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A Synthesis of Harvested Wood Product Carbon Models

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Cover: Cross section of a log showing growth rings. Photo by Tryfonov/Adobe Stock.

Abstract

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This report is a synthesis of current harvested wood product (HWP) carbon models available for use in the United States and Canada. As local, state, and national entities develop forest management objectives that address climate change mitigation, there is a need to consider the short- and long-term fate of carbon in HWPs. The goals of this synthesis were to (1) review and synthesize the functionality of current HWP carbon models; (2) describe the role of life-cycle assessment (LCA) to estimate overall greenhouse gas (GHG) implications of using HWPs instead of alternative nonwood materials or fossil-fuel-based energy (i.e., substitution); and (3) discuss economic feedbacks of timber supply, demand, and price between HWPs and forest management. We summarized information based on discussions with experts, extensive literature reviews, and examples of ongoing research and methods for estimating HWP carbon. We also reviewed the potential for integrating forest sector economic models that consider carbon implications and the value of life-cycle assessments when considering cross-sectoral tradeoffs. The HWP carbon models most extensively described are the U.S. Department of Agriculture Forest Service's USFS HWP-C v1 and its variants as well as model variants built from the Government of Canada Natural Resources Canada Forest Service's abstract Network Simulation Engine (ANSE) software, such as the National Forest Carbon Monitoring, Accounting, and Reporting System for Harvested Wood Products. These models are similar in functionality and have been used at national and subnational scales; however, they also have key differences. Despite the data intensity and level of detail built into HWP carbon model frameworks, there are limitations to each one; some updates and improvements are being addressed but are not yet in the literature. Future model improvements could include improved integration of HWP models with forest ecosystem carbon models, forest sector economic models, and substitution analyses to analyze tradeoffs and quantify management implications across space and time.

KEYWORDS:
Carbon
Cascaded utilization
Forest carbon dynamics
Forest sector economic models
Harvested wood products (HWP)
Life-cycle assessment (LCA)
Substitution factors

Contents

Background and Purpose

Forest ecosystems capture, store, and release carbon dioxide (CO_2), which can affect concentrations of atmospheric CO_2 and their contribution to climate change. Carbon sequestered by forests is also stored in wood products after tree harvesting (Johnston and Radeloff 2019), so considerations of forest policies and management practices related to carbon usually consider the entire forest sector, comprised of forest ecosystems and the carbon removed from them (bolded terms are defined in the Glossary; acronyms are listed in app. 1). Given the long time spans and many variations in ecosystems, silvicultural systems, and wood products to be considered, understanding how climate change will affect in-forest carbon stores and weighing tradeoffs between forest carbon sequestration and carbon stored in **harvested wood products** (HWPs)[1] are usually explored using forest ecosystem and wood product utilization models.

Carbon in HWPs is a relatively small portion of the U.S. forest sector's carbon compared to forest ecosystem carbon (4.6 percent of the stocks, 14 percent of the net sequestration) (Domke et al. 2021), but there is growing interest in estimating the carbon that remains stored in HWPs and emissions over time from historical, current, and future harvests. About half of the dry weight of wood is composed of carbon. Improving forest carbon estimation and greenhouse gas (GHG) inventories will help define the potential climate mitigation benefits and emissions associated with HWPs. The length of time that HWPs remain in use (e.g., furniture or buildings), the energy required to create a product, and the fate of products (HWP disposal) determine the overall carbon benefits of HWPs. For example, HWPs that have a long functional life, such as lumber, may result in fewer GHG emissions over time than those with shorter functional lifespans (e.g., paper). Understanding how alternative forest management scenarios and HWP production, use, and disposal influence carbon storage, emissions, and forest conditions are key needs for forest managers and policymakers. Exploring these tradeoffs requires accurate carbon storage and emissions estimates in the forest sector. Forest sector carbon inventories and modeling can help nations, states, and other jurisdictions, as well as individual landowners, explore how forests can be leveraged for climate mitigation.

The purpose of this synthesis is to (1) review HWP models that track the forest removals of **biogenic carbon** (i.e., carbon derived from plants) and estimate stocks over time in HWPs and disposal sites; (2) describe the role of **life-cycle assessment** (LCA) to estimate overall GHG implications of using HWPs instead of alternative nonwood materials or fossil-fuel-based energy; and (3) discuss economic feedbacks of timber supply, demand, and price between HWPs and forest management. Carbon fluctuations in the forest ecosystem and HWPs, including the emissions associated with the production, use, and disposal of HWPs, as well as carbon tradeoffs with more fossil-fuel-intensive products, can provide a more complete whole-sector assessment for possible climate change mitigation strategies. Economic analyses can further demonstrate the feasibility, or lack thereof, of different mitigation strategies (e.g., Xu et al. 2018). The information in this synthesis is aimed at guiding interested users in assessing model capabilities and limitations and choosing appropriate models for the forest sector carbon questions at hand.

Intergovernmental Panel on Climate Change HWP Estimation Approaches

The United Nations Framework Convention on Climate Change (UNFCCC) created an international treaty aimed at stabilizing atmospheric greenhouse gas (GHG) concentrations "at a level that would prevent dangerous anthropogenic (human-induced) interference with the climate system" (UNFCCC 2023), which was later operationalized by the 1997 Kyoto Protocol (UNFCCC 1998). Although the UNFCCC determines requirements for reporting GHG inventories, the Intergovernmental Panel on Climate Change (IPCC) provided guidance for GHG estimation approaches and methods, including for HWPs (IPCC 1997). Not until several years later was guidance containing conceptual frameworks, approaches, and estimation methods included to consider the carbon storage in HWPs and subsequent release as CO_2 (IPCC 2000, 2003, 2014; Pingoud et al. 2006; Rüter et al. 2019). These estimates were to be reported in the land use, land-use change, and forestry sector (LULUCF).

[1] Although sometimes referred to as "forest products," this synthesis uses the Intergovernmental Panel on Climate Change term "harvested wood product" (HWP) to capture all forest biomass, including tree bark, that is removed from forests (Pingoud et al. 2006).

Details for HWP carbon estimation for different levels (e.g., tiers 1 to 3) vary depending on the specific year of the IPCC guidance documents mainly because of global differences in available harvest and HWP utilization data (IPCC 1997, 2003, 2014; Pingoud et al. 2006; Rüter et al. 2019). Earlier tier 1 default recommendations assume that carbon in wood products harvested from the forest is immediately oxidized (i.e., an immediate emission to the atmosphere) (IPCC 1997, 2003). IPCC guidance documents for tier 1 and 2 methods were updated later (IPCC 2014, Pingoud et al. 2006). The 2019 refinement (Rüter et al. 2019) provided further clarifications about HWPs; tier 1 methods contain IPCC-defined default global algorithms and parameters to estimate carbon in HWP pools, whereas tier 2 combines some country-specific activity data or emissions factors with IPCC defaults, and tier 3 applies country-specific data as well as algorithms. Rüter et al. (2019) provided an updated decision tree on the choice of the level of complexity to use (tiers 1 to 3) for estimating HWP carbon. This report predominantly focuses on the United States and Canada, which use tier 3 methods based on detailed country-specific methods and data.

An "approach" in this synthesis refers to the conceptual framework and system boundaries for estimating CO_2 emissions and removals arising from HWPs. Some conceptual frameworks are **pool-based**, with a focus on storage in HWP carbon pools. Pool-based estimates imply net emissions and removals of CO_2 to and from the atmosphere through changes in carbon stocks in each pool. The framework can also be **flux-based**, which estimates HWP CO_2 fluxes to and from the atmosphere (Rüter et al. 2019). Both pool- and flux-based frameworks include biogenic carbon emission estimates from wood used for energy either implicitly, because harvested wood burned for energy is absent from both forest ecosystem and HWP carbon pools, or explicitly, because it is estimated as fuelwood emissions.

The **system boundary** defines which components of a system are included or excluded. Defining system boundaries can ensure transparency, completeness, and consistency in calculations and reported estimates (Rüter et al. 2019). This is particularly important when estimating HWP carbon because it also determines which storage pools, emissions categories, and imported or exported HWPs are included. The term "estimation method" (or simply "method") refers to the set of calculations used to estimate total carbon stock and annual stock change in HWP pools, or annual net flux with the atmosphere. The IPCC guidance explicitly describes the system boundary and conceptual framework for each approach (i.e., estimated harvest inputs and pools), while HWP carbon estimation methods describe the calculations to produce the estimates. Brief descriptions of the system boundaries and conceptual frameworks associated with different IPCC approaches for estimating carbon associated with HWPs (Rüter et al. 2019, Sato and Nojiri 2019) are provided in table 1.

There are key differences among approaches. The "simple decay" approach, for example, has the same system boundaries as the "production" approach and will yield identical net estimates of HWP impacts on the atmosphere. Under the production approach, HWP carbon is transferred out of the forest, contributing to a reduction in forest ecosystem carbon stocks, but the reduction is subsequently offset by a "sink" (referred to as "net

Table 1—Four different Intergovernmental Panel on Climate Change (IPCC) approaches for estimating carbon associated with harvested wood products

IPCC approach	System boundary	Exports included	Imports included	Conceptual framework
Stock-change	All harvested wood product pools consumed within country	No	Yes	Pool based
Production	All harvested wood products originating from within country included	Yes	No	Pool based
Atmospheric flow	Greenhouse gas flow between atmosphere and harvested wood products consumed in country	No	Yes	Flux based
Simple decay	Greenhouse gas flow between atmosphere and harvested wood products produced in country	Yes	No	Flux based

uptake") in HWP pools. Carbon is emitted from HWP pools through waste incineration or decay (with carbon retention times depending on the type of product) (e.g., US EPA 2021). Under the simple decay approach, harvest is represented as a lateral transfer of carbon from forests to the product sector, where carbon will be stored and subsequently released through burning (typically in the year of harvest) or decay (e.g., ECCC 2021). Thus, HWPs are not a sink but a carbon store that extends the retention time of carbon that was removed from the atmosphere by the forest.

Guidance from the IPCC (IPCC 2003, 2014; Pingoud et al. 2006) and Rüter et al. (2019) does not require a reporting country to follow a single approach, which may lead to carbon storage and emissions being double counted or not being accounted for at the global level. Under the rules associated with the Paris Agreement (UNFCCC 2015), the most recent international treaty addressing global climate change, if an approach other than the production approach is used in certain countries, then the production approach is requested as supplemental information (Sato and Nojiri 2019). These are important concepts for subnational analyses because HWPs are transported across jurisdictional boundaries. For example, under the atmospheric flow approach, the emissions of HWPs for exported wood are not counted in the producing country, but the emissions of imported HWPs are counted (Skog 2008). The IPCC guidelines specifically pertain to national greenhouse gas inventories for countries participating in international climate change treaties; however, subnational forest sector carbon inventory efforts can also follow IPCC guidance (CARB 2018a; Christensen et al. 2019a, 2019b). Carbon estimation in HWPs is especially challenging at state scales because availability of empirical data on utilization, disposal, and movement of HWPs is often lacking. Furthermore, not all provinces or states in a country produce HWPs, yet estimating HWP carbon storage and emissions would be relevant for any jurisdiction considering policies that encourage increasing the lifespan of wood, using wood for bioenergy, or exploring strategies aimed at decreasing carbon emissions in the building sector by using more wood materials. Thus, estimating HWP carbon dynamics over large geographic regions can be challenging because of differences in policy objectives, approaches for estimating carbon, and export practices.

Estimating carbon stocks in forest ecosystem pools is also a separate process from estimating carbon in HWP pools, even though both are included in the IPCC forest sector. It is important to consider HWP stocks and stock changes in the context of ecosystem removals and overall forest carbon flux to better understand the full picture. For example, well-managed, highly productive forests with high levels of harvest can have low forest carbon stocks but net positive carbon sequestration in both the HWP and ecosystem pools. Conversely, forests with low harvest levels and high forest carbon stocks can have net carbon losses in the HWP and ecosystem carbon stocks because of drought and disturbances, such as wildfires or insect outbreaks (Christensen et al. 2021). It is also critical that linked forest ecosystem-HWP analyses align to avoid double counting of carbon stocks or losses or missing carbon stocks and losses altogether. For example, double counting can occur when estimating bark used for bioenergy as an emission from HWP pools if it has already been included as an emission from the forest ecosystem.

Overview of HWP Carbon Estimation Models

Terminology used in HWP carbon storage and emissions models varies and can be unclear in the literature. HWP carbon estimates have been referred to as carbon models, carbon accounting, and carbon inventories (Jasinevičius et al. 2015). HWP carbon estimation models are used to track, monitor, and estimate carbon in woody biomass removed from the forest and the amounts that end up stored in harvested wood **products in use** or at **solid waste disposal sites** (SWDSs) and later emitted to the atmosphere at the end of a wood product's life. These same models have been used for national, state, and other jurisdictions to establish HWP carbon inventories or for other objectives, such as calculating future HWP carbon outputs based on alternative policy, management, or utilization scenarios. These models are typically designed to follow the IPCC guidance and approaches described above. However, there is no single modeling approach or software that addresses all the elements relevant to understanding the climate impact of HWPs, such as methane (CH_4) emissions associated with HWPs decaying in landfills, which are typically accounted for in the IPCC waste sector.

