table 2. We then calculate process emissions based on the SEC values for each product and each pulping process (36). Further, in the case of chemical pulping, the recovery boiler generates a large portion of the process energy required but is inconsistently accounted for by the IEA. For some countries with large chemical pulping industries, IEA reports that no biomass-based energy is used in the pulp and paper industry (e.g., China).[1] Instead, we ignore IEA's biomass energy category and calculate biomass energy directly as a function of paper production. We calculate biomass inputs based on the allocation of products to each pulping class (Table 2). For chemical pulping, we assume 24% of total energy is supplied by solid biomass fuel, as in the US (15). For mechanical and recycled pulping, this fraction is 0. We calculate the energy recovered as black liquor on a mass basis for both chemical and recycled pulping using production efficiency values compiled in (22) and lower heating values for mill waste reported in (21).

Table 2. Product allocations across three pulping types. Recycling allocation is calculated iteratively to achieve a total recycled input rate (RIR) of 0.70 for the Recycling scenario.

Pulping type	Product	Allocation	Recycling allocation
Recycled pulping	Newsprint	0.68	0.95

[1] This problem may also be true of sawmills where IEA reports no biomass-based energy use at the country level, but there are insufficient data to accurately calculate an alternative estimate of biomass energy inputs.

Recycled pulping	Printing	0.08	0.27
Recycled pulping	Sanitary and household	0.34	0.7
Recycled pulping	Packaging	0.56	0.95
Recycled pulping	Other paper	0.27	0.63
Chemical pulping	Newsprint	0	0
Chemical pulping	Printing	0.62	0.49
Chemical pulping	Sanitary and household	0.66	0.3
Chemical pulping	Packaging	0.22	0.03
Chemical pulping	Other paper	0.51	0.26
Mechanical pulping	Newsprint	0.22	0.03
Mechanical pulping	Printing	0	0
Mechanical pulping	Sanitary and household	0	0
Mechanical pulping	Packaging	0.11	0.01
Mechanical pulping	Other paper	0	0
Non-fibrous	Newsprint	0.1	0.02
Non-fibrous	Printing	0.3	0.24
Non-fibrous	Sanitary and household	0	0
Non-fibrous	Packaging	0.1	0.01
Non-fibrous	Other paper	0.23	0.12

We use carbon intensities reported by IPCC for fossil fuel energy sources and calculate the carbon intensity of biomass based on a lower heating value of 16.8 MJ/kg (Table 3). We use country-level grid intensity values from 2020 (37). Finally, we assume that "Heat", the broadly defined fifth energy category reported by IEA, has the same carbon intensity as the grid for each country. This assumption is likely inappropriate for the few countries with a high proportion of renewable energy in the grid mix, but these countries produce a small fraction of global HWP production.

Table 3. Carbon intensities for energy inputs during the HWP production phase.

Energy source	**Carbon intensity (kg CO_2/GJ)**	**Source**
Coal	95	IPCC
Oil products	77	IPCC
Natural gas	56	IPCC
Biomass	109	Calculated
Electricity	Country-specific	Our World in Data (2022)

Embodied carbon

There are limited product-specific data available for the use characteristics of HWPs, so we rely on IPCC HWP class-scale values to define half-lives for products in use, shown in Table 4 (38). End-of-life trajectories can vary significantly across countries, but no country-level data are available, so we use estimated global parameters. We assume that 90% of sawtimber products go to landfill after their useful life, while the remaining 10% is immediately oxidized (39). Similarly, we assume 47% of paper products go to landfill after their useful life, with the remainder being recycled (22).

We used decay rates (*k*), carbon storage factors (CSF), and methane production rates from studies where product classes were explicitly reported and roughly matched the product classes used by FAOSTAT (Table 4). These values were compiled from a series of field and lab studies on landfill decomposition of organic waste primarily conducted in the US and Australia. Unfortunately, decay values representative of variance across climates are not available, although temperature is likely a key driver of landfill decay (28). There is limited empirical work in this area, so many studies and models rely on rough assumptions (40). Methane evolution rates are available from accelerated-decay lab studies (Wang 2011), but these lab rates are difficult to translate to expected rates in the field. For this reason, we assume that 50% of decayed carbon is released as methane, and 25% of that methane is captured and fully oxidized (21). We use the 100-year global warming potential for methane (27.9). We vary these two assumptions in aggregate by +/- 12.5% to estimate a likely range of methane emissions, although the variance could be greater (Figure 5). We use the 100-year global warming potential for methane (27.9).

Decay of carbon in landfill is defined by the following function, where “C in use” is the fraction of in-use products at time t on a carbon mass basis:

$$C\ stored\ (t) = (1 - C\ in\ use(t)) * [(1 - CSF) * e^{-kt} + CSF]$$

Many studies do not attempt to quantify carbon stored permanently in landfills, despite a large body of research suggesting the much of the carbon in landfilled-HWPs remains inert, particularly for sawtimber products (26, 28, 29). Shorter-lived HWPs, in particular, are sensitive to the choice of CSF, so we also consider the bounding case in which the CSF is 0.

Table 4. Parameters used for end-of-life calculations for each product. Methane evolution rates are indicative of variability and are not used as model parameters. MDF/HDF is medium- / high-density fibreboard.

Item	In-use half life	Decay rate	CSF (gC/ gProduct)	Methane (mL/ gProduct)	Source
Sawnwood, non-coniferous	35	0.007	0.42	16.1	(27, 40, 41)
Sawnwood, coniferous	35	0.007	0.42	4	(27, 40, 41)
Plywood	25	0.007	0.44	6.3	(27, 40, 41)
Particle board	25	0.007	0.41	5.6	(27, 40, 41)
MDF/HDF	25	0.007	0.4	4.6	(27, 40, 41)
OSB	25	0.007	0.35	42.2	(27, 40, 41)
Hardboard	25	0.007	0.41	5.6	Assumed same as particle board
Other fibreboard	25	0.007	0.41	5.6	Assumed same as particle board
Newsprint	2	0.017	0.25	74.3	(27, 29, 40)
Printing and writing	2	0.015	0.06	84.4	(27, 29, 40)
Sanitary and household	2	0.066	0	217.3	(27, 29, 40)
Packaging	2	0.061	0.25	152.3	(27, 29, 40)
Other paper	2	0.061	0.25	132.1	Assumed same as packaging
Printing	2	0.015	0.06	84.4	(27, 29, 40)

CCS scenario

To model CCS retrofitting on existing mills, we rely on multiple studies that have shown the technical and economic feasibility of such retrofits (15–17). We assume a heat requirement of 37.4 kJ/mol CO_2 captured (15). This is an aggregated value that reflects variable capture requirements for the different boilers, due to varied CO_2 concentration from each. For simplicity, we assume here that CO_2 flows are integrated for CCS, although under some market conditions, mills may choose to capture CO_2 from the recovery boiler alone due to higher concentration of biogenic CO_2. We conservatively assume no waste heat availability for CCS, although many mills likely have waste heat, and that additional heat required is supplied from biomass combustion. We assume that the additional electricity requirement of 132 kWh / tCO_2-captured can be met by existing combined heat and power (CHP) output or new CHP associated with increased biomass combustion (15). In the recycling scenario, mills would generate significant excess electricity but we do not model substitution benefits associated with displacing grid electricity, nor do we model displacement of fossil energy in the production process because of uncertainties around heat quality requirements in paper mills. We assume a CO_2 capture rate of 90% (15, 17). To test limits on geological storage of CO_2, we use country-level storage estimates of identified storage potential in sedimentary basins (42).

Recycling scenario

In the recycling scenario, we model global paper production at the theoretical maximum of recycled feedstock inputs. This maximum Recycled Input Rate (RIR) has been estimated to be 70%, which corresponds to a 93% collection rate, similar to the present-day collection rate in countries with intensive recycling programs (22). The max RIR is calculated as an aggregate measure of feedstock input, so it is not specific to product classes. We estimate new product-specific pulping allocations by scaling recycled inputs proportionally to the base cases (RIR = 38%). We assume a saturation point of 95% recycled input, since some products, like newsprint, already have a high proportion of recycled inputs. Subsequently, we model the remainder of pulping allocation for each product class as proportional to the base-case allocations for chemical and mechanical pulping (Table 2). We calculate virgin feedstock subsequently available for alternative uses relative to HWP production and the yield ratio for each pulping process:

$$Available\ feedstock = \frac{Allocation_0 - Allocation_{Rec}}{Yield\ ratio} * Production$$

In the Recycling + CCS scenario, this available feedstock is assumed to be used for BECCS, whereas in the Recycling only scenario, the fate of the available feedstock is assumed to be outside the system boundaries for this analysis (e.g., changes in feedstock demand yield changes in forest cover or management).

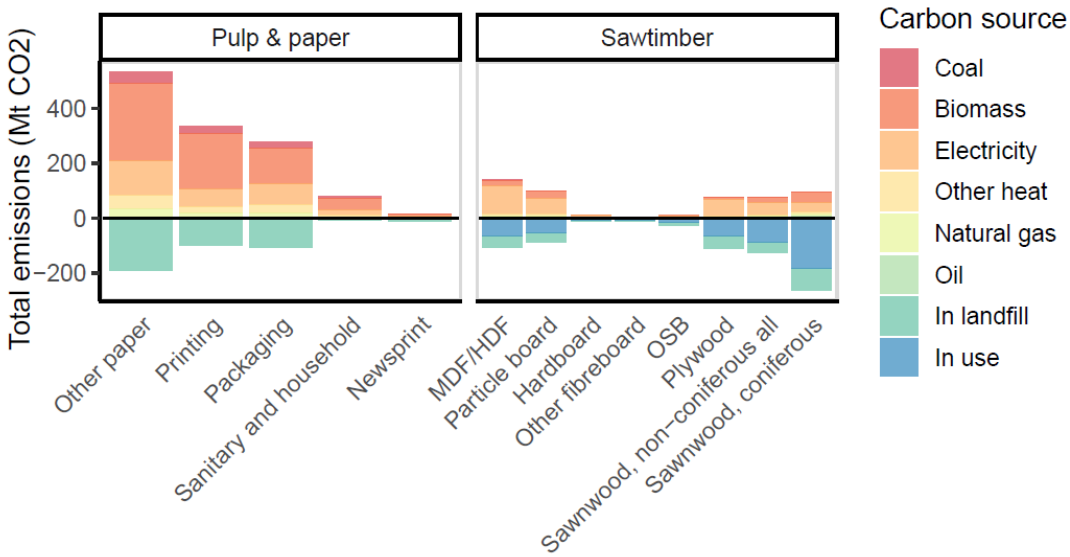

Figure 1. Total baseline global production emissions and carbon storage in landfills and in-use products across several HWP categories in 2020. Carbon emissions are shown as positive and carbon removals are shown as negative. MDF/HDF is medium- / high-density fibreboard. Biomass is the sole source of biogenic emissions.

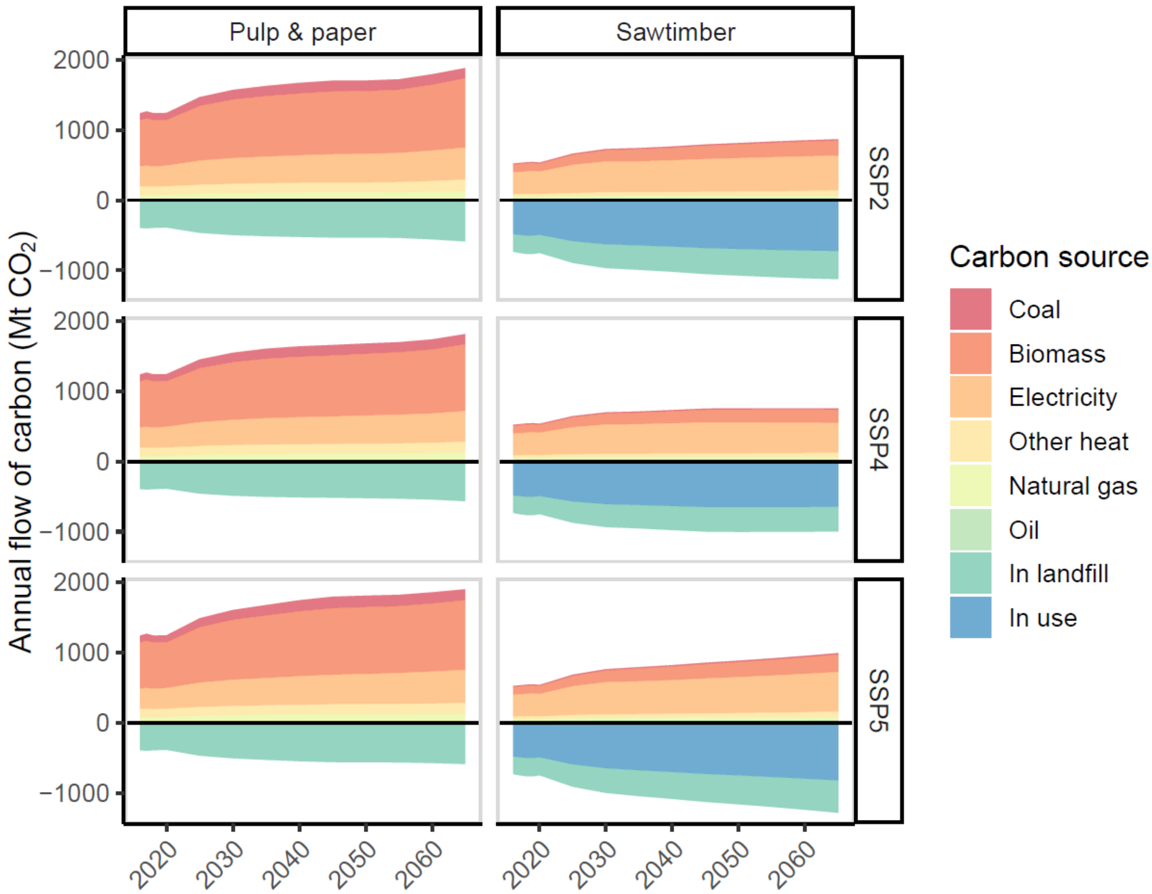

Figure 2. Projected annual baseline flows of carbon associated with HWPs through 2065 under three SSP scenarios. Values after 2020 are modeled. Carbon storage values for "In landfill" and "In use" are taken from a single point in time (year 20) for each projected year, rather than showing cumulative decay and accumulation of HWP storage over time. Carbon emissions are shown as positive and carbon removals are shown as negative. Biomass is the sole source of biogenic emissions.

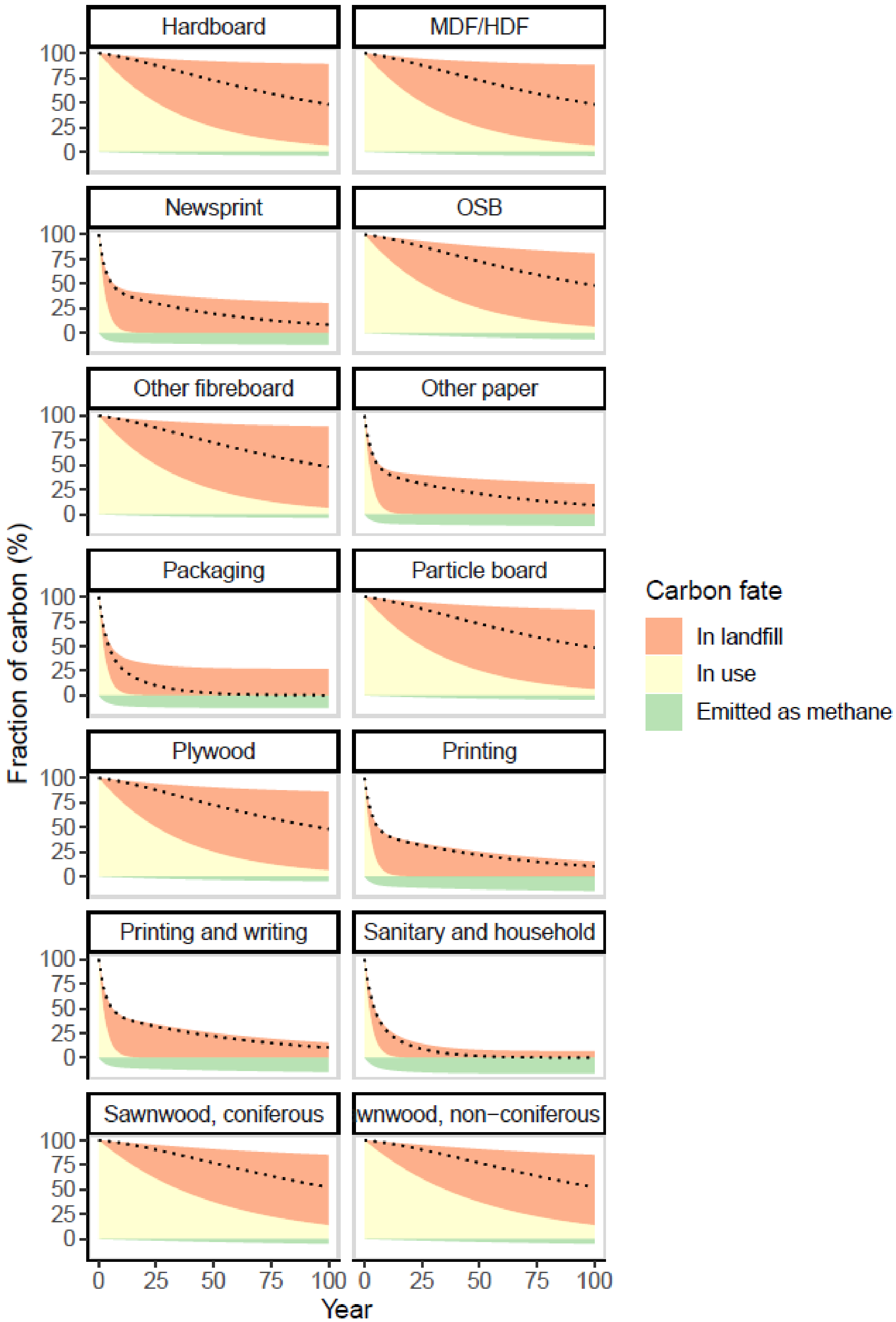

Figure 3. Fate of carbon stored in HWPs over time. Dotted black lines show the total carbon storage (in landfill plus in use) if HWPs are assumed to decay completely in the landfill (i.e., CSF = 0). Carbon emitted as CO_2 is not shown.

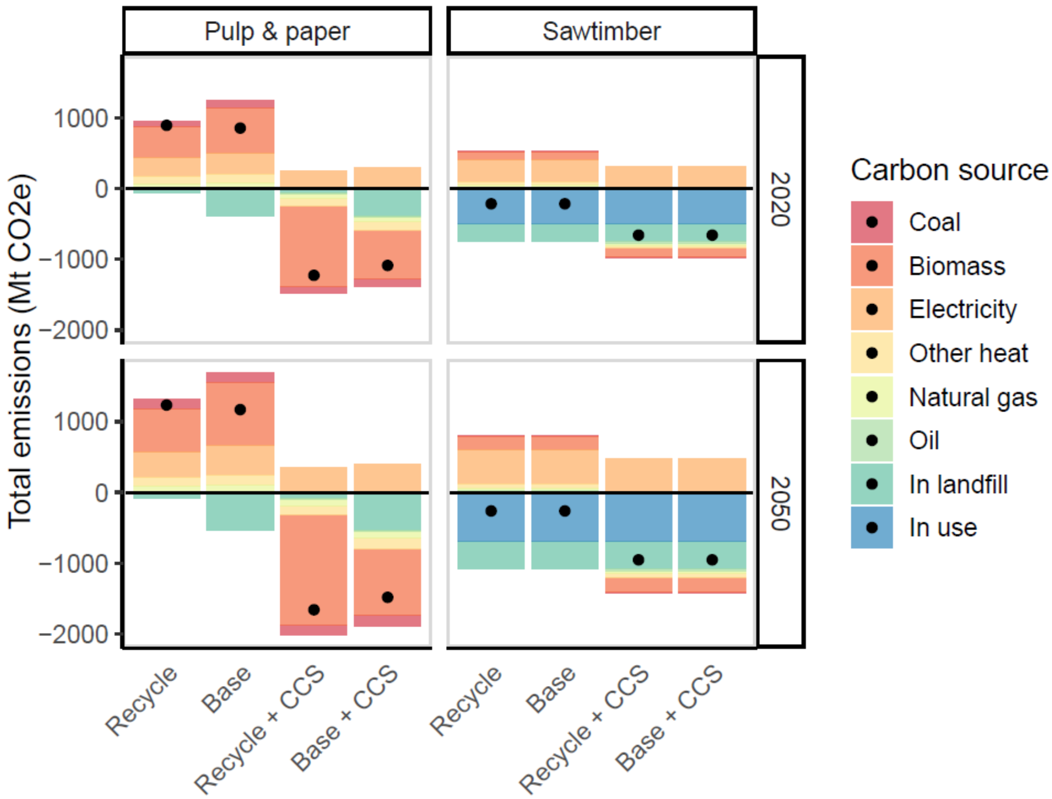

Figure 4. Comparison of four global HWP carbon management scenarios in 2020 and 2050. Black dots indicate the net value for each scenario. The Recycling and Base scenarios are identical for sawtimber products, but are still shown separately for consistency. Carbon emissions are shown as positive and carbon removals are shown as negative. Biomass is the sole source of biogenic emissions.