The HWP carbon estimation models and software reviewed in this synthesis are not comprehensive (table 2). Models or software that are listed are either commonly used in North America or have been identified in existing literature syntheses as being most user-friendly and closest to reality (Brunet-Navarro et al. 2016, Jasinevičius et al. 2015). As new models were created over the years, they have also been updated to reflect changes in IPCC guidelines. The U.S. Department of Agriculture (USDA) Forest Service created several tools over the years with the ability to estimate carbon storage and emissions associated with HWPs. Some of these tools may be more straightforward for smaller scale applications (e.g., stand level or specific ecological zone). The Harvest Carbon Calculator, for example, can be used at the stand level and is designed to be user friendly, with few data requirements. The minimum inputs for the Harvest Carbon Calculator are harvest volume, wood type, and geographic region (Hoover et al. 2014). The Harvest Carbon Calculator provides estimates for carbon storage in wood

Table 2—Examples of harvested wood product (HWP) carbon estimation models and software

Model or software	Spatial scale and use	Accessibility	Description
CO2Fix, (Eggers 2002, Schelhaas et al. 2004)	Carbon project scale with HWP component Predominantly used in Europe	Open access http://dataservices.efi.int/casfor/models.htm	IPCC stock-change approach. Tracks carbon stocks and stock changes over time in forest ecosystems. HWP component with product, end-of-life, disposal tracking. Contains both bioenergy and financial modules.
U.S. EPA State Inventory and Protection Tool (Birdsey and Lewis 2003, US EPA 2017)	State scale for land use, land-use change, and forestry reporting	Open access to model spreadsheet https://www.epa.gov/statelocalenergy/state-inventory-and-projection-tool	Calculates net carbon flux from forest sector, including HWPs in landfills (as net estimate). Used in Birdsey and Lewis (2003) for default HWP carbon estimates. Uses HARVCARB model developed by Row and Phelps (1991). Contains national-scale defaults when state-level data are not available.
WOODCARB II Excel® spreadsheet model (Skog 2008, Skog and Nicholson 2000)	Used for U.S. NGHGI national inventory report requirements to the UNFCCC USDA Forest Service Northern Research Station and Forest Products Laboratory carbon science group is evaluating feasibility of improving model by adding state-level data (Nichols 2021)	Available for internal USDA Forest Service use only	IPCC atmospheric flow, stock change, and production approaches. Estimates HWP carbon stocks from 1900 to current day and stock changes for products in use and at SWDSs and emissions with and without energy capture. Builds off methods and datasets provided in Smith et al. (2006).

Table 2 (continued)—Examples of harvested wood product (HWP) carbon estimation models and software

Model or software	Spatial scale and use	Accessibility	Description
Forest Vegetation Simulator (Hoover and Rebain 2011, Smith et al. 2006)	U.S. individual tree- and stand-level model	Open access https://www.fs.usda.gov/fvs/software/index.shtml Training recommended before use	Forest growth simulation model with harvested carbon report. IPCC production approach for HWPs. Allocates carbon harvested for HWPs according to Smith et al. (2006) methods to estimate HWP stocks for products in use, and in SWDSs, and emissions with or without energy production from time of harvest and forward. Contains extensions for disturbances and economic analyses.
Harvest Carbon Calculator in iTree (Hoover et al. 2014)	Available for 10 U.S. ecoregions	Open access https://harvest.itreetools.org	Formerly known as Product Estimation Tool Online (PRESTO). Designed to help U.S. land managers and landowners estimate carbon in HWPs after 100 years as well as annual average storage. Estimates based on wood type, harvest volume, and geographic region relying on methods from Smith et al. (2006).
USFS HWP-C v1[a] (Anderson et al. 2013; Butler et al. 2014; Loeffler et al. 2014; Stockmann et al. 2012, 2014)	USDA Forest Service national forest and regional model Has been applied/modified for state-level and other user-defined analyses	Not currently available online USDA Forest Service-led California variant (CA USFS HWP-C) model undergoing updates and expected to be available publicly as a new national version (USFS HWP-C v2) in 2024, coded in Python	IPCC production approach; model outputs can be used to accommodate IPCC simple-decay approach as well. Tracks cumulative HWP carbon stocks from first recorded National Forest System harvests (1906) to current day and annual HWP carbon stock changes for products in use, and in SWDSs, and emissions with and without energy capture. Relies on methods and data from McKeever (2009), McKeever and Howard (2011), Skog (2008), Skog and Nicholson (1998, 2000), and Smith et al. (2006). Users can provide their own data to run analyses. Refer to figure 1 for a depiction of the evolution of this model.

Table 2 (continued)—Examples of harvested wood product (HWP) carbon estimation models and software

Model or software	Spatial scale and use	Accessibility	Description
HWP-C vR (Groom and Tase 2022)	State-led (California and Oregon) effort that modified USDA Forest Service-led California variant (CA USFS HWP-C) of the agency's HWP-C v1 for use in state-level inventories Can be used at any scale	Free access online https://groomanalyticsllc.shinyapps.io/HWP-C-vR Coded in R User guide provided Training advised	IPCC production approach; model outputs can be used to accommodate IPCC simple-decay approach as well. Tracks cumulative HWP carbon stocks from historical and current harvests based on user-provided data and annual HWP carbon stock changes for products in use, and in SWDSs, and emissions with and without energy capture. Relies on methods and data from McKeever (2009), McKeever and Howard (2011), Skog (2008), Skog and Nicholson (1998, 2000), and Smith et al. (2006). Users can provide their own data to run analyses. Refer to figure 1 for a depiction of how this model evolved from USFS HWP-C v1/CA USFS HWP-C.
NRCan ANSE version 1.0 software[b] (CFS 2022)	ANSE software is used to create and run a variety of HWP carbon estimation models[c]	ANSE v1.0 and user guide available for download by contacting Natural Resources Canada (NRCan) Requires training	Formerly Carbon Budget Modeling Framework for HWPs (CBM-FHWP). Any IPCC approach. ANSE is a wood products modeling framework or software system. Tracks HWP carbon storage and emission from historical harvests to current day and annual net flux to the atmosphere. Can be paired with ecosystem models, such as Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3), to estimate HWP carbon associated with future forest management and HWP utilization scenarios.

Table 2 (continued)—Examples of harvested wood product (HWP) carbon estimation models and software

Model or software	Spatial scale and use	Accessibility	Description
BC-HWP v1 (Dymond 2012)	Constructed from ANSE v1.0 Represents British Columbia harvest within the North American wood market Can be used at the provincial or project scale	Free access to a project-scale calculator and user guide Calculator and user guide available at: https://www2.gov.bc.ca/gov/content/environment/natural-resource-stewardship/natural-resources-climate-change/natural-resources-climate-change-mitigation/tools-resources	Province-wide model estimates stocks and emissions from all wood harvested in British Columbia and used in North America from 1965 to 2065. Includes substantial new datasets on building half-lives. Includes dumps and landfill storage and landfill-gas management. Project-scale calculator allows architects, homeowners, etc. to enter wood products in common units, back calculates to include manufacturing emissions, and provides life-cycle estimates.
National Forest Carbon Monitoring, Accounting and Reporting System (NFCMARS-HWP) (ECCC 2021)	Primary model constructed from ANSE v1.0 used for the forest sector monitoring, inventorying, and reporting system in the Canadian national inventory report to the UNFCCC (ECCC 2021)	Available for download by contacting NRCan Requires training	IPCC simple decay approach. Tracks HWP carbon storage and emission from historical harvests to current day and annual net flux to the atmosphere. Can be paired with ecosystem models, such as CBM-CFS3, to estimate HWP carbon associated with future forest management and HWP utilization scenarios.
MitigAna (Xie et al. 2021)	Constructed from ANSE v1.0 Used for mitigation analyses in Canada	Not publicly available	Tracks HWP carbon storage and emission from historical harvests to current day and annual net flux to the atmosphere. Can be paired with ecosystem models, such as CBM-CFS3, to estimate HWP carbon associated with future forest management and HWP utilization scenarios. Permits users to undertake mitigation analyses for different HWPs. This model includes modules with the ability to calculate substitution benefits and cascading uses.

[a] Model name given in this synthesis.

[b] HWP carbon modeling software and many functionalities are possible that may not currently be used in every ANSE-based model.

[c] For example, Dymond (2012), ECCC (2021), Smyth et al. (2013), and Xie et al. (2021).

ANSE = Abstract Network Simulation Engine; EPA = Environmental Protection Agency; HWP = harvested wood products; IPCC = Intergovernmental Panel on Climate Change; NFCMARS = National Forest Carbon Monitoring, Accounting and Reporting System; NGHGI = national greenhouse gas inventory; SWDS = solid waste disposal site; UNFCCC = United Nations Framework Convention on Climate Change; USDA = U.S. Department of Agriculture.

products after 100 years as well as annual average storage based on USDA Forest Service Forest Inventory and Analysis (FIA) data and HWP carbon estimation methods described in Smith et al. (2006), which is one of two primary documents that guides methods and provides datasets for HWP carbon estimation in the United States. More information on Smith et al. (2006) can be found in appendix 2. The Harvest Carbon Calculator is limited because it applies only to the Western United States.

Another USDA Forest Service tool, the Forest Vegetation Simulator (FVS), is a data-intensive, nonspatial forest ecosystem model that runs at the stand level and simulates individual tree dynamics. Users can specify a variety of tree removal simulations resulting in merchantable volume model outputs. Within FVS is a "stand carbon report" that provides forest ecosystem carbon estimates. and a "harvested carbon report" that provides estimates for carbon storage for HWPs in use, carbon storage in landfills, and emissions associated with or without energy capture (Hoover and Rebain 2011). These estimates are also calculated following the methods and parameters in Smith et al. (2006). FVS allows users to explore how the carbon in HWP pools from simulated harvest shift over time into the future. In FVS, carbon in HWP pools are a result of simulated harvest scenarios and not empirical harvest data. This makes it difficult to use FVS as an inventory tool for HWP carbon estimation at a national or state scale using harvest data. Regional wood utilization datasets are integrated into the model (Smith et al. 2006) to estimate HWP carbon pool calculations, so changing parameters with newer or regionally specific datasets is not possible at this time. Although it would be challenging, FVS could be used for forest ecosystem and HWP carbon inventories by calibrating the model with FIA data and simulating current or historical management activities and disturbances using activity data. This would require extensive input datasets and would result in a modeled rather than measured inventory (e.g., Maritato et al. 2021). This approach is also limited by the difficulty of applying the ecosystem model at large scales. Hoover and Rebain (2011) suggested that FVS is not appropriate for "entity-wide carbon accounting" purposes; FVS was designed as a growth and yield model rather than a carbon estimation model and may leave out some carbon pools of interest.

Carbon can remain stored in HWPs for decades or longer, which means that HWP carbon pools contain carbon from historical harvests. While the Harvest Carbon Calculator and FVS focus on projecting carbon storage from current harvest into the future, HWP carbon estimation for national and subnational GHG inventories are centered on understanding carbon stocks, stock changes, and emissions from present-day HWP pools that are the result of previously completed harvests (i.e., what remains in those pools and the rate of carbon being added to them). Consequently, the USDA Forest Service also developed a Microsoft®[2] Excel® (hereafter Excel) spreadsheet model called WOODCARB to provide cumulative HWP carbon storage and emissions estimates from historical (dating back to 100 years) to current harvests (Skog 2008). A more recent version of this model (WOODCARB II) is used to estimate HWP carbon stocks and stock changes in the *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (the national greenhouse gas inventory) (US EPA 2021) as part of the U.S. national inventory report to the UNFCCC (US EPA 2021). WOODCARB II is an Excel spreadsheet model, but it is not publicly available and has not yet been used for individual states.[3] The Forest Service is currently exploring how to downscale national estimates for use in individual states (Domke 2021).

One tool currently available for U.S. state-level forest ecosystem and HWP carbon estimation is the U.S. Environmental Protection Agency (EPA) State Inventory Tool (US EPA 2017). The State Inventory Tool combines carbon stored in products in use and those that have been disposed of in landfills. These estimates are then combined with other forest ecosystem carbon pools, such as aboveground biomass, for either a net "emissions" or net "sequestration" estimate (US EPA 2020a). Unless states have their own HWP data to use in the State Inventory Tool, default data inputs come from Birdsey and Lewis (2003), which contains estimates for individual

[2] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

[3] A research group at Appalachian State University converted the Excel version of WOODCARB II to a version called WOODCARB3R in an R package platform (available at: http://benjones2.github.io/WOODCARB3R). This version enhances the capability for data manipulation; analysis such as uncertainty analysis, sensitivity analysis, alternate model dynamics, and alternate pathways; reporting; and integration with other datasets.

state HWP carbon for products in use and landfills from 1987 through 1997 (Denny 2021). Many of the underlying assumptions and inputs in the State Inventory Tool come from older datasets that have since been updated but have yet to be included in the tool.

Other models have been useful for differing spatial scales; for example, CO2Fix was originally designed for the Finnish forest sector. However, others developed the European Forest Information Scenario Model for Europe's forest sector (Eggers 2002, Schelhaas 2004). Although user-friendly and available online, CO2Fix is primarily designed for carbon credit programs, such as agroforestry and afforestation projects (e.g., Panwar et al. 2017). CO2Fix has yet to be used for analyses in the United States.

Another USDA Forest Service tool, USFS HWP-C v1, was originally developed for regional-scale and forest-level analysis of HWP harvested from federal public lands in the National Forest System (NFS) (e.g., Anderson et al. 2013; Stockmann et al. 2012, 2014). Like WOODCARB II, this model relies on similar methods and datasets and calculates cumulative HWP carbon storage and emissions from historical and current harvests. Variants of this model (e.g., CA USFS HWP-C and HWP-C vR) have since been created for state-level assessments and include options to estimate HWP carbon across multiple landowner types in California, Oregon, and Washington (e.g., Loeffler et al. 2019, Morgan et al. 2021, Nichols et al. 2020).