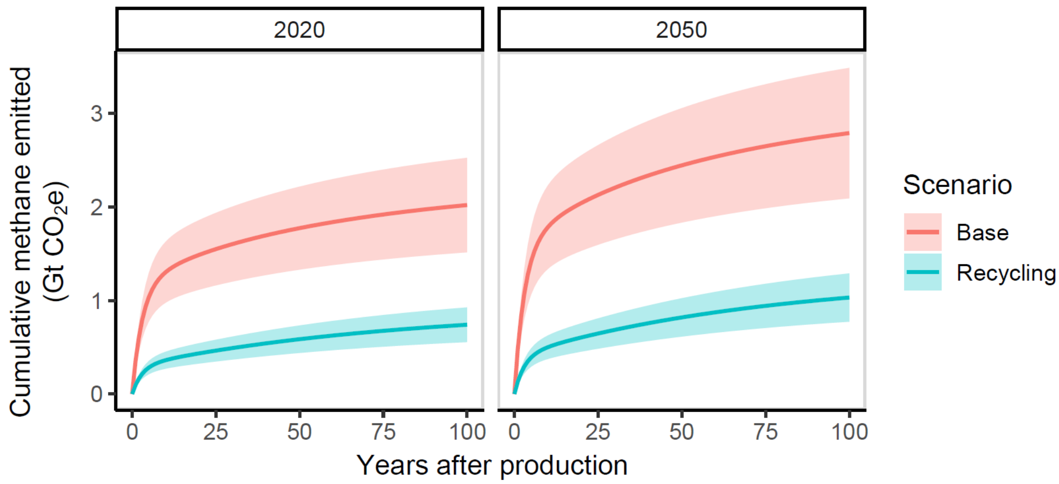

Figure 5. Comparison of methane emissions for all HWPs produced in 2020 and 2050. The x-axis indicates the lifetime of a product produced in either 2020 or 2050. Shaded areas represent expected confidence range on methane emission rates. The Recycling and Base scenarios differ in the recycling rate of pulp and paper products.

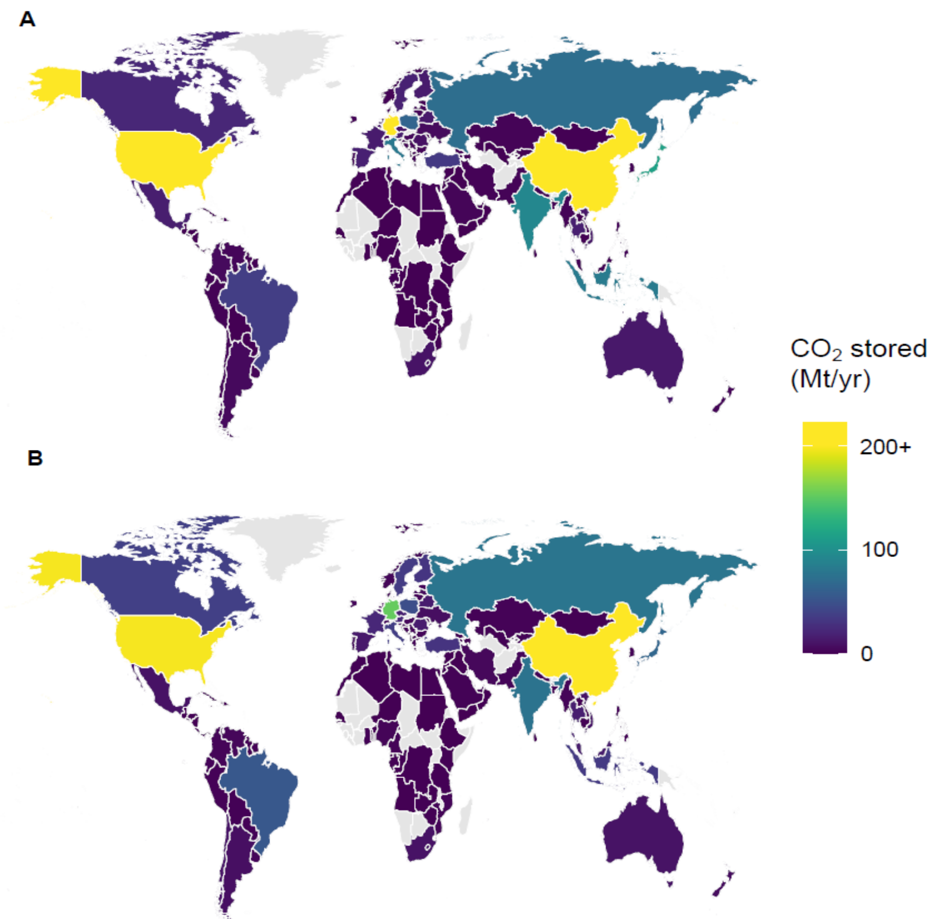

Figure 6. Global distribution of carbon removal potential in 2050 for CCS (A) and storage in HWPs (B) in the Base + CCS scenario. Outliers are grouped to increase legibility of observations closer to the median values. Gray countries are missing observations from one or more of the data sets used here (most often IEA).

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Conclusion: Implications for the durability of stored forest carbon

Recently, carbon dioxide removal (CDR) has become an increasingly important climate change mitigation strategy, but it depends on the ability to store carbon for long time periods. Much of the preceding chapters focus on the durability of carbon stored in forests and in wood products. Carbon stored in forests is inherently dynamic: both social and natural disturbances contribute to the flux of carbon from forests. In many forests, these disturbance events are expected to increase into the future, particularly with climate change. Forest management interventions like thinning, prescribed burning, and species selection can mitigate natural disturbance risks, but these strategies will have to evolve as climate change disrupts the resilience of forest ecosystems. Carbon stored in harvested wood products (HWPs) has the potential to be durable for decades to 100's of years. But the durability of carbon stored in HWPs is highly variable and depends on numerous factors that are difficult to accurately quantify. Forests play a key role across the spectrum of CDR opportunities, from reforestation to bioenergy with carbon capture and storage (BECCS). Throughout this dissertation, I work to understand the full carbon life cycle of some of these strategies and the implications for climate change mitigation efforts.

In Project 1, I considered the potential carbon and wildfire impacts of different management trajectories in California timberland. I formulate three representative scenarios and, in the most aggressive treatment scenario, I find that wildfire hazard reduction treatments in California could produce 7.3 million (M) oven-dry tons of chip biomass per year, an eight-fold increase over current levels. However, the fate of this biomass critically affects the climate impact of widespread forest treatments, with climate benefits of individual wood uses spanning an order of magnitude. With a suite of innovative wood uses, increased management and wood use could yield net climate benefits between 5.4-15.9 million tonnes of carbon dioxide equivalent (Mt CO_2e) per year when considering impacts from management, wildfire, carbon storage in products, and displacement of fossil-intensive alternatives. I find that products with durable carbon storage confer the greatest benefits, including traditional timber products and products with carbon capture and storage. Concurrently, I find that treatment could reduce wildfire hazard on 12.1M acres, 3.1M of which could experience stand-replacing effects without treatment. I conclude that innovative wood use can support widespread fire hazard mitigation and reduce net-CO_2 emissions in California by increasing the stability of carbon stored in forests and in harvested wood.

In Project 2, I investigated the impact of HWP carbon accounting simplifications in California's precedent-setting compliance forest carbon protocol. This protocol has been used to generate 133 M offset credits for forest harvest deferral projects, which are then used by large polluters to forego reducing a portion of their carbon emissions under California's compliance cap-and-trade system. I identify that the way the protocol interprets the production and decay of HWPs produces a temporal misallocation of credits. I find this simplified accounting yields 42 $MtCO_2e$ of credits generated too early, nearly half of the credits in the study sample. Functionally, this error delays the climate benefits of the program by offsetting fossil fuel emissions today with emissions reductions that won't be realized for decades. This finding highlights the importance of careful accounting of HWPs and represents a critical example of how HWP accounting can influence climate policy outcomes.

In Project 3, I expanded my focus from the United States and analyze the global HWP life cycle, including products from all countries reported by the Food and Agriculture Organization of the United Nations. I model the total production emissions for global HWPs as well as the carbon stored in in-use products and in landfills. I find that global emissions from producing HWPs, exclusive of biogenic emissions, is 1429 $MtCO_2e$/yr, with most of those emissions coming from the pulp and paper industry. In contrast, I find that most of the carbon stored in HWPs is stored in sawtimber products. In total, carbon stored in HWPs represents a carbon sink of 1602 $MtCO_2e$/yr. I model multiple scenarios that reflect step changes in the production of HWPs and management of HWP carbon. I find that there are relatively limited benefits associated with increased recycling of paper products, but I identify a large opportunity in retrofitting existing pulp and paper mills with CCS (as well as a smaller opportunity for CCS on sawmills). In 2050, a full rollout of CCS on HWP mills globally could durably store 1145 $MtCO_2e$/yr. In this scenario, the HWP sector would represent a net carbon sink of 3319 $MtCO_2e$/yr. Retrofitting existing mills with CCS is a potentially low-cost application of BECCS technologies that requires minimal new infrastructure and minimal additional biomass feedstock. This study highlights the magnitude of the global HWP carbon sink and its potential to contribute to climate change mitigation goals.

In aggregate, these studies show that carbon accounting of HWPs can be critical to effective climate policy and can intersect with forest management policy. Where forest management policies intersect with climate policy, innovative use of harvested wood can resolve potential conflicts between policy goals, as I show in Chapter 1. Where HWPs are a single component of climate policies, mis-accounting for them can distort climate outcomes significantly and undermine the efficacy of policy, as I show in Chapter 2. And future and existing climate policy can leverage existing flows of carbon in the HWP lifecycle to reduce or remove significant quantities of carbon from the atmosphere, as I show in Chapter 3. In all these cases, a nuanced understanding of the life cycle of HWPs is essential to ensure that actual climate outcomes are aligned with policy goals.

Carbon offset protocols represent one prominent example of HWP carbon accounting that intersect with other climate policies. Offset protocols are illustrative of both the challenges and the potential of mobilizing forest carbon for the purposes of quantified climate action. HWP carbon accounting is critical in numerous existing and emerging protocols for carbon offsets, both in compliance and voluntary carbon markets across the globe. These protocols fall into two broad categories: (a) protocols that predominantly emphasize in-forest carbon storage, such as reforestation or improved forest management (IFM), and (b) protocols that emphasize durable, engineered storage of harvested forest carbon, like biochar. The first has been the focus of most existing protocols to date but interest in engineered carbon storage solutions is growing.

A large portion of carbon offset protocols to date have incentivized increasing or protecting carbon stored in forests. In many cases, HWPs are not considered at all, even where they are a central part of the forest carbon cycle. For example, a recent high-profile IFM protocol was proposed by the forest data company NCX for short-term harvest deferral in working forests under the Verified Carbon Standard. The protocol fully disregarded the existence of HWPs in the counterfactual scenario, essentially assuming immediate oxidation of all harvested carbon. If this protocol is accepted, it could generate millions of tons of carbon offset credits every year, many of which would not represent real emissions reductions. Even if the emphasis of a carbon protocol is on in-forest carbon storage, simplifying or ignoring HWPs can significantly undermine the integrity of protocols and inflate the climate impact of carbon projects. Particularly in the context of carbon offsets, accurate carbon accounting is essential because an offset credit is used to neutralize exactly one ton of CO_2 emitted. Inaccurate accounting can give the appearance of carbon neutrality as CO_2 continues to accumulate in the atmosphere. Further, if carbon stored in forests is eventually released to the atmosphere, an offset credit simply delays emissions but does not neutralize them permanently. Delaying emissions for only decades may pose difficult intergenerational tradeoffs of climate impacts.