In Canada, the Natural Resources Canada Abstract Network Simulation Engine (ANSE) software (Government of Canada 2023), formerly the Canadian Carbon Budget Model Framework for Harvested Wood Products (CBM-FHWP), has been used to develop a variety of HWP carbon estimation models. ANSE is unique because it is a software package that provides building blocks to model HWP carbon following IPCC approaches or estimation methods. The National Forest Carbon Monitoring, Accounting, and Reporting System for Harvested Wood Products (NFCMARS-HWP) is the main model constructed from ANSE used for the Canadian national inventory report to the UNFCCC (ECCC 2021). Variants built from ANSE have also been created and parameterized for several U.S. states, including South Carolina (Dugan et al. 2018) and Vermont (Dugan et al. 2021), with others (Oregon and California) in progress (American Forests 2021). These state-level efforts are aimed at carbon storage and emissions analyses to develop climate mitigation policies.

All models have limitations and uncertainties. Some models directly incorporate parameters from older assumptions and empirical data without the option to update their values, which may have limited utility (e.g., Harvest Carbon Calculator and FVS both use Smith et al. 2006). This could become especially apparent as climate change influences future forest growth, harvest practices, and wood products utilization. Among the models briefly described above, only USFS HWP-C v1 and some ANSE variants have been parameterized and used at the U.S. state level (e.g., Dugan et al. 2021, Loeffler et al. 2019, Morgan et al. 2021, Nichols et al. 2020). This synthesis focuses primarily on HWP models used in the United States and Canada, using USFS HWP-C v1, NFCMARS-HWP, and related variants as the dominant examples.

Comparison of USFS HWP-C v1, NFCMARS-HWP, and Related Variants

Different HWP models may generally follow similar pathways for allocating carbon removed from the forest ecosystem and tracking carbon storage and emissions resulting from use and disposal; however, the details of these pathways can vary depending on the chosen IPCC approach, tier, and available data. Types of HWPs, the functional lifespan of products before they are disposed of, as well as the manner in which they are disposed of (e.g., burned, composted, recycled, dumped, or landfilled) determine when and how carbon storage and emissions associated with HWPs occur.

Generally speaking, NFCMARS-HWP and variants constructed from ANSE software, as well as USFS HWP-C v1 and its variants, use similar system boundaries and are complex and data intensive. They have similar data requirements and follow similar processes to track HWP carbon pools as products move through their functional life and disposal. However, there are slight differences in the pathways for carbon flow and how underlying datasets are applied throughout the models. It is difficult to know how these differences influence overall HWP carbon storage or emissions estimates because there have not yet been any studies comparing outputs from these two models using the same dataset. The following subsections review the transfer of carbon from time of harvest to disposal of products in more detail, using USFS HWP-C v1 and its variants, NFCMARS-HWP, and other variants built from ANSE software as examples.

USFS HWP-C v1 Model Description

USFS HWP-C v1 was initially an Excel spreadsheet model developed by the USDA Forest Service for an NFS HWP carbon inventory in the USDA Forest Service's Northern Region (Anderson et al. 2013, Stockmann et al. 2012). The Excel model was later coded into the USFS HWP-C v1 model for all NFS regional HWP carbon inventories (e.g., Stockmann et al. 2014). The model was then modified and became CA USFS HWP-C, the California variant of USFS HWP-C v1, for use in the California state-level inventory for all forest land in California (Loeffler et al. 2019). CA USFS HWP-C was further modified by the California Department of Forestry and Fire Protection and Oregon Department of Forestry, as described in Christensen et al. (2019a, 2019b), for state-level inventories in California, Oregon (Morgan et al. 2021), and Washington (Nichols et al. 2020). This state-level variant of the model is referred to in this synthesis as HWP-C vR (Groom and Tase 2022). Although it is almost identical to CA USFS HWP-C, it is not a USDA Forest Service product and diverges from its agency-led efforts. USDA Forest Service-led efforts continue to expand upon the original model and additional features included in the California variant, with USFS HWP-C v2 anticipated sometime in 2024 (Stockmann 2022). The specific IPCC approach and estimation methods used in USFS HWP-C v1 and its variants are discussed in this section. On a technical level, modifications can be made to accommodate different IPCC approaches, estimation methods, or other features through consultation with the model creators. Detailed documentation for HWP-C vR model function and underlying datasets is provided in Groom and Tase (2022), most of which also applies to USFS HWP-C v1. Underlying R code for HWP-C vR is available for download for users to use and modify on their own. Figure 1 depicts the evolution of USFS HWP-C v1.

USFS HWP-C v1 calculates cumulative HWP carbon storage and emissions from historical and current harvests. The model and its variants are largely based on methods and datasets presented in McKeever (2009), McKeever and Howard (2011), Skog (2008), Skog and Nicholson (1998, 2000), and Smith et al. (2006). Parameters for the model were intended for large-scale forested landscapes

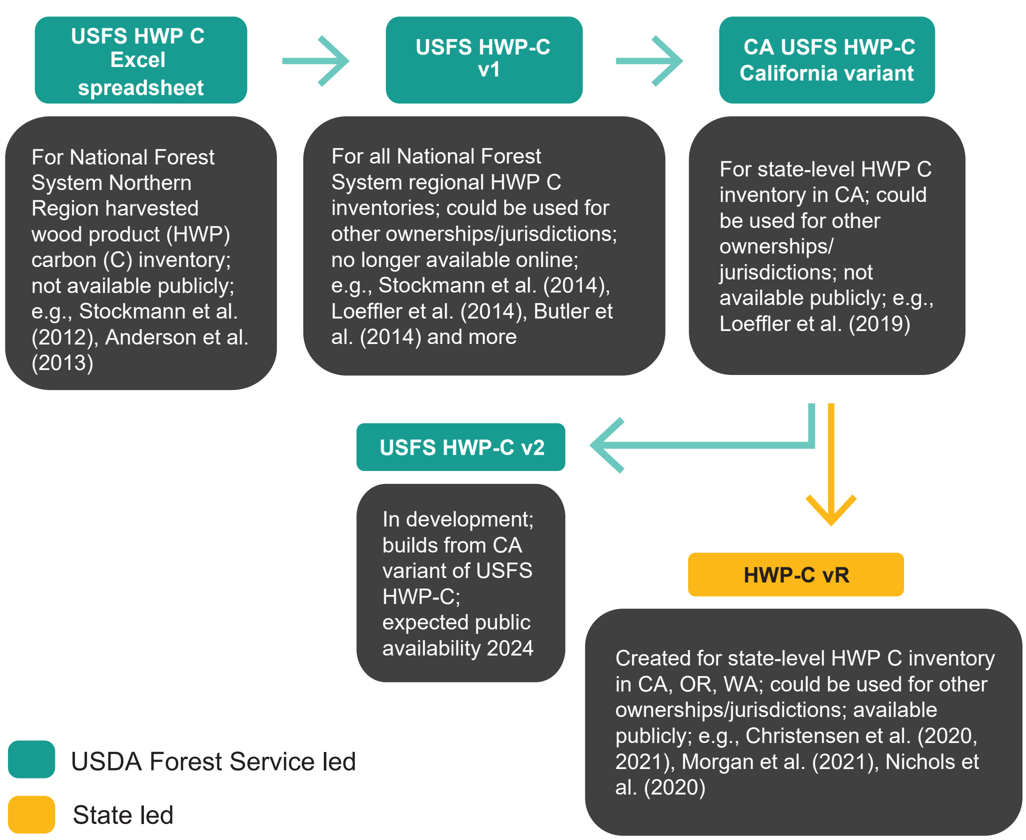

Figure 1—Evolution of the USFS HWP model. Before this analysis, the model did not have a name; "USFS HWP-C" was created in this analysis. CA = California, OR = Oregon, WA = Washington.

in the United States (e.g., NFS regions) with annual records of timber harvest and primary product output. In cases where annual records were incomplete, which was typically the case for early 20^{th} century records, assumptions were made about the distribution of the harvest into different timber and primary product classes.

Carbon estimation in USFS HWP-C v1 begins with **roundwood** (i.e., industrial roundwood and fuelwood) harvest volumes that users provide from harvest records. In the Forest Service NFS inventories, harvest records came from national forest or regional harvest records, including cut-and-sold reports (USDA FS 2022). In the state-level analyses (CA USFS HWP-C or HWP-C vR) that include all forest landownerships, harvest records came from FIA Timber Products Output data, county records, and other data sources (e.g., Simmons et al. 2021, UM-BBER 2021, USDA FS 2012).

The model can run a single year of harvest or multiple years at once, depending on what the user wants to evaluate. USFS HWP-C v1 and its variants follow the IPCC production approach. The system boundary for this approach includes all wood products produced from wood cut within a geographic area—exported HWPs are included, but imported HWPs are excluded. Because the model was originally designed for national forests, and raw timber from NFS lands cannot be exported except for the Tongass National Forest in Alaska, any exported HWPs are assumed to follow the same utilization pathways as domestically consumed HWPs. This is currently the assumption in the state-level analyses as well, where raw timber exports from other ownerships occur.

Annual harvest volumes are allocated to 1 of 40 historical and current **timber product classes** (e.g., sawlogs, nonsawtimber, pulpwood, fuelwood, etc.) using **timber product ratios,** which are based on, and in some cases, extrapolated from USDA Forest Service data or state-level utilization data. For example, in the case of U.S. national forests, harvest records partition the harvest into various timber product classes back to 1980, making ratios relatively simple to calculate; however, calculations for harvest years prior to 1980 must use more generalized timber product distributions. Users must provide annual harvest volumes and annual timber product ratios. **Primary product ratios** are then applied to timber product classes to allocate volumes to 1 of 64 historical and current **primary product classes** (e.g., lumber, plywood, panels, wood pulp, fuelwood, etc.), also referred to as "secondary" products by the IPCC (Pingoud et al. 2006). Primary product ratios in USFS HWP-C v1 can be selected from default primary product ratios for U.S. regions (common forest types within 10 U.S. regions) specified in Smith et al. (2006). These defaults have not been updated since the model was first created, but research is underway to update regional defaults as development of USFS HWP-C v2 continues. Alternatively, more geographically specific or recent data can be used if available, such as state-level FIA Timber Product Outputs Survey data (e.g., Marcille et al. 2020, Simmons et al. 2016). In CA USFS HWP-C, detailed state-specific primary product ratios were developed for California. With the release of the HWP-C vR model variant, original default templates from USFS HWP-C v1 were provided as well as detailed state-specific primary product ratios through 2019 for California, Oregon, and Washington state-level inventories, with more updates likely in the future (Groom and Tase 2022).

When debarked sawlogs are processed into primary products, they generate a significant amount of mill residues in the form of sawdust, shavings, and woodchips. Mill residues are the proportion of harvest volume or sawlogs that are removed from the forest, which can be used as feedstock for generating energy, or for manufacturing wood pulp or wood-based panels. In some states, little mill residues go unused, and in some mills, these residues provide a significant source of process energy (e.g., heat and electricity) for mill operations, such as lumber drying, or excess electricity that can be returned to local power grids (Donahue et al. 2021, Morgan and Donahue 2019). Primary product ratios used in USFS HWP-C v1 and its variants distribute mill residues into four primary product classes: fuelwood, wood pulp, oriented strand board (OSB), and nonstructural panels. This is in addition to any roundwood that is used in these primary product classes.

Once all primary product volumes are established for the associated timber harvest, product volumes are converted to carbon mass using conversion factors from table D1 in Smith et al. (2006). Primary product volume to carbon mass conversion factors in Smith et al. (2006) are based on a cubic foot per unit of primary product (e.g., thousand board feet or thousand square feet) conversion factor, a pounds-per-cubic-foot conversion factor to provide the weight of the product in pounds, a factor for the fraction of the product that is wood fiber, and a factor for the fraction of carbon in wood. When applied all together, the result is conversion factors of U.S. short tons or metric tons of carbon per unit of primary product. Smith et al. (2006) did not specify the sources they used for information on wood species-mix and densities and methods to develop the primary product volume to cubic feet, cubic foot volume to weight conversions, and fraction of product that is wood fiber. These factors may need to be updated periodically to reflect changes in species mix over time. This highlights the importance and potential pitfalls of unit conversions applied to forest industry unit conventions that change over time. In USFS HWP-C v1, additional calculations are applied to the primary product volume to carbon conversion factors provided in Smith et al. (2006) to derive metric tons of carbon per hundred cubic feet (CCF) of primary product conversions rather than for the primary product volume units provided in Smith et al. (2006).

After volumes and carbon are distributed to primary product categories, U.S. national **end-use product ratios** are then applied to primary product categories. **End-use products**, also called "secondary products" (Pingoud et al. 2006), include everything from fuelwood (assumed oxidized in the first year of harvest), to lumber in houses or furniture, or wood pulp used to make paper (McKeever 2009, McKeever and Howard 2011). USFS HWP-C v1 used 224 end-use classes, but only a fraction of these account for a large portion of the HWP carbon. End-use product ratios originally provided in Smith et al. (2006) are limited to estimates from 1998 (derived from McKeever 2002). Skog (2008) provided end-use product ratios for 1900 through 1947 (from an undocumented USDA Forest Service estimate), 1948 through 1961 (USDA FS 1982), and 1962 through 1998 (McKeever 2002). USFS HWP-C v1 provides ratios using updated work from McKeever (2009) and McKeever and Howard (2011) from 1950 through 2009; values prior to 1950 were applied backward through 1906. It is unclear how often these ratios should be updated, but they have not been updated since 2009. It is also possible that these ratios vary at regional or local scales. The USDA Forest Service Forest Products Laboratory (FPL) is in the process of updating national end-use product ratios based on various newer datasets.