At the same time, engineered carbon removal solutions that store carbon for centuries or more are gaining interest in carbon markets. Biomass carbon removal and storage (BiCRS) solutions that aim to slow or eliminate the release of biomass carbon appear to be scaling up quickly. BiCRS includes technologies like biochar, biomass burial, and engineered wood products – often made from forest biomass. Unlike BECCS technologies, BiCRS recognizes that carbon storage may be the highest-value end-use for biomass resources, particularly as demand for carbon removal credits increases. It represents a unique solution set that can sequester large volumes of carbon at relatively low prices, filling a gap between carbon storage in ecosystems and advanced engineering approaches like direct air CO_2 capture and storage (DACCS). Carbon registry bodies like Puro.Earth have developed several protocols for biochar, engineered wood products, and other BiCRS solutions and more protocols and projects are being actively developed. Increased emphasis on BiCRS emphasizes the need to resolve questions around the durability of carbon storage and the emissions associated with production. As with all climate policy that involves HWPs, careful attention to the full life cycle HWPs is necessary to validate the intended climate benefits.

Forests are a focal point for climate change mitigation efforts because of their significant role in the global carbon cycle. Many studies, policies, and companies have emphasized the potential of forests and, implicitly or explicitly, the role HWPs in the climate response. Adequately addressing climate change will require a diverse set of approaches to both mitigation of emissions and removal of atmospheric CO_2. While forests are not a singular solution, it is critical to understand the role they can play in mitigating climate change, both in the forest and in the way we use harvested wood.

Supplementary Information for

Innovative wood use can enable carbon-beneficial forest management in California

Bodie Cabiyo, Jeremy S. Fried, Brandon M. Collins, William Stewart, Jun Wong, Daniel L. Sanchez*

INTRODUCTION

In this study, we employ multiple models to understand the net carbon balance of forest treatment, wildfire, and harvested wood utilization (Table S1). These three components are described in detail in the following methods. In Section 1, we describe our approach to modeling forest management, wildfire hazard, and the carbon accounting associated with both. In Section 2, we describe our approach to lifecycle accounting of harvested wood, including the process emissions, substitution benefits, and end-of-life for several residue-based products (Table S2), as well as for conventional sawtimber products.

To understand the net carbon balance of management, we evaluate four management and wood-use scenarios which are designed to bracket a range of likely futures in California (Table S3). These scenarios represent the aggregation of our in-forest modeling, including management and wildfire, and our harvested wood accounting (i.e. "out-of-forest" modeling).

Table S1. *The three main components of this analysis rely on the combination of multiple models.*

Component	Model used	Overview	Section
Forest growth, management, and wildfire hazard	FVS, BioSum	We model six treatments in FVS over 40 years, model the economics, wildfire hazard, and wood supply for each, and then choose the optimal treatment for each plot.	1.1, 1.2
Wildfire carbon impacts	Stochastic wildfire model	We build a stochastic model to estimate wildfire occurrence based on future fire probabilities and fire weather.	1.2
Substitution benefits, process emissions, and end-of-life carbon accounting of wood products	Cradle-to-grave accounting based on several published LCA papers	We combine published lifecycle values by harmonizing system boundaries and emissions from harvest, transport, and electricity across all pathways.	2

Table S2. *Lifecycle carbon benefits for nine forest residue product pathways, in terms of tC benefit/tC in feedstock. Storage includes landfilled wood and carbon in long-lived products, but does not include storage from CCS, which is included in process emissions. Technologies included in the IWP scenario are indicated with (*).*

Residue pathway	Substitution	Process emissions	Storage	Total	Primary references
Biopower	0.13	-0.02	0.00	0.11	(1)
Decay	0.00	0.00	0.14	0.14	(2)
Biochar	0.12	-0.02	0.26	0.36	(3, 4)
Pyrolysis fuels + char	0.63	-0.34	0.14	0.43	(5)
Pyrolysis fuels	0.63	-0.20	0.00	0.44	(4–6)
Ethanol + CCS*	0.11	0.53	0.00	0.64	(7, 8)
FT fuels + CCS*	0.35	0.46	0.00	0.81	(9)
Biopower + CCS*	0.10	0.72	0.00	0.82	(1)
OSB*	0.94	-0.30	0.54	1.18	(10, 11)
GluLam*	0.94	-0.16	0.48	1.26	(11, 12)
Hydrogen + CCS*	0.80	0.85	0.00	1.65	(13, 14)

Table S3. *We combine the in-forest modeling and lifecycle accounting of wood products across four scenarios.*

Scenario	Residue utilization	Sawtimber utilization	Economic criteria
Low BAU	Biopower / Decay	Current product mix	Manage only corporate-owned land where net revenue >$2500/ha (>$1000/acre)over the modeling period
High BAU	Biopower	Current product mix	All possible management with a residue price of $0
Innovative Wood Products (IWP)	Even mix of IWP product basket	Current product mix	All possible management with a maximum residue price of $100/ODT
IWP + Housing	Same as IWP	Additional sawtimber over Low BAU is used for multi-use and multi-family buildings	Same as IWP

IN-FOREST CARBON

BIOSUM AND FVS MODELING APPROACH

This analysis applies the FIA BioSum modeling framework (1, 2; http://biosum.info) to understand management outcomes on California timberland. We start with data collected from 5,404 field-sampled Forest Inventory and Analysis (FIA) plots between 2005 and 2016 that represent approximately 33 million acres (13.4M ha) of California forest land. We refined this comprehensive, representative forest land sample to limit our analysis to forests offering the most promising potential for management. We retained in the dataset forested "conditions" (full or partial plots) that are classified as timberland[1] and as one of four common California forest types: mixed conifer, Douglas-fir, True fir, and Ponderosa pine. We exclude coast redwood forests, which present little fire hazard, and hardwood forests, which are rarely managed for timber. We consider only the three owner classes that account for nearly all of California's timberland: private (corporate and non-corporate) and National Forest System (NFS).

Forest growth, management, and potential fire outcomes are simulated over 40 years with the Forest Vegetation Simulator (FVS) and the associated Fire and Fuels Extension (FFE), after converting those forested conditions from the FIA database into FVS stand data. Because each "stand" comes from an FIA plot, it represents a known area of California's forest.

Subsequently, we evaluate effectiveness, and estimate costs incurred by and revenues generated from five management sequences. We use multi-criteria optimization to choose the best management sequences for each stand. Based on this optimization, BioSum calculates quantities of merchantable wood and residues[2] that could be delivered from these forests to an existing network of processing facilities. Fried, et al. 2016, upon which this work is based, explains the modeling approach for both FVS and BioSum in detail.

1.1.1 Management Sequences and FVS

We simulate five management sequences (Table S4) designed to represent forest restoration-motivated thinning regimes compatible with provisions of the Sierra-Nevada Forest Plan. Sequences are defined as a repeated treatment over the modeling period. Treatments can occur only once or twice over the 40-year modeling period under rules that set the minimum re-treatment interval at 20 years. The treatments differ with respect to thinning style, maximum size of trees allowed to be harvested, and approach to addressing surface fuels. Each treatment considers a basal area reduction of up to 33% and implements thinning with a whole-tree harvest system, using either a mechanical harvester (on gentle slopes) or manual (chainsaw-based) felling (on steep slopes), so harvested trees of merchantable size generate

[1] Land capable of producing an average of at least 20 ft^3/acre/year (0.56 m^3) of wood and not legally reserved from timber management.
[2] Inclusive of branches, bark and foliage.

very little surface fuel[3]. After thinning, surface fuels are either reduced by prescribed fire or rearranged via lop and scatter (severing and scattering stems near where they're cut). We also modeled a "Grow Only" sequence to represent the hands-off approach currently typical on most publicly owned forestland.

Table S4. *Management sequences modeled. Thinning styles are thinning proportionally across diameter classes ("ThinDBH") and thinning from below ("ThinBBA"). Entry threshold is the basal area that triggers thinning (provided there is a 20-yr hiatus between thinning entries). Max DBH is the breast height diameter of the largest tree allowed to be harvested as part of the thinning, on public (pub) and private (pvt) land. Surface fuel method is either rearrangement (lop and scatter) or reduction (via prescribed fire).*

Thinning Style	Entry Threshold (m^2)	Max DBH (cm) cut, by ownership	Surface fuels treatment
ThinDBH	BA≥11	91 pvt/76 pub	Rx Fire
ThinDBH	BA≥11	91 pvt/76 pub	Lop/scatter
ThinDBH	BA≥11	91 pvt/53 pub	Lop/scatter
ThinBBA	BA≥11	91 pvt/76 pub	Lop/scatter
ThinBBA	BA≥11	91 pvt/76 pub	Rx Fire
GrowOnly	NA	NA	NA

The modeling approach used in FVS is described in detail in (17).

1.1.2 Management Optimization with BioSum

FVS simulation of forest management sequences, described above, is one part of the analysis workflow supported by the BioSum modeling framework. BioSum's four major modules include: (a) **Database**, for loading and managing forest inventory data from the FIADB, the national FIA database; (b) **FVS**, for creating FVS input files from the FIA database, defining prescriptions and management sequences, and importing FVS outputs to drive later stages of the analysis; (c) **Processor**, which calculates harvest costs and revenue from sales of harvested wood for each combination of stand and sequence, from "cut list" data output from FVS, and (d) **Optimizer**, where the analyst sets criteria for what constitutes successful management outcomes, chooses one or more attributes to optimize, subject to constraints involving other objectives and/or economic feasibility, and establishes rules to break ties among alternative sequences that achieve the same optimum to arrive at a single, optimal management sequence for each stand. Extensive documentation on BioSum and examples of its use are available at http://biosum.info.

[3] However, on both gentle and steep slopes, trees 51 cm DBH and larger are assumed to require manual felling and bucking with chainsaws, so where such trees are cut, they contribute "activity fuels".

1.1.2.1 Modeling management costs and revenues

We define management sequences to allow thinning entries in each stand to occur in up to two of four years over a 40-yr time horizon: years 1, 11, 21, and 31. When basal area exceeds a prescription's threshold and the rule that separates thinning entries by 20 years is satisfied, FVS simulates the harvest activity and resulting changes to stand characteristics and BioSum calculates associated management costs and revenues. Fixed costs of moving harvesting equipment to a treatment site are distributed over an assumed 40-acre (16-ha) harvest area. The 16% of stands derived from FIA plots that are more than 2500 ft. (760 m) from an existing forest road (surfaced or unsurfaced, excluding skid trails) are assumed to be inaccessible and ineligible for management, as modeling construction of new access roads is beyond the scope of a BioSum analysis.

Delivered merchantable (sawtimber) wood values are set according to the California Board of Equalization timber tax reports (18). Residue prices are set at maximum price of $100 per ODT delivered ($50 per green ton). Trees 20 cm DBH and larger are processed and valued as merchantable saw logs. Small (10 – 20 cm DBH) harvested trees are assumed to be chipped (boles and branches), along with trees with at least 50% of bole volume classified as cull and non-commercial species (mainly hardwoods) of all sizes. Trees less than 10 cm DBH are cut, lopped (cut in half), and scattered near where they are felled, except on steep slopes where this threshold is 13 cm. Tree harvest systems and associated costs are dependent on plot slope. We classify "steep" slopes as 40% grade (18 degrees) or more. On steep slopes, we model treatment cost using more expensive harvest systems (cable manual whole tree).

To account for the costs of moving both saw logs and wood residues from the forest to processing facilities, BioSum simulates road travel time from the point on the road network nearest to each FIA plot to each potential processing site. Travel time is determined by rated road speed and the locations of existing processing facilities. BioSum identifies the closest (in time) merchantable facility and bioenergy facility for each plot and assumes that harvested wood would be transported only to those facilities. We use a list of 36 active and idle wood-processing facilities, and assume these will be scaled-up, restored, and/or supplemented with additional, newly constructed, co-located facilities in response to increased wood supply. We assume a round-trip haul cost of $7 per green ton-hour for both types of wood. For each management sequence and stand, these transportation costs are combined with harvest costs and wood revenues to calculate the net revenue expected to result from management in a given decade. These can be summed over all decades in which treatment occurs to obtain a 40-year net revenue estimate.

1.1.2.2 Low-value feedstock suitability

About one fifth of the forest residue wood basket in IWP is composed of small trees (10-20 cm), and the rest is composed of branches and tops of larger trees, as well as the entirety of non-merchantable trees (mostly hardwoods). While we model these residues as an aggregate supply of chipped material (i.e. inclusive of boles, branches, bark, and foliage), we assume that this material could be sorted at the landing site or a receiving facility. For example, sorting at the landing site could involve stripping branches off treetops and small trees to produce boles suitable for OSB stranding (19) and dirty biomass chips suitable for biofuels or biopower. Sorting at the receiving facility could involve filtering fines and bark for use as process heat and chips as core material in three-layer OSB (20). While these sorting processes are not directly modeled here, we expect that they would not substantially change the cost of delivered feedstock because the haul cost, which makes up the largest portion of the total delivered cost for these materials, remain unchanged given that the costs of their felling and yarding are already accounted in the cost of merchantable wood. In some cases, it may not be possible to sort low-value wood because of economic constraints or insufficient quantity. The ability to tolerate bark and needles for innovative wood products is process specific. However, we are confident that numerous technologies can accommodate dirty chips, including biofuel production technologies. Modern gasification technologies, which underlie many of the biofuel technologies we consider here (including hydrogen), are tolerant of varying wood quality. For instance, Red Rock Biofuels, a Fischer-Tropsch diesel biofuel facility under construction in Lakeview, OR, will accept “dirty” chips including bark and needles. In California, a recent budget proposal by the Department of Conservation included $50M for a 30,000-ODT/year forest residue-to-fuels gasification facility. Further, there are several examples of commercial gasification technologies that process municipal solid waste, a feedstock with similar heterogeneity to biomass chips. Examples include gasifiers developed by Omni Conversion Technologies and multiple gasification technologies currently operational in Canada (21). We model a suite of technologies to account for the fact that no single technology can be expected to utilize all the low-value wood produced on California timberland.

1.1.2.3 Management decision criteria

We set three sequential optimization criteria to select the optimal management sequence for each stand in BioSum. First, all combinations of stand and management sequence that were incapable of generating positive net revenues were dropped from further consideration. The second criteria, which defines treatment effectiveness, is defined as a reduction in the 40-yr mean fire-induced mortality, as a fraction of total live basal area, predicted by FFE-FVS under severe fire weather conditions relative to the same hazard metric calculated for the Grow Only sequence (see Section 1.2, below). Finally, the optimal sequence was defined as the effective sequence that maximized live tree carbon at the end of the 40-year analysis period. In cases where two or more effective management sequences had the highest 40-year live tree carbon, the sequence with the greatest reduction in fire mortality (as defined above) was chosen.

WILDFIRE MODELING

We model potential fire outcomes for each stand, year, and treatment sequence with the FVS Fire and Fuels Extension (FFE). These potential fire outcomes are modeled independently for each year and represent “what-if” fire hazard metrics. We develop a stochastic model to understand how these potential outcomes would manifest under a realistic fire regime, since only a small portion of the forest will burn in any given year. We run 5000 Monte Carlo simulations to reflect the inherent spatial and temporal variability in wildfire. In each simulation, we randomize (a) how many plots burn, (b) which plots burn, and (c) when they burn. In the first step, we randomly select a predicted fire frequency from a log-normal distribution. We assume a 3.8% increase (22) over an observed annual fire probability of 0.089% (23), so our projected annual fire probability is 0.092%. We assume this new frequency varies by the same amount (i.e. the standard deviation is 3.8%). These assumptions are within the bounds of the modeling conducted by Mann et al. (2016) for the 2030-2050 time period. Subsequently, we randomly select a cohort of plots that will burn over a forty-year period and then randomly select the year in which they burn. Predicting future wildfire occurrence and extent is inherently problematic, but a multi-step approach to modeling spatial and temporal stochasticity can improve accuracy of models (24).

Combustion, Post-Fire Decay, and Reduced Growth of Fire-Affected Stands

We consider three primary fire effects: combustion emissions, post-fire decay of fire-killed trees and reductions in stand growth.

Combustion

We model combusted carbon as:

$$C_{combusted} = C_{FWD} + C_{CWD} * F_{CWD} + C_{LT} * F_{LT}$$

Where the total carbon emitted in a wildfire event is the sum of fine woody debris carbon (FWD), coarse woody debris carbon (CWD) and live tree carbon (LT) times the fraction (F_{CWD} and F_{LT}) of aboveground carbon expected to combust. We assume fine woody debris combusts completely. We parameterize $F_{combustion}$ with values observed in comparable dry conifer forests in southern Oregon (25). We apply $F_{combustion}$ values that are specific to both wood class (CWD and LT) and wildfire basal area mortality class (low: <20%, moderate: 20-95%, severe: >95%). For most fires (i.e. “moderate” mortality), $F_{combustion}$ for live trees is 0.07. The values for $F_{combustion}$ used here are very low compared to values commonly used in combustion models but are likely a better representation of observed combustion rates (26). For comparison, we estimate $F_{combustion}$ for CWD and LT combined at 0.15 based on a study of pre- and post-fire observations on FIA plots across California, although the exact value is not specified in the study (27). While this combustion rate increases the magnitude of combustion emissions from wildfire, thus improving the carbon benefits of management, it does not change the core findings of our scenario analysis.

Post-Fire Decay

We assume a post-fire decay rate constant of 0.016 yr^{-1}, which is based on observed decomposition rates in similar dry, coniferous forest after a variable-severity fire (28). We take this decay rate to be representative for the species and environmental conditions in the present study. Decomposition rates vary depending on climate, species, char-content, and whether dead wood remains standing or falls to the forest floor. We do not explicitly model snags (standing dead wood), although the decay rate of snags is similar to our generalized rate (28). The decay rate constant we use here is specific to post-fire decay, which few studies have quantified. This rate is lower than reported rates for non-post-fire decay. For example, Douglas fir, the most common species in our study, has a CWD decay rate of 0.021 in California (2). Using a low rate constant limits the magnitude of wildfire-induced emissions in our simulations, largely because we consider a 40-year modeling period. When we test a decay rate of 0.033 yr^{-1} (i.e. a half-life of 20 years), we find that decay emissions roughly double, increasing the magnitude of carbon benefit associated with management. This does not, however, change the core conclusions of our scenario analysis. We assume simple exponential decay across all wood classes. Although some models suggest a two-stage decay equation to capture variable rates across the decay process, this effect is poorly defined in the literature and likely to be small (29).

Post-Fire Growth Adjustment

After a fire occurs in a stand, we reduce the total live tree carbon and future growth of the stand proportionally to the basal area mortality. We model post-fire live tree carbon during any given year *i* after a fire (LTC_i) with the following equation:

$$LTC_i = LTC_{0i} * (1 - BA_{mortality})$$

Where LTC_0 is the FVS-modeled live tree carbon during year *i*, absent fire, and $BA_{mortality}$ is the modeled basal area mortality fraction during the year of the fire. While basal area mortality is not always a perfect proxy for mortality volume (and thereby carbon), in our dataset of over 300,000 FVS observations of predicted annual stand-level wildfire effects, we find a Pearson correlation coefficient of 0.96 between ($BA_{mortality}$ * live tree volume) and FVS-reported mortality by volume. In the case of fires with very high $BA_{mortality}$ (>95%) on corporately managed land, we assume that forests grow 50% faster than the base case to account for actively managed regeneration (30), although this approach likely underestimates regrowth on these stands. At the high end of this effect, stands with 95% mortality will regrow at a rate of 7.5% of their pre-fire growth rate, including the 5% of BA (often large trees) that was not killed. The effect of this assumption is very small during our 40-year modeling period. We also assume zero decay of merchantable-sized trees after fire in these forests, because rapid salvage logging is common in corporately owned forests.

Fire Weather Definitions

Fire weather parameters are required for FVS-FFE to provide estimates of potential fire behaviour under different fire weather conditions. All of the forests we modeled in this study using the NC, WS, CA and SO FVS variants are considered to be in arid or semi-arid climate types (31). Fire weather for 90^{th} and 97.5^{th} percentile conditions, derived from 30 years of Remote Access Weather Station (RAWS) data from multiple locations, filtered for the fire season, was analysed with Fire Family Plus to generate parameters supplied to FVS-FFE. We relied on the 97.5^{th} percentile and 90^{th} percentile weather parameters to represent a range of likely future fire weather conditions. Temperature, wind speed, and relative humidity assumptions are given in Table S5.

Table S5. *Weather Parameters Used to Model Fire in Conifer Forests of the Sierra and Interior Coast Ranges*

Parameter	97.5th% Weather	90th% Weather
Wind speed, km/h (mph)	32 (20)	26 (16)
Temperature, °C (°F)	33 (91)	32 (90)
Relative humidity (%)	15	17

Assumed fuel moisture parameters (Table S6) are similar to those used by others who have modeled fire potential in California forests, and are assumed constant over the 4 decades of FVS projection. For example, 1-, 10- and 100-hour, live herb and live woody fuel moistures and wind speed data are similar to those reported by (32–36). 1000-hr and duff moisture parameters are similar to the observations reported in (37, 38) and are appropriate for Sierra, Cascade, and interior coast range coniferous forests. It is possible that the effect of this assumption will tend to understate severity of fire and mortality rate if climate change increases temperature and wind speed and/or reduces humidity.

In the Results, we present a mean of wildfire simulation results conducted with 97.5 and 90th percentile fire weather. While these weather scenarios represent more extreme conditions, they reflect both the observed tendency that a majority of fire area burns in a relatively small number of very large fires, which occur under more extreme fire weather (24, 39), and the likelihood of increasing incidence of severe fire weather within our modeling period (40). Several recent studies have used similar increased incidence of severe fire weather in their future projections (41–44).

__Table S6.__ Fuel Moisture Parameters Used to Model Fire in Conifer Forests of the Sierra and Interior Coast Ranges

Fuel Type	Description	97.5th% Weather Conditions	90th% Weather Conditions
1-hour fuel	The 1-h time lag fuel pool consists of dead and down fuel particles less than ¼-inch (6 mm) in diameter (i.e. litter).	1.8	3
10-hour fuel	The 10-h time lag fuel pool consists of dead and down fuel particles between ¼-inch (6 mm) and 1-inch (25 mm) in diameter.	2.3	3.7
100-hour fuel	The 100-h time lag fuel pool consists of dead and down fuel particles between 1-inch (25 mm) and 3 inches (75 mm) in diameter.	4.2	6.6
3" fuel	The 1000-h time lag fuel pool consisting of down fuel particles larger than 3 inch (75 mm) diameter.	8	12
Duff	Duff	20	40
Live woody	The live woody fuel pool is the foliage of shrubs and small trees plus the fine live branch wood of shrubs and small trees. Fine live branch wood is generally considered branches less than ¼-inch (6 mm) in diameter.	70	80
Live herb	The herbaceous fuel pool is the load of standing live and dead grass stems and other herbaceous fuel. Both the live and dead standing components are included in this fuel pool; the live and dead components are separated at the time of fire behaviour simulation	30	30

Wood Products Lifecycle Accounting

Structural Wood Products Modeling

We use a harvest-to-grave system boundary for the lifecycle accounting of merchantable wood products over 40 years. We consider one ton of harvested carbon as the primary unit of analysis. We model the in-forest carbon outcomes from increased management as previously described. The methods used to calculate product substitution and end-of-life are described below.