USFS HWP-C v1 assumes an 8-percent loss factor as primary HWPs are put into various end-uses, to account for material such as trim in manufacturing and wood waste in construction.[4] Users can provide a different loss factor if desired. This loss factor in the model sends the "lost" material through the various discard pathways described later in this section. Skog and Nicholson (2000) previously noted an 8-percent loss factor for solid wood and 5 percent for paper products, although it is unclear where they derived these estimates. Row and Phelps (1996) provided loss factors for poles and piling (2 percent), plywood (7 percent), lumber (8 percent), oriented strand board/waferboard (7 percent), particleboard (12 percent), and paper and board (5 percent); Row and Phelps (1996) stated these values were judgments from industry professionals and the disposition of similar materials in SWDSs. Further investigation into these values is warranted. USFS HWP-C v1 did not apply a separate loss factor for paper. HWP-C vR was adjusted to allow users to enter a separate loss factor for paper; other future variants may include this function as well.

In USFS HWP-C v1 and its variants, carbon in primary products that are not burned for energy or lost through immediate discard enter the products in use storage pool. End-use **product half-lives**, or the amount of time (typically expressed in years) it takes for half of the wood material (e.g., single-family houses) produced in a given year to be discarded (e.g., when the single-family houses are remodeled or demolished), are applied to the HWP in-use pool to determine the annual amount of carbon that is discarded. Once discarded, **discard disposition ratios** allocate the amount of carbon to either stored or emitted discard pathways. Discard **emissions** pathways include discarded HWPs that are burned with or without energy capture and compost. Discard storage pathways include HWPs that are recycled (products reenter the in-use pool) or disposed of into landfills or dumps. A landfill is a regulated, covered waste collection facility, whereas a dump is uncovered, excavated area where waste is deposited. Dumps are technically no longer legal in the United States, but were historically used for waste collection. The model takes into account historical waste deposition

[4] In the Microsoft Excel version of this model, the loss factor was applied to both solid wood and paper. This difference in USFS HWP-C v1 was likely unintended and was not identified until later variants were being developed.

at a time when dumps were still in operation which affects decomposition and emissions rates. USFS HWP-C v1 refers to the combined landfill and dump discard storage pools as the SWDS pool. Although HWPs disposed of in open air dumps are subject to high rates of decay, some portion of HWPs disposed of in the anaerobic environments of landfills decay little and are assumed to be permanently stored (Barlaz 1998; Wang et al. 2011, 2013). The ratios that establish the proportion of HWP carbon that is not subject to decay (i.e., nondegradable) compose the **landfill fixed carbon ratio**. **Discarded product decay half-lives** are applied to the degradable proportion of carbon in solid wood and paper products disposed of in SWDSs in perpetuity to determine the rate at which carbon is emitted to the atmosphere. The representation of product disposal rates and the application of decay half-lives in this model is considered a first-order (i.e., exponential) decay model and a simplification of a gamma distribution to determine carbon retention patterns (Bates et al. 2017). End-use product ratios, discard disposition ratios, landfill fixed carbon ratios, discarded product decay half-lives, and decay models are discussed in more detail in "HWP Half-Lives" and "HWP Disposal: Discard Ratios and Discard Half-Lives." Recycling is discussed in greater detail in "HWP Recycling."

USFS HWP-C v1 and variants use annual harvest inventory data (converted to carbon) and allocate it to various HWP carbon storage and emission pools. These values are added together to estimate the cumulative HWP carbon stock and emissions for each inventory year (fig. 2). It should be noted that emissions estimates generated by this model represent the carbon content of HWPs only and do not describe the type of GHG (e.g., CO_2, CH_4), particulate matter, or other carbon compounds. Although this model provides emissions estimates under the chosen tier 3 IPCC production approach, stock change to HWP carbon storage pools is the metric reported in an inventory. Typically, stock change is reported as a time series of annual change, which can be calculated as the difference between any two inventory years. HWP-C vR has modified outputs so that the flux-based IPCC simple decay approach can also be evaluated.

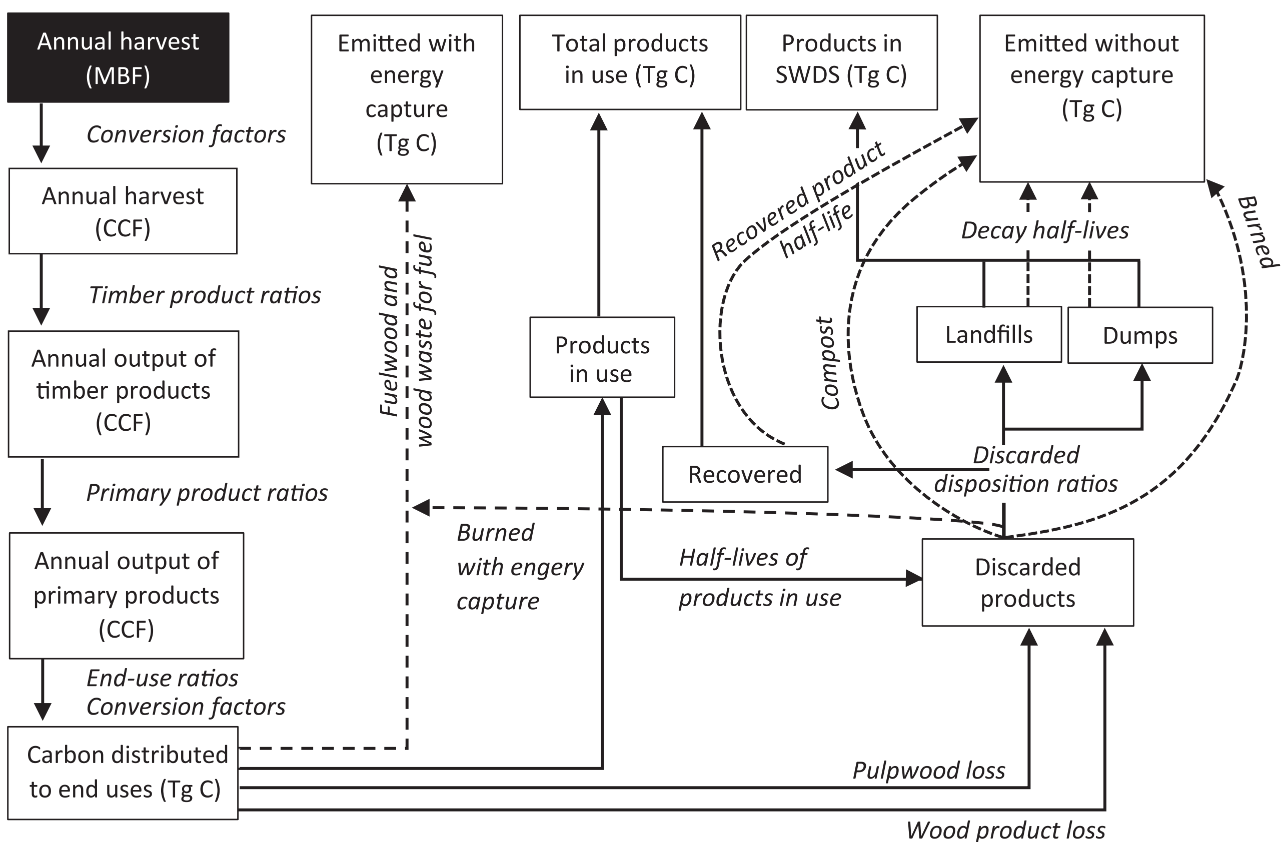

Figure 2—Example of how annual harvest is allocated to timber products, primary products, end-uses, and discarded dispositions in the USFS HWP-C v1 model and its variants (Groom and Tase 2022) (adapted from Stockmann et al. 2012). CCF = metric tons of carbon per hundred cubic feet; MBF = thousand board feet; SWDS = solid waste disposal site; Tg C = teragrams of carbon.

There are several assumptions and potential uncertainties in the values used for the input datasets used in USFS HWP-C v1. Although no complete sensitivity analysis has been conducted to determine which parameters have the greatest impact on the overall outcome of USFS HWP-C v1 model results (Stockmann et al. 2014), a Monte Carlo simulation of uncertainty has been incorporated into the model and its variants. This simulation includes a range of uncertainty for 18 of the input variables described above. These ranges were determined from publications and expert opinion, but users can adjust the ranges. Generally, greater ranges of uncertainty are given to variables further back in time—for example, there is a higher uncertainty associated with older harvest estimates. This analysis results in confidence intervals for estimates of HWP carbon stocks and emissions. Analyses with HWP-C vR for California and Oregon demonstrated that uncertainty increases with time; however, the uncertainty around current stock and emissions is about 20–25 percent of the estimated state-level values at the 90 percent confidence interval (Christensen et al. 2021, Morgan et al. 2021).

NFCMARS-HWP Model Description

ANSE version 1.0 is computer software that can be used to create an HWP carbon model (e.g., Dymond 2012, Xie et al. 2021, Smyth et al. 2013). This software can accommodate different IPCC approaches, estimation methods, or data inputs. Individualized modifications to models built using ANSE can be incorporated from direct consultation with the Government of Canada Natural Resources Canada Canadian Forest Service. Although it is a data- and time-intensive effort, a detailed and technical ANSE user's guide can be provided for users interested in modifying or creating new ANSE-based models. NFCMARS-HWP is the main model constructed from ANSE and is used for the Canadian national inventory report to the UNFCCC (ECCC 2021). Throughout the following review, there will be important distinctions between the general functionality within the ANSE software and specific models built from ANSE. Most of the following discussion around specific models pertains to NFCMARS-HWP unless other variants are specifically mentioned.

Although there are some key differences, NFCMARS-HWP follows a similar process as USFS HWP-C v1 (fig. 3). Models created with ANSE can follow any IPCC approach, but NFCMARS-HWP runs the IPCC flux-based simple decay approach, which uses the same system boundary as the stock-based production approach. Both approaches include exported wood in the system boundaries and do not include imported wood. However, the simple decay approach focuses on tracking emissions associated with harvest and HWPs, while the production approach focuses on tracking HWP carbon stocks remaining in the system. Carbon estimation in NFCMARS-HWP begins with harvested roundwood outputs from records of scaled logging truck loads for public land plus a factor for private land. These volumes are converted to units of carbon based on wood densities before modeling. Harvest volume in units of carbon is used to set harvest targets in the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) forest ecosystem model (Kurz et al. 2009). Outputs from CBM-CFS3 are then used as the inputs for NFCMARS-HWP.

Through an annual **milling efficiency ratio**, NFCMARS-HWP specifies the portion of industrial roundwood (used for commodities or primary products) that becomes a product, with the remainder being mill residues (UNFCCC 2003). Mill residue utilization in NFCMARS-HWP is estimated through "mill residue splitting," in which a portion is allocated to fuelwood or wood pulp inputs, or is sent straight to the landfill as unutilized residue. In USFS HWP-C v1, rather than determining a quantity of mill residue upfront that is then distributed to primary products, primary product ratios for fuelwood and wood pulp already include the quantity of mill residue derived from the processing of some timber products (e.g., sawlogs) that serve as inputs to these products. In NFCMARS-HWP, industrial roundwood undergoes further "commodity splitting" into primary products, such as sawlogs, panels, and wood pulp. Both milling efficiency and primary product ratios come from national-scale Canadian HWP production data (e.g., CCFM 2022, ECCC 2021) as well as trade statistics in the United Nations Food and Agriculture Statistics database (e.g., UN FAOSTAT 2022). Although primary product categories are predefined in NFCMARS-HWP, ANSE allows the creation of any number or type of primary products to be specified in models built using this framework.

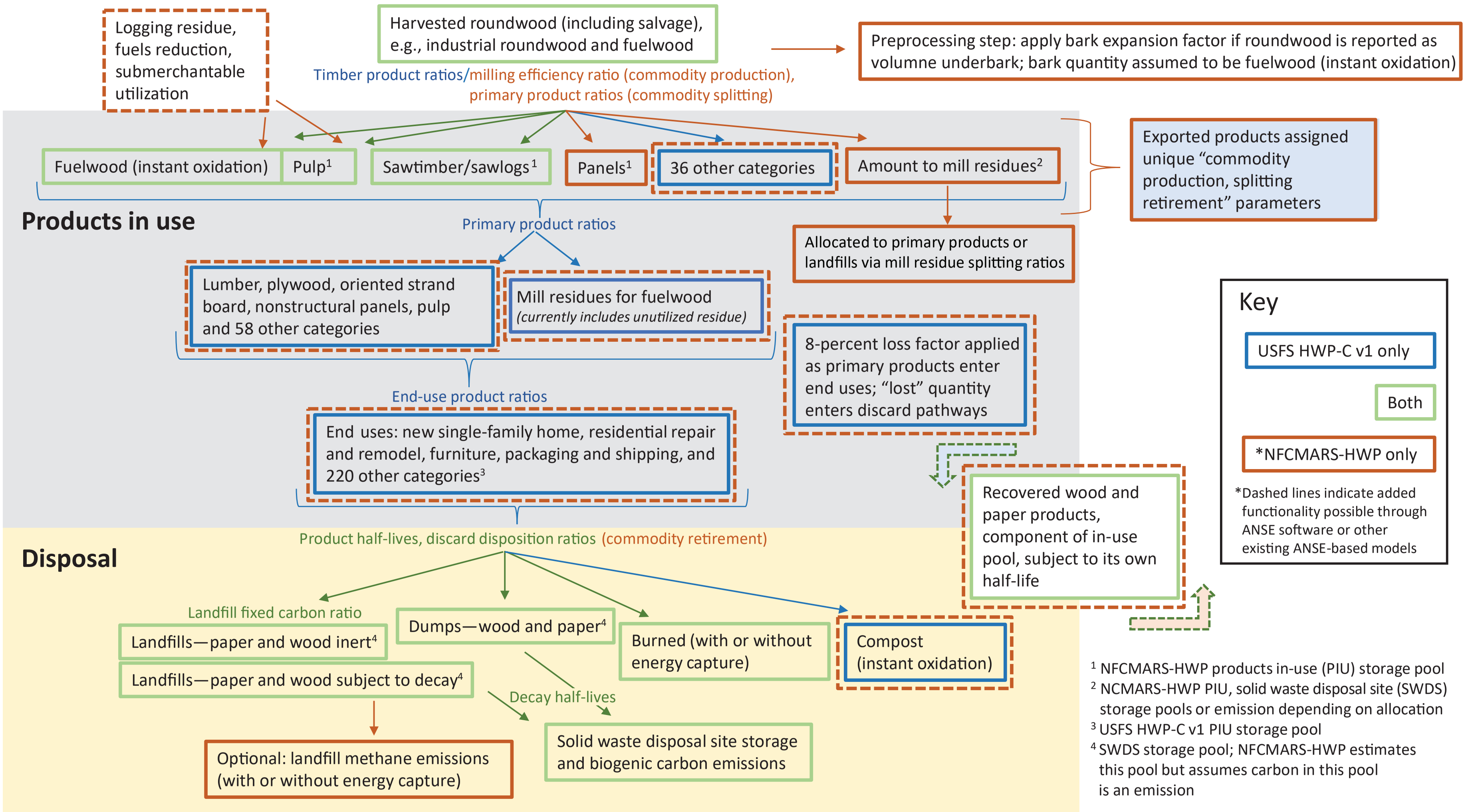

Figure 3—Comparison of USFS HWP-C v1 and NFCMARS-HWP models' carbon pool transfers. Note that other variants built from Abstract Network Simulation Engine contain functionality described for USFS HWP-C v1 only (not depicted).