Substitution and Production Emissions

To model the substitution benefits attributable to the use of merchantable wood, we adapt the methodology of (11) to the California market context. Smyth et al. (2017) calculate an economy-wide displacement factor for wood products in construction using published values for emissions from extraction, transportation, and production for common building materials. Here, we retain all values used by Smyth except for end uses for wood products, which are economy-specific. In place of the Canada-specific values used in Smyth et al. (2017), we use historical California HWP end use data (45). Because the end use categories reported by Smyth et al. (2017) and by Christensen et al. (2017) are different, we aggregate the categories used by Christensen et al. into the less granular categories used by Smyth et al. (Table S7). As in Smyth et al. (2017), here we disregard two end-use categories: packaging/shipping materials and non-disclosed end uses (e.g. home-made furniture). Christensen et al. aggregate saw timber biomass (e.g. sawdust) and biomass chips, so here we assume that 5.2% of the merchantable yield goes towards biopower and 94.8% goes towards wood products (46). Using this approach, we calculate the substitution benefit of harvested merchantable wood to be 0.75 tC/tC, which is within the range of estimates for other regions (11, 47).

We also consider an alternative wood end use scenario in which 100% of increased wood supply is used to displace steel and concrete buildings (Table S7). As a result, a larger fraction of wood is directed towards the new, multi-family and multi-use building categories. In this scenario, the substitution benefit is 1.75 tC/tC. While this scenario may be in part achieved through increased production of mass timber products (e.g. Cross-Laminated Timber), we do not explicitly model those products here.

***Table S7**. Saw timber end use ratios and their associated half-lives.*

Christensen et al. (2019) category	Smyth et al. (2017) category	Half-life from Skog (2008)	End-use fractions of wood used	Housing Scenario end-use fractions
New housing, single family	Single family	80	0.18	0.06
New housing, multi-family	Multi-family	50	0.2	0.34
New non-residential	Multi-use	30	0.13	0.38
Residential remodel	Flooring	26	0.34	0.12
Manufacturing	Furniture	30	0.15	0.05
Other industrial products	Decking	30	0.17	0.06
Total			**1.00**	**1.00**

End of Life and Embodied Carbon

To model the lifecycle of embodied carbon in HWP, we calculate a category-weighted mean half-life from primary wood product half-lives defined in (48) for the United States of 38 years (Table 4). After a wood product's usage, we assume that 65% of all post-consumer residues (retired wood products) are sent to landfills, 25% to bioenergy facilities, and 10% are not collected (49). Of all carbon in post-consumer residues sent to landfills, we assume that 90% is permanently inert, and we conservatively assume that decay of the remaining 10% happens instantaneously (50). We assume that post-consumer residues generate electricity with a lower heating value of 13.9 GJ/ODT and a heat rate of 80 kWh/mmBtu (76 kWh/GJ) (51), and that the bioenergy produced displaces grid electricity with a carbon intensity of 225 gCO2e/kWh (52). In sum, we consider the downstream storage in use, product substitution benefits, and end-of-life, including post-consumer residue bioenergy generation, fossil electricity substitution and carbon permanently sequestered in landfills.

FOREST RESIDUES LIFECYCLE ACCOUNTING AND ECONOMICS

As with saw timber, we use a harvest-to-grave system boundary for the lifecycle accounting of forest residue products over 40 years. We include harvest and transport, production emissions, product substitution, and end-of-life. Biogenic (in-forest) carbon accounting is described in Section 1. We consider one ton of harvested carbon as the primary unit of analysis. The assumptions and methods for each product are described below. We aggregate values from several published Lifecycle Assessments (LCA's) and adjust those values where necessary to achieve consistency. For every product, we rely on LCA's that have either a wells-to-wheels or cradle-to-grave system boundary. We normalize harvest and transport emissions for all products to be consistent with values used in The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model (9). Where necessary, we adjust values so that forest residues are carbon neutral, because we account for in-forest carbon changes in Section 1. We assume a travel distance of 145 km (90 mi) with backhaul. We assume all electricity to have a California-average carbon intensity of 225 gCO_2e/kWh (52). Cumulative carbon benefits for each product pathway are given in Table S3.

To roughly estimate potential delivered forest residue prices, we assess the internal rate of return for several innovative wood products under a range of delivered residue prices in California over a 20-year financial period. We assume a fuel price of $2/GGE (gallon of gasoline equivalent) and an LCFS credit price of $100/tCO2 abated. We assume $224 / MSF (3/8" basis) for oriented strand board, and no additional policy support. We derive our cost and performance assumptions from existing techno-economic analyses of large-scale production (5, 19, 53, 54). Figure S6 shows the internal rate of return for each of these products for varying delivered residue prices. Based on this analysis, we expect innovative wood products to return positive returns for delivered wood prices as high as $100/ODT. This value is higher than current market rates, but similar to what has been modeled in previous work (14, 55, 56).

1.1.3 Lignocellulosic ethanol with CCS

For the LCA of lignocellulosic ethanol production with carbon capture and storage (CCS) from forest residue we rely on modeling done by McKechnie et al. (2011) and Liu et al. (2011) (7, 8). We obtain relevant process information about forest biomass harvesting and operations from McKechnie et al. (2011), and fuels production with CCS from Liu et al. (2011). We analyze an E85 (85% ethanol, 15% gasoline) pathway from forest biomass, which allows for more direct substitution of gasoline relative to E100. To account for the efficiency loss when switching from conventional gasoline to E85, we assume an efficiency of 5 km/L for E85 and 7.69 km/L for gasoline (7). McKechnie et al. (2011) also include a coproduct credit from natural gas-fired sources for electricity, which we modify to assume displacement of average California grid electricity in 2016 (225 gCO_2e/kWh).

1.1.4 Fischer-Tropsch diesel with CCS

For the LCA of Fischer-Tropsch (FT) liquids production with CCS from forest residue, we rely on Xie et al. (2011) (9), who document various combinations of feedstocks for FT liquids generation, including 100% forest biomass. Xie et al. (2011) use GREET for their LCA using a well-to-wheels boundary: from biomass collection to tailpipe emissions. Forest residues are assumed to have an LHV of 13.243 mmBtu/ODT (13.9 GJ/ODT) and the FT process has a 0.5 LHV efficiency, with 92% are FT liquids and 8% is electricity by LHV. In their BTL-CCS case, they assume 89.9% CO_2 capture ratio with a recycling design, the well-to-wheels emission factor is -150 kgCO_2e/mmBtu (-142 gCO_2/MJ). Xie et al. (2011) assume that the coproduced electricity displaces the average US grid electricity in 2009 (554 gCO_2e/kWh). We update this assumption so that the co-produced electricity displaces the average California grid electricity in 2016. We also update the forest operation and transportation emissions, using data from (7) as discussed above.

While the FT process produces a mixture of diesel and gasoline, we assume a constant baseline carbon intensity (CI) of 100.45 gCO_2e/MJ (diesel's CI, as compared to 100.82 gCO_2e/MJ for gasoline) for all FT liquids produced since Xie et al. (2011) do not report a breakdown by fuel type.

1.1.5 Biochar production

The LCA of biochar production and use from forest residues relies on data from Roberts et al. (2010) and Woolf et al. (2010) (3, 4). Roberts et al. (2010) analyze feedstocks most similar to the forest residues modeled here, and Woolf et al. (2010) provide general characteristics of biochar. We assume that biochar is produced from the slow pyrolysis of forest residues.

Roberts et al. (2010) use a cradle-to-grave system boundary, which begins from feedstock handling to carbon sequestered by biochar. We assume that the char yield is 29.6% (by weight) of the feedstock input in a slow pyrolysis process. The net stable C in char is 574 kgCO_2e/ton feedstock, which includes emissions from pyrolysis. Roberts et al. (2010) assumes that transport and residue collection emits 19 kgCO_2e/ ton feedstock, but we modify this assumption to be consistent with other pathways. Lastly, Roberts et al. (2010) estimate a natural gas substitution benefit of 229 kgCO_2e/ton feedstock. For end-of-life, we assume that the biochar has a labile fraction of 15% with a half-life of 20 years and a recalcitrant fraction of 85% with a half-life of 300 years (4).

1.1.6 Biopower with and without CCS

The LCA of the electricity with CCS pathway using forest residue feedstock relies on data from Sanchez et al. (2015) and Xie et al. (2011) (1, 9). While Sanchez et al. (2015) consider electricity generation from a blend of lignocellulosic biomass, we supplement this with the LHV of forest residues from Xie et al. (2011) to be consistent across scenarios. We use the biomass integrated gasification combined-cycle (IGCC) with CCS scenario in Sanchez et al. (2015).

Sanchez et al. (2015) consider the growth of the biomass until the generation of the electricity and the subsequent CO_2 storage to be its system boundary. Since forest carbon is accounted previously in our model, we remove the agricultural phase (0.004 tCO_2/mmBtu (3.8 gCO_2/MJ)) from the total carbon intensity (-0.0802 tCO_2/mmBtu (-76 gCO_2/MJ)). Instead of an LHV of 17 mmBtu/ODT (18 GJ/ODT), we assume 13.2 mmBtu/ODT (13.8 GJ/ODT) from Xie et al. (2011) to be consistent with the Fischer-Tropsch diesel pathway. We use a heat rate of 16.32 mmBtu/MWh, implying a facility with a 21% efficiency. For the biopower without CCS pathway, we use a heat rate of 12.5 mmBtu/MWh (13.1 MJ/kWh) (1). We also assume that the generated electricity displaces the average California grid electricity in 2016. We update the transportation assumptions in Sanchez et al. (2015) to our common assumptions, given above.

1.1.7 Pyrolysis fuels, with and without biochar co-production

The LCA of biofuels production and use from forest residues relies on Li et al. (2017), who present a techno-economic assessment of a 2000 t/day facility with red-oak (*Q. rubra)* feedstock (5). This facility produces 50.73 gallons (192L) of gasoline/ODT feedstock and 37.01 gallons (140L) of diesel/ODT, which we assume replace conventional gasoline and diesel. The facility burns the non-condensable gas and biochar for process heat, but natural gas and electricity are also used for the bio-oil stabilization process. This process has a reported carbon intensity of 31.8 gCO_2e/MJ (5). The non-condensable gas and biochar burnt for process heat is assumed to displace natural gas.