The proportion of exported primary products is specified after commodity splitting in NFCMARS-HWP and other ANSE-based variants. Exported commodities are subject to utilization parameters and overall wood flow pathways specific to the usage in the exports destinations rather than assumed to be used in the same manner where the local wood is consumed. For example, in NFCMARS-HWP, wood that is exported to the United States, Japan, and all other export destinations combined is subject to unique wood utilization pathways and parameters. A newer model built from ANSE used in the Canadian forest sector mitigation analysis called MitigAna uses different wood utilization pathways and parameters for wood exported to China and the United States. (Xie et al. 2021). IPCC primary product half-lives and categories (paper, wood panels, and sawnwood) are applied to wood used in Canada, with different values from or derived from IPCC applied to wood used in defined export destinations (IPCC 2003, 2014; Pingoud et al. 2006).

Similar to USFS HWP-C v1, HWPs remain in use or are disposed of by applying half-lives of products in use after carbon is allocated to either domestic or exported primary products. The application of primary product half-lives is referred to in NFCMARS-HWP as "commodity retirement."

A key distinction between NFCMARS-HWP and USFS HWP-C v1 is that because half-lives are applied to primary products, this model considers primary products to be the "end-use product." USFS HWP-C v1 specifies end-uses and the half-lives of those end-uses, such as for lumber used in single-family housing construction. However, ANSE software provides a user-defined estimation framework that allows users to specify different primary product categories and associated half-lives. Users could choose to follow a pathway similar to USFS HWP-C v1 and allocate primary products to end uses with end-use product half-lives, like the British Columbia Harvested Wood Products version 1 (BC-HWPv1) (Dymond 2012) or MitigAna ANSE-based variants (Xie et al. 2021). In BC-HWPv1, primary products are allocated to end uses with loss factors from construction (Franklin Associates 1998) applied to lumber (7.5 percent), panels (4 percent), and plywood (4 percent). End-use product ratios in variants of ANSE-based models are regionally located and data dependent; for example, MitigAna uses data similar to that used in the United States, such as from McKeever and Howard (2011).

Once commodities are retired from the in-use pool, NFCMARS-HWP currently assumes instant oxidation (even though continued carbon storage occurs in SWDSs). In addition to CO_2 emissions, there are CH_4 emissions associated with landfill decay (reported in the IPCC waste sector), which have a 100-year global warming potential that is 25 times higher than CO_2 (IPCC 2007). Excluding carbon stored in SWDSs is therefore a conservative approach to estimating the carbon storage benefits associated with HWPs. However, the model is able to track carbon through the various discard pathways to provide emissions information for the waste sector portion of the national inventory report. When the landfill function is turned on in the model, carbon is transferred to discard pools, such as incineration with or without energy capture or SWDSs using discard ratios, similar to the process in USFS HWP-C v1. Currently no data exist to support utilizing the waste incineration with energy capture pathway. IPCC (2014) default discard half-lives are applied to HWPs in landfills to estimate emissions or storage. Similar to USFS HWP-C v1, a landfill half-life is applied only to the portion of HWPs subject to decay as determined by landfill fixed carbon ratios. These discard pathways and application of decay rates are all representations of the first-order decay method. However, ANSE allows for the use of any carbon retention pattern (e.g., other gamma distributions), which has been implemented in some ANSE-based models like MitigAna.

Although the process described here typically begins with roundwood, users can build an HWP carbon model from ANSE software that begins estimation at any stage, such as starting with a known quantity of primary products. Although NFCMARS-HWP and the other examples provided here have been used for Canadian HWP carbon estimation, other ANSE variants have been parameterized using national- and state-level data for a few U.S. states, including South Carolina (Dugan et al. 2018) and Vermont (Dugan et al. 2021). Some parameters from NFCMARS-HWP were retained in those variants, and U.S. and state-specific data were used whenever possible.

Firewood, Logging Residues, and Other Noncommercial Biomass

In the United States and Canada, most carbon removed from forest ecosystem pools for HWPs is accounted for in HWP carbon estimation. However, some carbon is removed from the ecosystem for uses that are not explicitly accounted for or have limited data available, such as bark or personal-use firewood. In other cases, material such as logging residues (a component of harvest slash) or noncommercial biomass from fuel reduction projects represent additional biomass that could be used in future forest management scenarios. Utilization of bark, firewood, logging residues, and other noncommercial biomass may result in relatively short-lived products (e.g., animal bedding, mulch, wood pulp), may be burned for energy production, or could represent feedstock for innovative products (Cabiyo et al. 2021). These uses may or may not be explicitly estimated depending on the chosen HWP carbon model and IPCC approach and tier. However, omission of these materials may have negligible effects on carbon stock estimates because current uses generally result in instantaneous emission in the year of harvest (e.g., fuelwood), or the materials have such short-lived uses that they can still be considered as an emission in the year of harvest. However, quantifying the carbon associated with the use of these materials is helpful for conducting an LCA and for understanding the potential climate implications from current or additional utilization. For example, feedstocks used for energy production may be substituting for more fossil-fuel-intensive energy production (e.g., coal, natural gas, heating oil) and result in fewer total emissions. The following sections discuss how these materials are, or can be, addressed in HWP carbon estimation.

Bark

Bark is a byproduct or coproduct of commercial harvest and is generally burned for energy production (e.g., as hog fuel) or used for short-lived products, such as landscaping materials. According to the IPCC (Pingoud et al. 2006: 12.5), HWPs include "all wood material (including bark) that leaves harvest sites." In HWP carbon estimation, it is important to know if harvest data is reported in metrics that include bark (e.g., roundwood) or exclude bark (e.g., commodity production statistics that are reported in cubic- or board-foot units underbark). In U.S. tier 3 IPCC production approach HWP carbon estimates, explicit estimates of carbon associated with bark are excluded (e.g., USFS HWP-C v1 and WOODCARB II). Bark is excluded because the starting point for HWP carbon estimation is commercial harvest volume once bark has been removed (i.e., **underbark volume**). Bark that is removed through commercial harvest is still accounted for because it is not present in forest ecosystem pools or HWP carbon storage pools and is therefore an emission that is not explicitly identified and quantified. In NFCMARS-HWP, the starting point for HWP carbon estimation is roundwood removals generated by the CBM-CFS3 forest ecosystem model. CBM-CFS3 applies a **bark expansion factor** to volumes that are reported as underbark in national harvest statistics, ultimately resulting in bark being included in the HWP carbon estimation. However, if users have a different source of harvest data to drive an ANSE-based HWP carbon model, they would need to know if bark is included and adjust their input data accordingly. Bark is assumed to be combusted in the first year of harvest for energy production as hog fuel, but these emissions are not explicitly separate from other fuelwood products (ECCC 2021, Head et al. 2021).

If users are interested in explicitly estimating carbon associated with bark, a bark expansion factor can be applied to harvest volumes that are reported as underbark or by applying a bark loss factor to roundwood volumes reported as overbark (Pingoud et al. 2006). This step helps explicitly estimate emissions associated with energy production generated from bark (Bruckner et al. 2014). The IPCC (2014) recognizes that there are other sources of feedstock for commodities besides industrial roundwood or wood pulp and encourages countries to use country-specific data. In California, for example, bark has been estimated to contribute an additional 16 percent of harvest volume removed during a commercial harvest volume (Marcille et al. 2020). Of the reported underbark wood volume, about 60 percent is used for bioenergy and comprises an estimated 22 percent of the total wood bioenergy production in the state (Marcille et al. 2020). Although short-lived, another 40 percent of the bark becomes landscaping materials, which is a temporary storage pool with a slightly delayed emission (Marcille et al. 2020). In this way, it may be important for states such as California to consider additional measures to incorporate bark utilization and still adhere to IPCC guidelines. Methods to include bark utilization are being explored for HWP-C vR. Emissions from using bark for energy to support quantification approaches will be incorporated into a new Excel workbook tool being prepared in conjunction with the USDA's update to *Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory* (Murray et al., n.d.; Stockmann 2022).

Fuelwood

The United States and Canada estimate HWP carbon associated with fuelwood because it is a large category comprising many different products. However, there are some limitations and caveats to estimating fuelwood carbon. In FIA Timber Products Output data, commercial fuelwood is estimated as part of the fuelwood timber and primary product categories. Commercial fuelwood includes wood burned at biomass energy facilities, hog fuel used for boilers at wood products facilities (e.g., pulpmills or sawmills), and commercial firewood and other fuel products (e.g., personal-use fuelwood, firewood). Firewood collected under fuelwood permits from NFS lands is generally included in harvest volumes reported from USDA Forest Service cut-and-sold reports (USDA FS 2022). Consequently, in U.S. HWP carbon estimates (i.e., USFS HWP-C v1, WOODCARB II, HWP-C vR), this specific use of firewood is included, although not distinguishable from commercial fuelwood. However, illegal firewood removal from NFS lands and unreported firewood removal from other public or private lands may result in an underestimation of firewood removals. The U.S. Energy Information Administration (US EIA) publishes data on residential firewood use at the national and state levels (US EIA 2021a, 2021b). These data are currently published only in British thermal units rather than in physical units (i.e., thousands of cords) (US EIA 2021b). This makes it difficult to convert British thermal units into units of firewood consumed and potential energy substitution that may be occurring. The USDA Forest Service FIA program is currently working to address this data gap, and this remains an area for future research. Additionally, although harvest volumes may include some degree of residential firewood, it is unclear how this information is factored into the development of timber and primary product distribution ratios from mill survey data, as this material would not generally enter a mill. NFCMARS-HWP includes an option for residential firewood and incorporates these data and parameters from Environment and Climate Change Canada (ECCC 2021, Head et al. 2021).

Harvest Slash or Logging Residues

The components of harvest slash (dead trees, branches, tree tops, nonmerchantable tree species, and small-diameter trees) are byproducts of commercial harvest and present unique challenges in HWP carbon estimation. In the United States, a portion of harvest slash is typically left in the forest ecosystem to decay or it is burned. This unutilized harvest slash is generally referred to as logging residues. However, a portion may also be used for energy, landscaping materials, or other innovative products, such as transportation fuels or biochar (Benson and Johnston 1976, Kizha and Han 2016, Sahoo et al. 2019). Like bark and firewood, most of these uses represent an emission rather than long-term storage, with the exception of biochar. In both U.S. and Canadian HWP carbon estimation, harvest slash utilization is included to the extent it is reflected in commercial harvest volumes (e.g., Morgan et al. 2012).

In terms of understanding the climate mitigation potential of the forest sector, logging residues that are left in the forest represent additional material that could be utilized with the potential to reduce overall emissions when substitution benefits are considered in comparison to open pile burning in the forest (Jones et al. 2010). This type of scenario analysis requires the ability to estimate how much of this material is available from the forest ecosystem and a pathway for the material to enter into the HWP carbon model. In Canada, users can choose to estimate additional utilization of logging residues removed from the forest in the CBM-CFS3 forest ecosystem model by applying a parameter value for the portion or volume of residue recovered (Smyth et al. 2013). In the analysis by Smyth et al. (2013), this was a range of values (2–12 percent additional recovery) depending on the province or territory in Canada and can be an average across a range of years (Smyth et al. 2013). This additional material can then enter ANSE-based HWP carbon models into existing pools (e.g., fuelwood) or into its own distinct pool. This material can then join preexisting pathways (e.g., the fuelwood pathway) or new life-cycle pathways determined by the user. In the United States, logging residues potentially available for utilization could be determined through logging utilization studies (e.g., Morgan and Spoelma 2008), other field data, or forest ecosystem models. Once these quantities are established, scenarios of additional logging residue utilization evaluated through models, such as USFS HWP-C v1, would likely need to be reflected as increased harvest volume inputs and changes to the timber and primary product ratios or through the development of a new pathway for logging residues in the model.

Similar to logging residues, other noncommercial material from precommercial mechanical thinning or fuel reduction activities are also often left to decay in the forest or are burned onsite. This material represents another portion of cut trees that could be available for removal from forest ecosystem pools and used for bioenergy or other innovative products. Because it is noncommercial material and therefore not removed for HWP production, it is excluded from U.S. and Canadian HWP carbon estimation (e.g., USFS HWP-C v1 and NFCMARS-HWP). Using this material could be explored in either model by adding these quantities to harvest volumes or roundwood estimates and adjusting timber and primary product ratios according to their possible use, or through the development of a new pathway for this material in the model.

HWP Half-Lives

Assumptions about HWP half-lives can have a profound effect on HWP carbon storage and emissions estimates over time (Dymond 2012). Product half-lives vary depending on the wood types, functional purpose, maintenance, and other environmental factors (Hoover et al. 2014, Skog 2008, Smith et al. 2006). Other variables, such as socioeconomic factors and demand for housing, additionally influence half-lives because they affect consumption and turnover rates, but these variables are seldom incorporated in HWP carbon models. End-use product half-lives are important for HWP carbon estimation, especially if more harvest volume is going into longer lived uses, such as lumber in single-family homes, rather than shorter lived uses, such as pulpwood in paper and cardboard. International or national defaults for half-lives are often applied to HWPs in carbon modeling, although it is unclear whether these values vary nationally or subnationally. As technology improves, products may remain in use for longer periods, thereby increasing the half-lives and delaying carbon emissions associated with HWP decay. For example, Skog (2008) recalculated half-lives for wood used in U.S. single- and multifamily homes (including repairs and remodels) using USDA Forest Service and U.S. Department of Commerce Bureau of Census data on housing. Skog's (2008) estimated half-lives varied among time periods of construction, with shorter values for homes built earlier in the century compared to later in the century. In USFS HWP-C v1, end-use product half-lives cannot be modified. However, in all subsequent USDA Forest Service and state variants of the model, users have the ability to update or change half-lives.

NFCMARS-HWP uses IPCC (2014) half-lives for the three primary product categories (table 3) for the national inventory report. Primary product half-lives in NFCMARS-HWP incorporate the range of half-lives to which each product may be subject. For example, sawnwood half-lives incorporate the long-, medium-, and short-lived uses of the product, such as lumber in homes versus lumber in other manufactured products. However, other variants built from ANSE apply half-lives to end uses (e.g., Dymond 2012, Xie 2015). Dymond (2012) contributed several new datasets to the estimates of end-use product half-lives in North America that supported values much longer than the default IPCC values. Using the Vermont variant built from ANSE, Dugan et al. (2021) compared these updated primary product half-lives with IPCC (Pingoud et al. 2006) and Skog (2008) to understand the impact on HWP carbon.

Table 3—International Panel on Climate Change (IPCC) default primary product half-lives used in the NFCMARS-HWP model compared with half-lives used in three modeling scenarios using the Abstract Network Simulation Engine (ANSE) Vermont variant

Harvested wood products	IPCC default half-lives[a]	Three ANSE variant half-lives[b]
	YEARS	
Sawnwood	35	30, 53, 90 (sawlogs)
Wood panels	25	27, 38, 50 (veneer logs)
Paper	2	2, 2.5, 3 (pulp and paper)

[a] IPCC (2014).
[b] Dugan et al. (2021).

USFS HWP-C v1 and subsequent variants apply half-lives to primary products in end-uses rather than directly to primary products as in NFCMARS-HWP. Original documentation for USFS HWP-C v1 (e.g., Anderson et al. 2013, Stockmann et al. 2012) and related models did not specify a data source for end-use product half-lives (Stockmann 2021); however, these country-specific tier 3 values came from Skog and Nicholson (2000). These values are also summarized in Smith et al. (2006), with additional data sources listed. However, some half-lives in USFS HWP-C v1 that were used for one product category were assumed to be sufficient for others that lacked data. For example, "manufactured products" are assumed to have a 12-year half-life in Skog and Nicholson (2000). This same half-life was applied to the "other industrial uses" end-use category in USFS HWP-C v1 because there were no data for this particular end-use category at the time. Lastly, USFS HWP-C v1 uses a half-life for paper of 2.6 years. This is the only unique value in Smith et al. (2006) and it differs from the half-life for paper in Dymond (2012), Skog (2008), Skog and Nicholson (2000), and in IPCC publications (IPCC 2003, 2014; Pingoud et al. 2006). Table A3-1 in appendix 3 of this synthesis provides data sources and assumptions made where data were missing for end-use product half-lives in USFS HWP-C v1 and subsequent variants as well as differences in values with more recent publications.

Newer data on end-use product half-lives, such as from Dymond (2012) or Skog (2008), could be reviewed to determine whether these more recent estimates should be incorporated into HWP models. Skog (2008) estimated that the half-life for solid wood in single-family homes built since 2001 is 83.91 years; an earlier publication, Skog and Nicholson (2000), suggested 100 years, which is currently the estimate used for that category in USFS HWP-C v1. Meanwhile, using BC-HWPv1, Dymond (2012) applied a 90-year half-life to single-family homes, but estimated a range of 78 to 350 years for this category. Updated values could either be directly incorporated in model parameters or used to update required parameters, such as in the case of the Vermont variant from ANSE for which updated end-use product half-lives from Dymond (2012) were used to estimate a range of updated primary product half-lives. Updated information could also be incorporated through modifications to how the models currently function. For example, Skog (2008) suggested that half-lives varied through time and provided new half-lives for homes in 20-year increments. Models could be updated to allow application of different half-lives for different time periods. Sensitivity analyses could be used to determine whether updating half-lives for certain uses or certain time periods make more or less of a difference in overall results. Uncertainty parameters, such as in the USFS HWP-C v1 Monte Carlo analysis, could also be modified to reflect the potential ranges in these values, in general or in different time periods, thereby providing a range of uncertainty in the resulting estimates.

HWP Disposal: Discard Ratios and Discard Half-Lives

Once HWPs reach the end of their functional life, they may be transferred to a SWDS, recycled, composted, or burned with or without energy capture using discard disposition ratios. For USFS HWP-C v1, these ratios are provided in Skog (2008), which are based on an unpublished spreadsheet (Freed and Mintz 2003) using data from US EPA (2005) and other data from Melosi (1982, 2000). These ratios change over time; for example, prior to 1970, most wood and paper products were disposed of into dumps, whereas in recent decades, HWPs are transferred to landfills with lower decay rates. Although it was not possible to alter discard disposition ratios in USFS HWP-C v1, subsequent variants allow advanced users to change inputs as desired. Updated national data (US EPA 2020b) are available to update discard disposition ratios for use with U.S. HWP carbon estimation models. It is also possible that local data could be used to develop new ratios for subnational estimates, such as waste characterization surveys from state agencies (e.g., CalRecycle 2020). However, these datasets include all waste, including waste from imported products, making it difficult to separate the proportion of waste associated with domestic HWPs (or HWPs from the area of interest). Another caveat is that subnational waste surveys are a relatively recent effort, so waste data may only exist for the past two decades and are often not surveyed on an annual basis. Data sources estimating the portion of discarded products burned for energy capture (e.g., municipal waste incineration with energy generation) were not available and therefore were not provided in USFS HWP-C v1 and subsequent variants. Subnational data on waste incineration for energy or fuel may also be available from state agencies (e.g., CEC 2022); however, it may be difficult to separate the proportion of waste incineration associated with just HWPs from the general waste stream in these reports.

When HWPs are disposed of in a SWDS, a decay half-life is applied to the biodegradable portion of the discarded products. The decay half-lives of disposed HWPs are as equally important as product half-lives because they influence continued carbon storage and emissions estimates from SWDSs. USFS HWP-C v1 currently uses discarded product decay half-lives for wood and paper in landfills and dumps listed in Skog (2008), which originate from IPCC (2000) and Pipatti et al. (2006) defaults (table 4). HWP disposal rates and the application of decay half-lives in USFS HWP-C v1 follow a first-order decay (i.e., exponential) method and is a simplification of a gamma distribution to determine carbon retention patterns (Bates et al. 2017). This assumes the maximum rate of decay in the first year. Other decay models that determine the carbon retention pattern, such as the chi-square application of the gamma distribution, may more accurately represent when HWPs are retired from use and decay (Marland and Marland 2003, Marland et al. 2010). The representation of other decay functions is being explored in future iterations of USFS HWP-C v1 (Stockmann 2022).

Table 4—Discarded product half-lives for the biodegradable portion of wood and paper in landfills and dumps used in the USFS HWP-C v1 model

Discard pool	Discarded product half-life	Data source
	YEARS	
Landfills—wood	29	Pipatti et al. (2006)
Landfills—paper	14.5	Pipatti et al. (2006)
Dumps—wood	16.5	IPCC (2000)
Dumps—paper	8.25	IPCC (2000)

The forest sector in the Canadian national inventory report excludes HWP carbon stored in SWDSs; however, NFCMARS-HWP provides an option to estimate the carbon storage and emissions associated with SWDSs. This information is provided for SWDS emissions reporting in the waste sector portion of the Canadian national inventory report. Discarded product pathways in NFCMARS-HWP function similarly to USFS HWP-C v1, although there is no default option for composting products. However, NFCMARS-HWP appears to have a more complex way of addressing disposal into SWDSs—discarded wood can be retired as either (1) landfilled wood pulp/paper in industrial waste sites (i.e., associated with wood processing facilities) or (2) landfilled wood or paper in municipal solid waste sites (ECCC 2021). Data sources to determine the discard fate of HWPs at the end of their useful life are documented in the waste section of the national inventory report (ECCC 2021). Discarded product decay half-lives are applied to HWPs in SWDSs depending on the province. This is because of differing environmental conditions in these locations and SWDS type, affecting the HWP decay rates; for example, decomposition occurs more slowly in dry climates than in humid climates (Dugan et al. 2021, Row and Phelps 1996). Specific discard half-lives are also provided for specific export countries such as the U.S. or Japan. In the Vermont variant built from ANSE, dry, mesic, and wet landfill types were specified, each with different half-lives (Dugan et al. 2021)

Landfill Fixed Carbon Ratios

Landfill decay half-lives for USFS HWP-C v1 and its variants as well as ANSE variants are only applied to the biodegradable portion of wood and paper waste due to the anaerobic decay process in covered landfills (Barlaz 1998; Wang et al. 2011, 2013). Using a combination of estimates (Barlaz 1998, Freed and Mintz 2003), Smith et al. (2006) and Skog (2008) suggested that the nondegradable portion, or fixed carbon ratio in landfills, is 77 percent for solid wood and 44 percent for paper. These estimates are used in USFS HWP-C v1 and its variants as well as BC-HWPv1 (Dymond 2012). More recent publications have noted that these earlier estimates are uncertain, may vary among and even within landfills, and may not be appropriate for every solid wood product (Wang et al. 2011, 2013; Ximenes et al. 2018). The U.S. EPA Waste Reduction Model (hereafter US EPA WARM v-15) (US EPA 2020b) published updated nondegradable portions of various wood and paper products (table 5). US EPA WARM v-15 is designed to help solid waste managers understand the GHG emissions profile and economic impacts of their facilities.

Table 5—Estimated landfill fixed carbon ratios of paper and solid wood products in the U.S. Environmental Protection Agency's Waste Reduction Model (US EPA 2020b)

Product type	Estimated fixed carbon ratio in landfills
Newspaper	0.84
Corrugated containers	0.55
Office paper	0.12
Coated paper	0.74
Dimensional lumber	0.88
Wood flooring	0.95
Medium-density fiberboard	0.84

Compared to values used in the HWP models, US EPA WARM v-15 data in table 5 indicate higher fixed carbon ratios for HWPs in landfills than are currently being used in USFS HWP-C v1 and other model variants. These ratios are also product specific within waste categories. Rüter et al. (2019) suggests a landfill fixed carbon ratio of 90 percent for wood materials, based on averages in the literature for engineered wood products, sawnwood, and tree branches (Wang and Barlaz 2016, Wang et al. 2011, Ximenes et al. 2018). Rüter et al. (2019) suggests a landfill fixed carbon ratio of 50 percent for paper (and textiles and diapers), based on averages derived from a variety of sources (Eleazer at al. 1997, Jeong 2016, Wang et al. 2015). NFCMARS-HWP currently uses the updated default values provided in Rüter et al. (2019). If these estimates are more accurate than previous estimates applied in Smith et al. (2006) and Skog (2008), then models parameterized with those estimates are potentially overestimating landfill emissions and underestimating carbon storage in landfills.

Initially, it was not possible to change landfill fixed carbon ratios in USFS HWP-C v1, but all subsequent variants of the model, including HWP-C vR, allow users to change these inputs to the new averages for wood and paper from Rüter et al. (2019). However, further exploration is needed to determine if the US EPA WARM v-15 disaggregated wood and paper categories listed in table 5 can be used in these later variants of USFS HWP-C v1 or in NFCMARS-HWP. Currently, discarded HWPs in those models are lumped into the generalized “paper” and “wood” categories. Use of these values would require either HWP production estimates with more specific product categories (including paper) or product-specific discard ratios in waste streams, which have data limitations (e.g., CalRecycle 2018. We performed a partial sensitivity analysis of HWP-C vR using landfill fixed carbon ratios for these disaggregated paper and wood categories to determine the potential magnitude of impact on HWP carbon estimates (app. 4).