We model an alternative process in which biochar is reserved. Pyrolysis of loblolly pine residue yields 50.7 (wt%) bio-oil, 10% char, and 25.3% non-condensable gas (6). Meanwhile, the facility still produces the same amount of gasoline and diesel as above. Li et al. (2017) use an HHV of biochar of 23.05 MJ/kg, and we assume for this to be the same for the biochar that is produced from loblolly pine instead of red oak. We calculate the energy produced from combustion of biochar and replace it with the equivalent amount of energy from natural gas combustion. We use a natural gas carbon intensity of 50 gCO_2/MJ (1). The reserved biochar is assumed to be the same as described by Woolf et al. (2010), although it may have different properties given that it is produced via fast pyrolysis instead of slow pyrolysis.

1.1.8 Hydrogen production

For hydrogen production, we rely on the LCA conducted by Antonini et al. (2021) of hydrogen gas produced from wood waste (13). We model their entrained flow gasifier with pre-combustion CO_2 capture and storage, which has a CI of -130 gCO_2/MJ. This process was chosen because it has the highest rate of carbon capture amongst all modeled hydrogen production processes. We adjust this CI to account for using California grid electricity, which has a lower CI than the EU grid used in their analysis (400 gCO_2/kWh). We further adjust this CI to include harvest operations and transportation to a processing facility, consistent with our other pathways. To convert the functional unit from MJ to ODT-feedstock, we use an energy conversion efficiency of 70%. To model substitution benefits, we assume this hydrogen displaces conventional hydrogen produced from natural gas via steam reforming in California, which has a carbon intensity of 120 g/MJ (14).

1.1.9 Oriented Strand Board (OSB) and GluLam production

We rely on Puettmann et al. (2013) to represent the lifecycle greenhouse gas emissions from oriented strand board (OSB) production (10). Puettmann et al. (2013) use a well-to-wheel approach to assess the lifecycle impacts of producing OSB in the Southeastern United States with loblolly pine (*P. taeda L.)* and slash pine *(P. elliottii E.)* feedstock. Puettmann et al. (2013) assume a transportation distance for feedstock of 89 miles, consistent with our other pathways. We modify their calculated production emissions (275 kg CO_2e/m^3) to account for a lower carbon intensity for California grid electricity. CO2e/MWh, and an averaged Southeast United States grid carbon intensity of 454 gCO2e/kWh (52). Puettmann et al. (2013) assume wood comes from intensively managed plantations, so we instead use the carbon intensity from forest operations (e.g. harvesting) described above. We assumed a density of 614 kg / m^3 of OSB, resins excluded (10). For substitution benefits and end-of-life, we make the same assumptions for OSB as for sawtimber products.

To model the production emissions for glue-laminated beams (GluLam) we rely on the LCA conducted by Bowers et al. (2017) (12). We use values associated with their Pacific Northwest facility because the wood species mix is most similar to California (true fir species and Douglas fir (*Pseudotsuga menziesii*)). We make three adjustments to the values they report. First, Bowers et al. (2017) assume electricity comes from the WECC in 2008, which was reported as 432g/kWh (52). We update this grid carbon intensity for the California grid. Second, we adjust the sawmill efficiencies for lamstock production used (50%) to be consistent with California sawmill efficiencies (75%) (49). Third, we replace the harvest and transport-to-mill emissions reported by Bowers et al. (2017) with GREET values, consistent with our other pathways. We assume mill residues are combusted for biopower, like our other structural wood product pathways. For substitution benefits and end-of-life, we make the same assumptions for GluLam as for sawtimber products.

Supplementary Results

Wood Availability

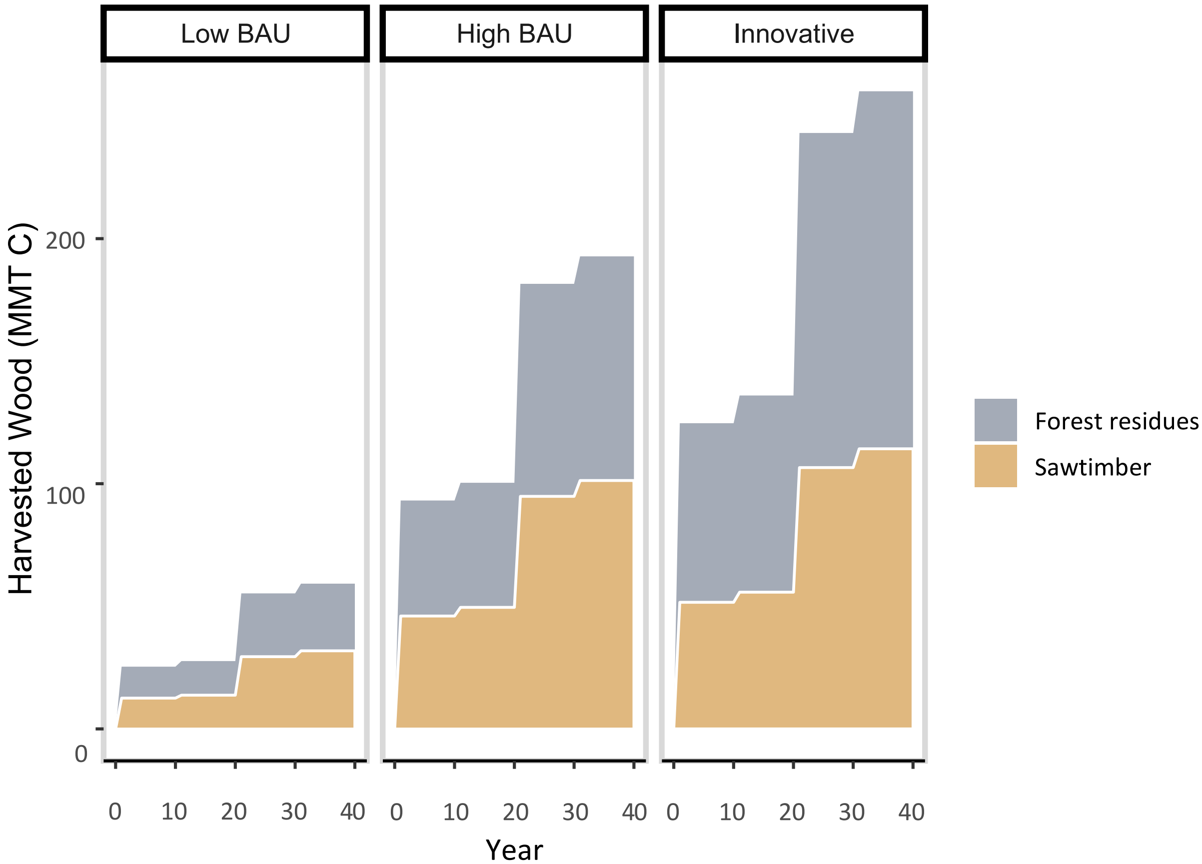

Figure S1. Cumulative harvested wood for three scenarios over 40 years, by wood class. Harvest timing is a function of both technical potential and policy constraints, including a 20-year enforced hiatus between harvest events. In Low BAU, we model management only on corporate land, where potentially profitable (net revenue >$2500/ha). In High BAU, we model management wherever it is net revenue positive with a delivered residue price of $0. In Innovative (IWP), we model management wherever it is net revenue positive with a delivered residue price of $100/ODT.

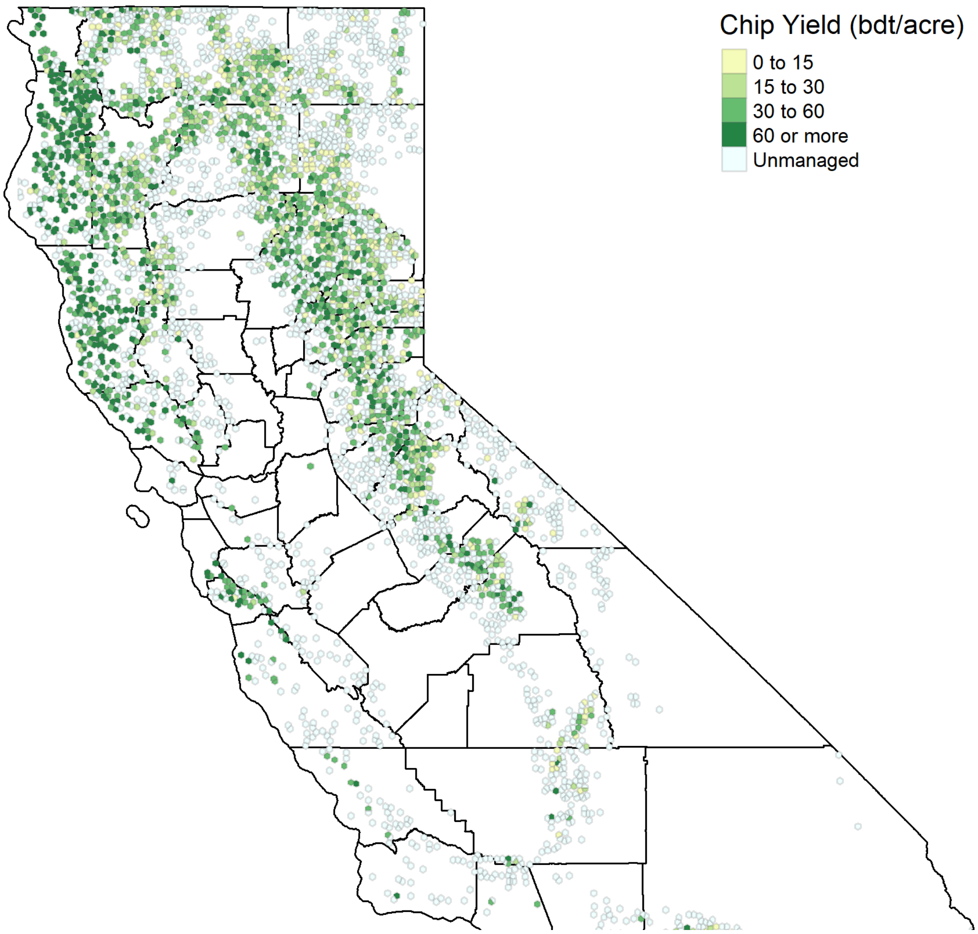

Figure S2. Forest residues produced over 40 years of management under the IWP scenario, with county boundaries shown. Each hexagon represents a single FIA plot, which is statistically representative of a larger area of forest (usually, ~2000-2500 ha).