Landfill CH_4 Emissions

A portion of HWP carbon subject to decay in landfills also generates CH_4 emissions, a GHG about 25 times more potent than CO_2. Default CH_4 emission equations from Pipatti et al. (2006) can be applied to the portion of landfill wood waste subject to decay. Although IPCC guidance recommends that non-CO_2 emissions be reported in the waste sector, it is important to understand the impact that these emissions have on the climate and potential mitigation opportunities associated with HWP disposal. NFCMARS-HWP and other models built from ANSE have an option to estimate landfill CH_4 emissions. Models built from ANSE also have the capability of identifying different technologies to mitigate landfill CH_4 emissions, such as CH_4 capture and flaring, which can represent reduced CH_4 emissions or emissions associated with energy capture. Such technologies may result in potential substitution benefits as well, affecting the overall impact of these CH_4 emissions. However, if substitution effects exist, they would be captured in inventories for other sectors, rather than in the forest sector, and in a product LCA. Because of the forest sector emphasis of USFS HWP-C v1 and subsequent variants, these models are not aimed at quantifying other carbon-based GHG emissions, carbonaceous air pollutants, or particulates and therefore do not account for landfill CH_4 emissions from HWPs. However, HWP-C vR has been modified to isolate the portion of landfill stock subject to decay, allowing the potential for these emissions estimates to be included in the future. Rüter et al. (2019) and the US EPA WARM v-15 (US EPA 2020b) contain parameters that could be used to estimate landfill CH_4 emissions. These estimates could furthermore be made for subnational scales with the addition of moisture content in landfills using average rainfall data. Decay parameters for product-specific wood and paper categories have also been calculated (US EPA 2020b), which could affect resulting CH_4 emissions estimates (Barlaz 2022).

HWP Recycling

Recycling is another pathway for discarded HWPs (Pipatti et al. 2006, Rüter et al. 2019). Although IPCC contains guidance for recovered paper products, there is currently no guidance for recovered solid wood. However, HWP carbon models can vary in their treatment of recycled products, and some do not explicitly include recycling because there are limited data about products diverted for reuse rather than being landfilled, especially at the subnational scale. USFS HWP-C v1 accounts for recycled products using a recycled product discard disposition ratio using estimates from US EPA (2007). These ratios specify that 50 percent of paper-based products (beginning in 1960) and 9 percent of wood-based products (beginning in 1990) are transferred to a "recovered products" category and are tracked separately in two different classes—wood and paper—without any end-use specified. The recovered products category then becomes a component of the in-use pool. A single recycled product half-life is then applied to the paper and wood products in the recovered products category, which appears to correspond with the half-life of 2.6 years for paper from Smith et al. (2006). Retired recycled products are then allocated directly to the emissions without energy capture category rather than cycled through the various discard pathways based on the discard ratios.

Many paper and solid wood products are reused at least once in the same or similar end-use, which would support the use of the half-life for the recycled material's previous end-use (Row and Phelps 1996). Additionally, recovery options could include alternative or multiple stages of reuse or reprocessing, such as reprocessing lumber products as particleboard, pulping to form paper products, or burning for energy recovery (Bergman et al. 2013, Gustavsson and Sathre 2006). Traditional recycling material into the same use and reusing products as raw material inputs instead of virgin material are different forms of **cascaded utilization,** although the latter will be referred to as cascading throughout this synthesis.

Regardless of recycling assumptions, there is a limit to how many times a forest product can be recycled; in other words, no single product can be recycled in perpetuity as it degrades or decays over time (Höglmeier et al. 2015, Smith et al. 2006). To account for this, Smith et al. (2006) created decay tables with an efficiency rate of reuse (i.e., percentage of fibers that are recovered and incorporated into "new" products), which assumes that none of the fiber from paper is still in use after five rounds of recycling. However, in USFS HWP-C v1, products are recycled once and then subject to the paper half-life without a pathway to reenter the recovered products pool after the first entry. It is also unclear if a single half-life of 2.6 years is appropriate for recycled solid wood products. Adjusting the product half-life could be a simple way to address recycling; however, half-lives for recycled products are generally uncertain (Row and Phelps 1996). Row and Phelps (1996) provide a rate of recycling for 12 common end-uses and an adjusted end-use product half-life to reflect an extension of functional life of a product enabled by recycling. USFS HWP-C v1 and subsequent variants represent recycling in a simplified manner. USFS HWP-C v2 and a new USDA FS substitution calculator will have improved recycling parameters (Stockmann 2022); however, more research and data are needed to determine if there is a more complex way that recycling should be represented, such as cascaded utilization pathways.

Models built from ANSE can estimate recycling at both technical and conceptual levels because recycling is a user-defined input; however, Canada also lacks consistent data about recycled products (ECCC 2021). Consequently, NFCMARS-HWP does not include recycling. The flexibility of ANSE allows users to create their own recycled or cascaded utilization pathways. ANSE-based BC-HWPv1 simulates paper recycling and solid wood product recycling by allocating products to a generic pool with a 38-year half-life (Dymond 2012). MitigAna, another ANSE-based model, has a cascading uses module, which allows for the repurposing or reprocessing of wood into different commodities or end-uses as well as the ability to place limits on how many times the material can be recycled (Xie 2020).

Extending the life of wood products through traditional recycling and cascading could be beneficial because it can delay HWP carbon emissions from SWDSs, including associated landfill CH_4 emissions. In addition to the current challenges to representing recycling in HWP carbon models, additional considerations are merited for understanding tradeoffs related to recycling. For example, GHG emissions associated with recovering products for secondary or tertiary purposes could be considered in an LCA, including the fossil fuels needed to transport and manufacture recovered products. Effects on land use based on reduced demand for virgin wood material could also be considered and are discussed in more detail in "Applicability of Variants of ANSE and USFS HWP-C v1 to Evaluate Climate Mitigation Potential of Harvested Wood Products."

Imported and Exported HWPs

USFS HWP-C v1 and NFCMARS-HWP follow different IPCC approaches; USFS HWP-C v1 uses the pool-based, production approach, while NFCMARS-HWP uses the flux-based, simple decay approach (ECCC 2021). As mentioned previously, both approaches include exports and exclude imports, applying the same system boundaries. USFS HWP-C v1 was originally designed to track cumulative carbon storage in HWPs and focuses on carbon stock changes of HWP storage pools; however, model variants also include HWP carbon emissions with and without energy capture. HWP-C vR has been modified to use emissions outputs to accommodate the simple decay approach. Similarly, outputs from NFCMARS-HWP could be used to accommodate the production approach. Other models built from ANSE can be constructed to accommodate any IPCC approach or estimation method.

Assumptions made about the end use and disposal of exported HWPs (exported and consumed beyond the origin of harvest) contain many uncertainties and caveats. Because USFS HWP-C v1 was originally designed for national forest reporting, and most wood cut from national forests is processed in the United States, the model applies the same end-use distribution, product half-lives, discard assumptions, and discard half-lives for harvest used outside the geographic region as harvest used within the region. In other words, all harvested wood is treated the same. Given this assumption, if exported wood from non-NFS ownerships is used substantially differently, the product allocations and half-lives could be inaccurate. For example, low-grade logs might be used as pallet stock for pallets and shipping crates in one location, but if the same logs are exported, they might be used for energy in the form of wood pellets or fuel chips. This could result in an overestimate of the carbon storage associated with exported logs. The magnitude of potential errors due to this assumption depends in part on the proportion of harvested wood that is exported and the differences in primary products and downstream end-uses between harvest origin and timber or primary product export locations. Data about the fate of exported timber products is limited, especially for logs exported abroad. Exports can flow to dozens of countries and involve supply chains that include additional exports. It is also possible that harvested wood processed outside of its location of origin is returned to its place of origin as secondary and end-use products.

A recent study indicated that 3–4 percent of British Columbia, California, Oregon, and Washington harvests are exported to China in the form of logs (Christensen et al., n.d.; Dillon and Morgan, in press). Functional lifespans of HWPs in China are about 80 percent lower than those in the U.S. and British Columbia (Xie et al. 2021). China uses most wood for short-lived uses, such as concrete casings and packaging (Manley and Evison 2017). NFCMARS-HWP allows for different product uses and half-lives and discard pathways and half-lives for exported wood compared to wood consumed domestically. Country-specific data are used only for the major Canadian export markets, although a pathway for China has not yet been established. ANSE-based models allow the construction of any export pathway—as such, MitigAna allows for a pathway to China. The U.S. states could perform sensitivity analyses of alternate use and discard pathways and assumptions applied to the given export quantities of timber products to determine if these changes affect the estimates in a significant way before deciding to invest in identifying use and disposal datasets and developing pathways for exported wood in HWP carbon estimation models.

Focusing on harvest from domestic origins is useful for making policy decisions about forest management and wood utilization resulting from processing raw materials; however, imported HWPs can contribute to climate impacts. For example, most of California's HWP demand is met by domestically and internationally imported products (Christensen et al., n.d.). A recent analysis indicated that California is importing almost twice as much as what it harvests in primary products, with most coming from other U.S. states (Christensen et al., n.d.). Under the IPCC production approach, wood waste in landfills and dumps, and therefore end-of-life emissions from HWPs, would be accounted for in the jurisdiction of harvest. However, California may be interested in determining the climate impact from consumption, rather than just production, of HWPs in the state. Consequently, the California Air Resources Board, the entity responsible for the statewide GHG inventory for all sectors, has used the IPCC flux-based atmospheric approach to better reflect the carbon emissions associated with imported HWPs (CARB 2018b). If California imports wood from jurisdictions that follow the IPCC production approach, this could lead to double counting of emissions because those emissions would have already been estimated in those jurisdictions. Regardless of the IPCC approach used in the forest sector, landfill CH_4 emissions for all wood consumption within a jurisdiction are accounted for in the waste sector. This is ultimately a conservative approach for California to take when setting GHG emission reduction targets.

Applicability of Variants of ANSE and USFS HWP-C v1 to Evaluate Climate Mitigation Potential of HWPs

The HWP carbon estimation models described are limited to carbon in the HWP portion of the forest sector. However, ANSE-based HWP carbon models can be easily paired with the CBM-CFS3 forest ecosystem model. This creates an opportunity for forest management and wood utilization scenarios to be incorporated in tradeoff analyses. For example, available carbon stocks for HWP production can be analyzed in CBM-CFS3 based on increased or decreased harvest rates or removal efficiencies while simultaneously evaluating the changes in forest ecosystem pools. The ANSE-based models can be used to create future wood utilization scenarios by tweaking model parameters. For example, the amount of harvest going to short-, medium-, or long-lived HWPs through the use of timber product, primary product, and end-use product ratios. The ANSE-based models can also be parameterized to add new or emerging product categories (e.g., mass timber products), new half-lives if longer lived products emerge, and changes in discard pathways or discard ratios (e.g., enabling cascading pathways to extend the life of HWPs or increased waste incineration with energy capture). One ANSE software variant, MitigAna, has already improved recycling pathways and added cascaded uses and end-use categories for innovative materials to explore tradeoffs with the forest ecosystem (Xie 2020, Xie et al. 2021).

USFS HWP-C v1 and subsequent variants were not designed for tradeoff analyses and have not been used for such analyses to date. However, with future harvest volume estimates and anticipated utilization and decay parameters, variants of USFS HWP-C v1 that allow for user-defined inputs (e.g., HWP-C vR) have the potential to evaluate forest management and utilization scenarios, similar to ANSE-based models. This would require forecasts of harvest, primary product production, end-use, and disposal for future harvest years. Estimated harvest volumes resulting from different forest management scenarios still must come from the use of other models or planning efforts. Inclusion of innovative or emerging products, such as greater use of mass timber products, could be accomplished by a combination of increased amounts of lumber production and a longer associated half-life. HWP-C vR can also accommodate additional product categories if new products or end-uses emerge in the future. Exploring different recycling or cascaded use scenarios would be limited as recycling is simplified and cascaded utilization pathways do not currently exist in USFS HWP-C v1 and its variants. Improvements to recycling are anticipated with the release of USFS HWP-C v2.

Outputs from HWP carbon models can be used for subsequent substitution analyses. Economy-wide **substitution factors** (SFs) following methods similar to Smyth et al. (2017) or Nepal et al. (2016) can be developed for state or national levels and applied to overall product mixes under different wood utilization scenarios to calculate potential substitution benefits from HWPs tracked in HWP carbon estimation models. Several studies in Canada and the United States have already used ANSE-based HWP carbon models paired with CBM-CFS3 to evaluate the climate mitigation potential of HWPs resulting from different forest management and wood utilization scenarios that include substitution benefits (Dugan et al. 2021; Smyth et al. 2017, 2020; Xu et al. 2018). Similarly, WOODCARB II and the United States Forest Products Model (USFPM) were paired to evaluate U.S. national and regional carbon benefits of greater use of wood in low-rise nonresidential buildings and greater use of wood for energy (Nepal et al. 2016). USFS HWP-C v1 and subsequent variants have yet to be paired with substitution analyses, but it is possible to use model outputs to do so. Neither USFS HWP-C v1 and its variants nor ANSE model variants are capable of incorporating market force influences on forest ecosystem and HWP carbon pools yet. However, research is underway to incorporate **cradle-to-gate** LCA and SFs into a new Excel workbook tool, which is being prepared in conjunction with the USDA's update to *Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory* (Murray et al., n.d.; Stockmann 2022).

Additional Approaches for Evaluating Climate Change Mitigation Potential of HWPs

The following sections of this synthesis highlight additional ways in which HWPs can be evaluated for their climate mitigation potential. The IPCC carbon estimation approaches for HWPs are focused solely on the fate of biogenic carbon and do not include the climate impact of wood (positive or negative) that are realized in other national inventory report sectors. Fossil fuel emissions associated with harvest, transport, and manufacturing of HWPs (i.e., **embodied emissions**) are included in other GHG reporting sectors, such as the transportation and energy sectors. HWPs can have a lower **carbon footprint** when compared with other, more energy-intensive building materials, such as cement and steel (Bergman et al. 2014b, Leskinen et al. 2018, Lippke et al. 2004, Puettmann and Wilson 2005, Puettmann et al. 2010, Sathre and O'Connor 2010). Use of innovative wood products, such as cross-laminated timber, provides opportunities for wood use in place of cement and steel in high-rise buildings. In addition, residues, discarded HWPs, and captured emissions from landfills can be burned for energy, potentially avoiding fossil fuel emissions for energy production (i.e., **wood energy substitution**). As a result, most projection modeling that evaluates carbon stocks resulting from different levels of harvest or mixes of HWPs also try to account for the global warming impact of HWPs and alternative products or energy sources that might have been used instead. Together, consideration of the carbon stored and emitted from the forest ecosystem and HWPs, including the emissions associated with the production, use, and disposal of HWPs, as well as carbon tradeoffs with more fossil fuel-intensive products, can provide a more complete system assessment on possible climate mitigation strategies.