WILDFIRE MODELING

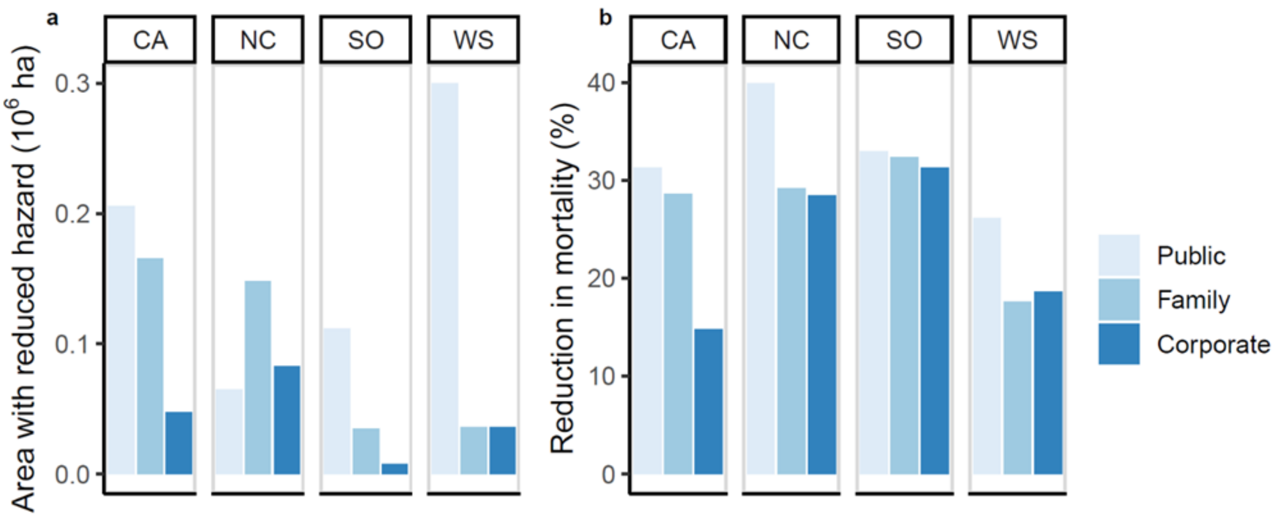

Figure S3. Reduction in stand replacing fire hazard by (a) area on which stand replacing fire may be avoided as a result of treatments and (b) mean reduction in predicted mortality under severe fire conditions in those stands (see SI Methods). Predicted mortality reduction is the difference between the percent of basal area that would die with and without treatment. Values are grouped by FVS Variants, where CA is Central California, NC is North Coast, SO is Northeast California, and WS is Western Sierra. Colors represent ownership groups, where 'Family' is non-corporate private land.

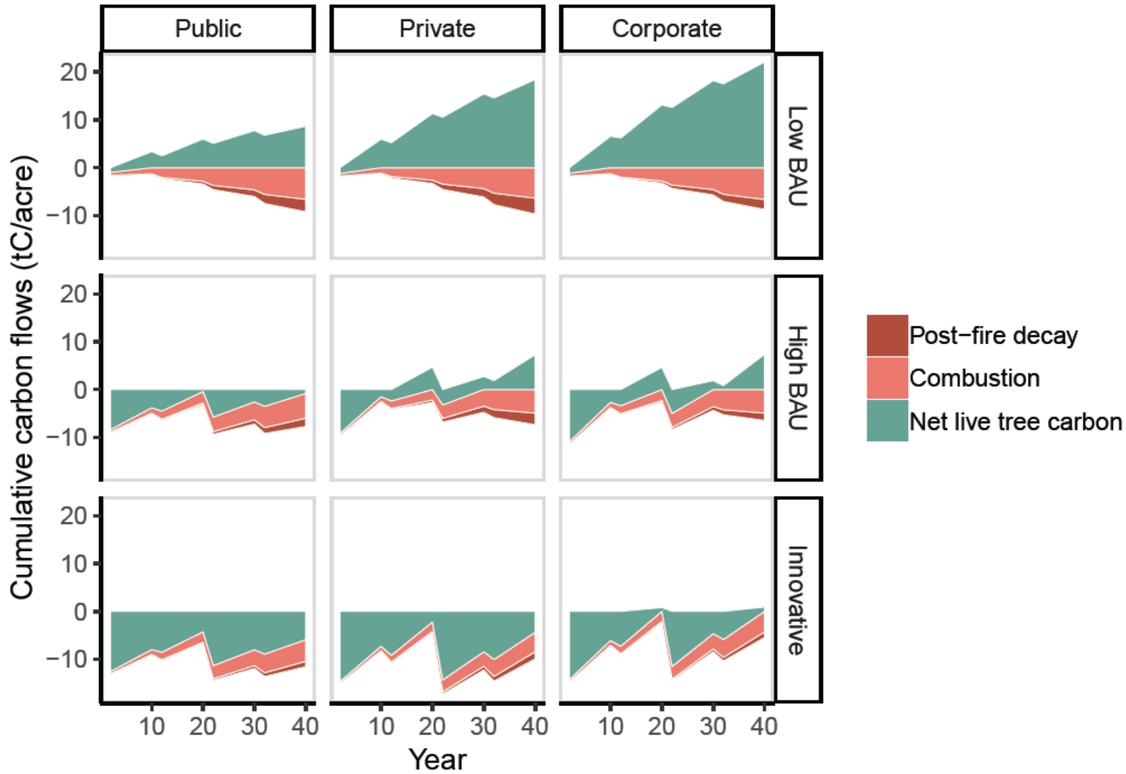

Figure S4. Cumulative in-forest carbon changes over 40 years across three scenarios and three owner groups. Net live tree carbon values are relative to carbon stocks in year zero, so negative values represent a carbon loss relative to year zero. Large changes in live tree carbon represent harvest events. In Low BAU, we model management only on corporate land, where potentially profitable (net revenue > $2500/ha). In High BAU, we model management wherever it is net revenue positive with a delivered residue price of $0. In Innovative (IWP), we model management wherever it is net revenue positive with a delivered residue price of $100/ODT. Treatment area under IWP defines the study area for High and Low BAU, which include untreated forest.

NET CLIMATE OUTCOMES, TECHNOECONOMIC ANALYSIS, AND ALTERNATIVE BAU

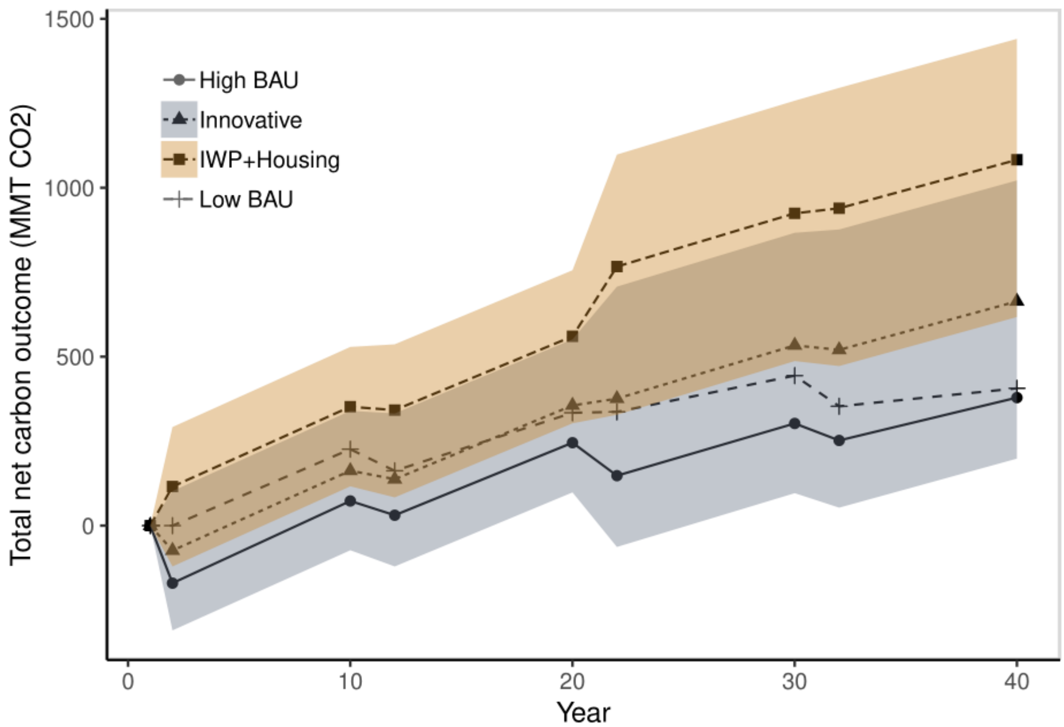

Figure S5. Net climate benefits for four scenarios, including changes to in-forest carbon and benefits from wood products. Shading represents the sensitivity of the two innovative scenarios to residue product choice. The minimum and maximum values represent the scenarios where all forest residues go to the product with the worst or best carbon outcomes, respectively, of all products examined (i.e. biopower or hydrogen + CCS). In Low BAU, we model management only on corporate land, where potentially profitable (net revenue > $2500/ha). In High BAU, we model management wherever it is net revenue positive with a delivered residue price of $0. In Innovative (IWP), we model management wherever it is net revenue positive with a delivered residue price of $100/ODT. In IWP+Housing, we assume additional sawtimber produced (over Low BAU) is used in multi-use and multi-family buildings.

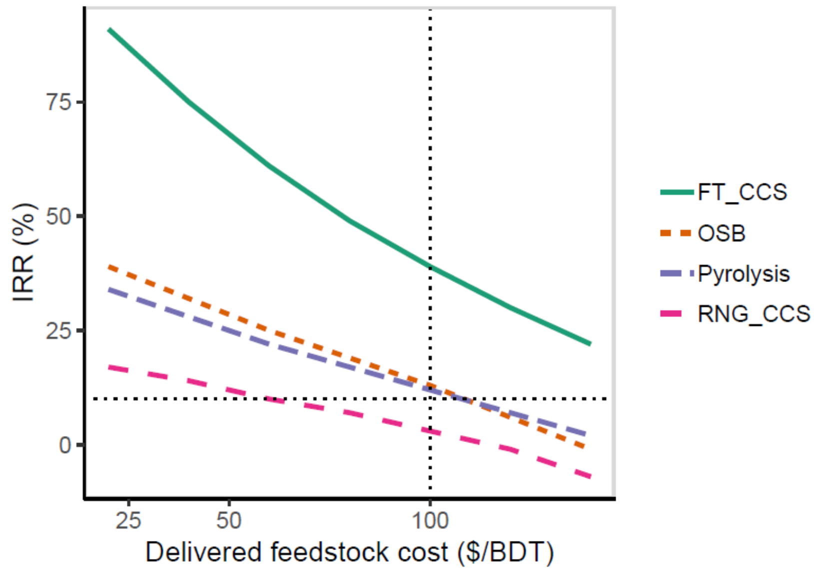

Figure S6. Technoeconomic assessment of select innovative wood technologies. Dotted lines represent the intersection of $100/ODT and an IRR of 10% (SI Methods). OSB is Oriented Strand Board. FT_CCS and RNG_CCS are Fischer-Tropsch and Renewable Natural Gas fuels with Carbon Capture and Storage.

Table S9. Mean harvested sawtimber flow and total area managed over 40 years by owner group in BAU-2.

Owner	Timber harvested (M ODT/yr)	Area managed (M ha)
Public	0.2	0.08
Corporate	0.9	0.65
Family	0.2	0.12

We investigate an alternative formulation of BAU, "BAU-2", with better representation of management on public and family forests. In BAU-2, we model active management on all lands where a net revenue of >$12,500/ha is possible without revenue from forest residues. This threshold yields timber volumes from public and private land that approximate those reported in (Christensen et al. 2019) for 2017 (Table S9). Under this management scenario, 1.3M ODT, or 1.4B board feet (BF), per year of saw timber are harvested over the next 40 years, on average. Comparing IWP to BAU-2, we find a net climate benefit of 6.4M tCO_2 per year when considering impacts from management, wildfire, carbon storage in products, and displacement of fossil-intensive alternatives over a 40-year period. For the Housing Scenario, the net climate benefit is 16.8M tCO_2 per year vs. BAU-2. This range of values is close to the range found when compared to Low BAU.

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