Changes to current harvest levels and wood utilization practices can have economic and ecological implications across the forest sector, thereby affecting tradeoffs between forest ecosystem and HWP carbon stocks. For example, reduced harvest in area A could increase forest ecosystem carbon stocks in area A but could result in reduced carbon stocks due to increased harvest elsewhere in area B (i.e., **leakage**). In contrast, **spillage** (e.g., Nepal and Skog 2014), or spillover (e.g., Magnani et al. 2009), is another type of leakage and refers to the unintended positive consequences of forest carbon management in area A on carbon storage in area B. Leakage and spillover could occur in many other alternative HWP consumption scenarios as well. If the life of HWPs is extended through cascaded utilization, demand for raw materials might be reduced, even though there could be an overall carbon benefit due to longer use of a given product. This could result in increased forest ecosystem carbon stocks resulting from decreased harvest. However, this could also lead to decreased economic returns to forest landowners, which might encourage conversion of forest land to more profitable nonforest uses. The future societal demand for wood products (whether greater or less than current demand) could also be an important determinant of actual HWP carbon sequestration. A third scenario could occur when there is increased harvest due to higher timber prices resulting from higher demand for wood products (rather than a timber supply shortage that triggered increased prices). Higher timber prices might decrease forest ecosystem carbon stocks in the short term through additional harvesting but may increase carbon stocks in the long term if higher prices lead to investment in new plantations or intensified forest management activities, such as thinning or fertilization (Daigneault and Favero 2021, Nepal et al. 2016, Skog et al. 2014). Although this would be a positive outcome for carbon stocks, it would be considered an example of spillage because an unintended consequence of changing markets was an increase in timber prices and intensified forest management. Forest sector economic models have been used to incorporate these market dynamics into HWP carbon evaluations.

Additional models or analyses beyond HWP carbon estimation are required to evaluate tradeoffs like those described above. No single HWP model can currently incorporate all of these tradeoffs—embodied carbon emissions, substitution benefits, land-use changes, and market forces. All of these factors are relevant for a full system's perspective of HWP carbon and potential constraints. The following subsections review important considerations in evaluating the climate mitigation potential of HWPs and the questions currently being asked of HWP model frameworks and LCAs that are yet to be fully answered.

LCA and System Boundaries

An LCA can be used to gain a more complete picture of the environmental impacts, climate costs, benefits, and mitigation potential of any product or process, including HWPs. LCAs extend beyond the forest-generated ("biogenic") carbon to evaluate the energy, materials, and environmental impacts needed to produce and use wood products for every stage along the supply chain, including forest regeneration and management, harvest, transportation, manufacturing, use, maintenance, reuse and recycling, and final disposal (Heath et al. 2010, Lippke et al. 2011) (fig. 4). For example, energy is used during harvest to run equipment, such as chainsaws, feller bunchers, and skidders; to fuel log transport to mills; and to power saws, planers, and dryers during production. Furthermore, the production of fertilizers, pesticides, and herbicides during tree planting consumes energy resources. Operating equipment that additionally requires fossil fuels like diesel, which have a higher carbon footprint than those operating with renewable biofuels or renewable electric power, must be considered in the LCA equation.

Depending on the energy source, these emissions contribute to several critical environmental impact categories, such as acidification (e.g., sulfur emissions), eutrophication (nitrogen), smog (particulates), and CO_2 and other emissions that contribute to global warming. Since most energy sources are extracted, refined, and transported before use, those impacts are included in LCA accounting as well. LCAs (fig. 5) track the impacts to air, water, and land from harvesting trees and production of HWPs from the forest (i.e., cradle) to when wood leaves the production site (i.e., gate out) all the way through to final disposition (i.e., grave). However, CO_2 is by far the most significant GHG in wood product life cycles from forest cradle-to-gate (Puettmann and Wilson 2005, Puettmann et al. 2010). By measuring all the necessary energy and material inputs to the manufacturing of a product and quantifying the GHG emissions per unit of product, the carbon footprint of a product can be calculated (ISO 2018, Negro and Bergman 2019).

Therefore, while an LCA assesses the overall environmental impact of a particular product, a carbon footprint

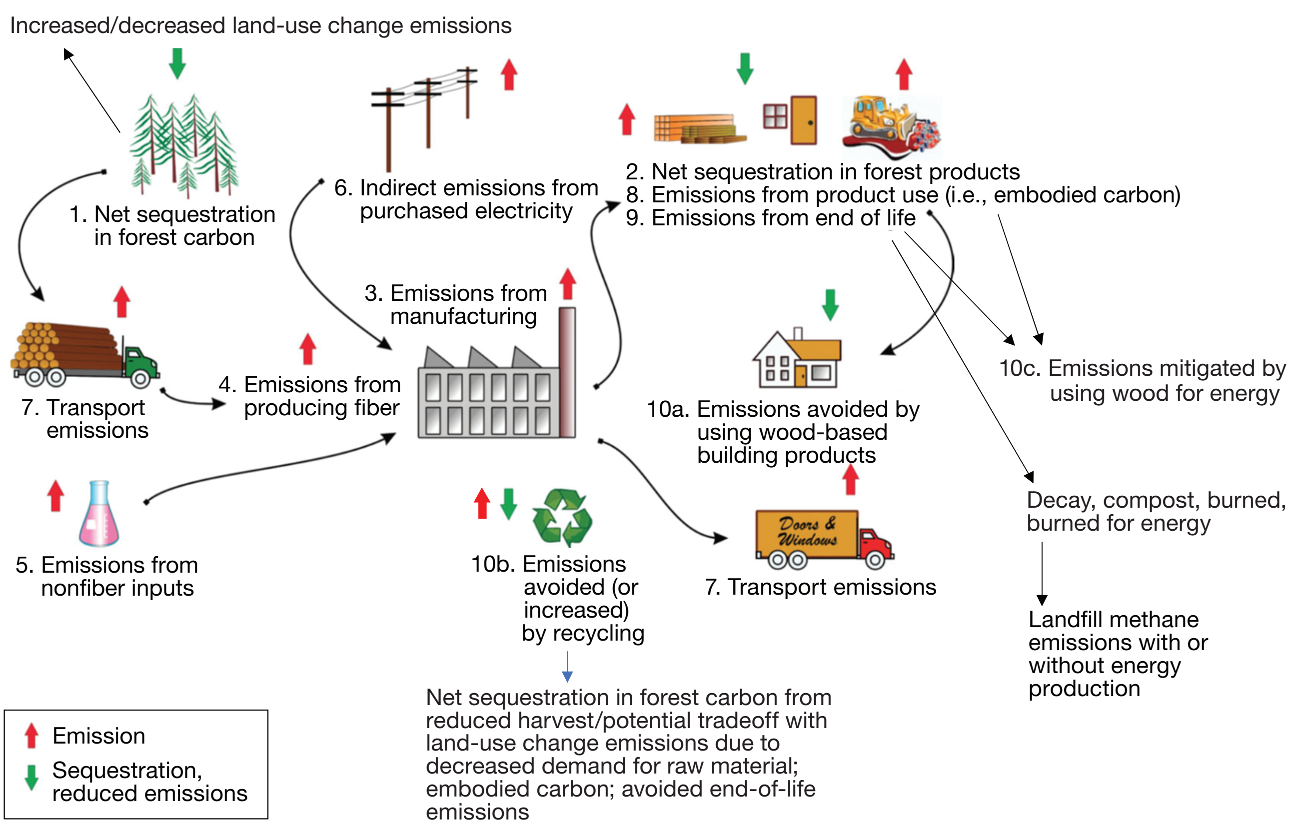

Figure 4—Depiction of emissions along a product's life cycle. Adapted from Heath et al. (2010).

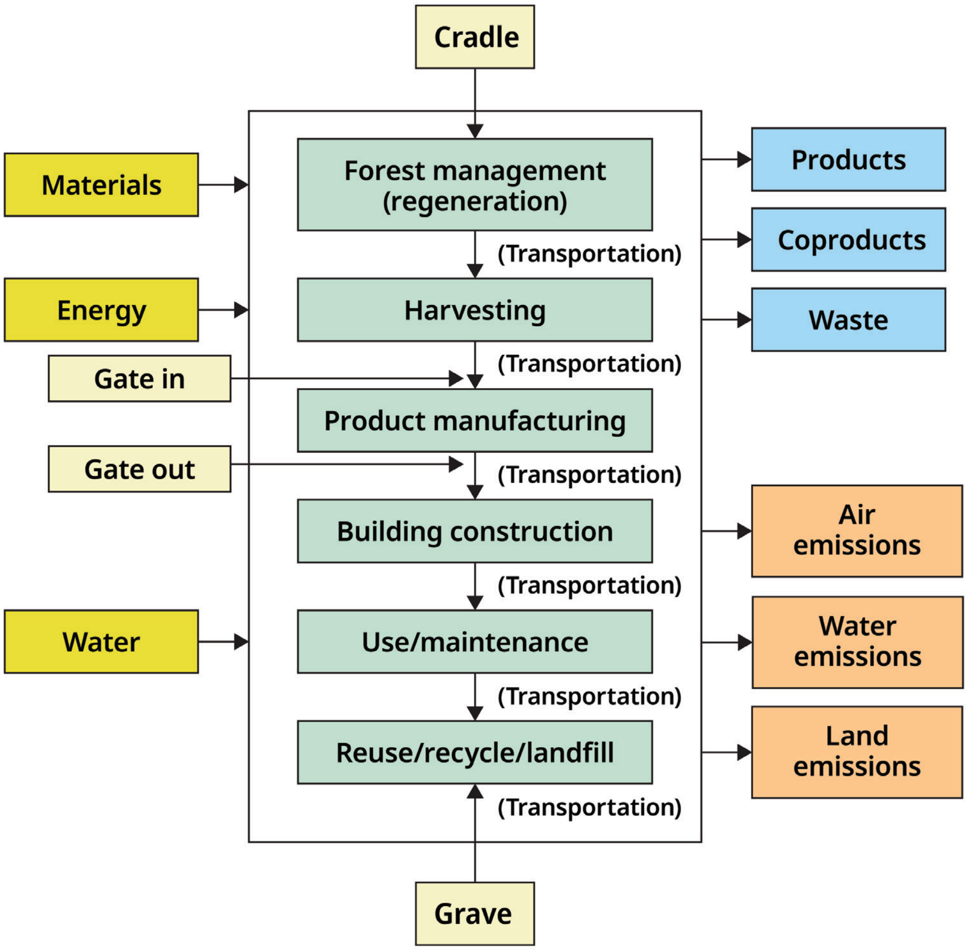

Figure 5—Full life cycle (cradle to grave) from forest management to disposal of wood. Adapted from Bergman et al. (2014a).

is the outcome of an LCA limited to emissions that have an impact on climate change.

Tracking embodied carbon emissions throughout a product's life cycle requires a comprehensive and detailed perspective as carbon flows from forests and associated harvested wood products are complex. In response to this complexity, the International Organization of Standardization (ISO) has developed and codified rigorous standards for conducting LCAs. There are publications that have stated they conducted an LCA, but the methods are not explicit and do not follow ISO standards, based on assertions by LCA scientists (Oneil et al. 2018). As defined by standards like ISO 14040 and 14044, LCAs are composed of four stages (phases):

1. **Goal and scope definition—**Establishes the system boundaries.
2. **Life-cycle inventory (LCI) analysis—**Tracks all raw material and energy inputs to manufacture a specific product, process, or service on a per-unit basis within precise system boundaries, such as raw materials (e.g., wood removed from harvest), energy inputs (e.g., heat, electricity, transportation fuels), and water inputs. Includes outputs, such as air emissions (e.g., CO_2, CH_4, particulate matter) and water emissions (e.g., suspended solids).
3. **Life-cycle impact assessment (LCIA)—**Uses these LCI flows to explore impacts for four areas: human health, resource depletion, social health, and ecosystem function.
4. **Interpretation—**Systematically evaluates scenarios for action to lower impacts (ISO 2006a, 2006b).

An LCA study includes all four stages, but an LCI study does not include an LCIA. LCAs are often referred to as "attributional" life-cycle assessment (ALCA) because they focus on current year net emissions for a given product or service instead of "consequential" life-cycle assessment (CLCA) where a "what if" scenario (e.g., changing an energy source) is evaluated (Bergman et al 2014a, Nepal et al. 2016). An ALCA is referred to as **"cradle-to-grave"** (raw material extraction to waste disposal) analysis. This assessment quantifies the environmental footprint on a per-unit basis using LCI data from individual life-cycle stages that can be used to compile results over most or all of the life cycle depending on the scope. An ALCA can locate "hot spots" of environmental impacts within the product system (cradle-to-gate) to provide information for manufacturers (decision makers) regarding process improvements and design (Gaudreault et al. 2010, Thomassen et al. 2008).

An ALCA stays within precisely defined boundaries, whereas a CLCA does not. For example, a sawmill produces planed dry lumber as its final product while producing coproducts, such as green sawdust from the freshly-cut logs or dry shavings from planing the rough-dried lumber (Bergman and Bowe 2008, 2010, 2012). The sawdust is burned for fuel onsite or sold offsite. In an ALCA of lumber production, coproducts, such as sawdust, are not tracked once they leave the system. These coproducts still carry a certain percentage of environmental impacts from the manufacturing stage. Impacts from sawdust that has left the system are considered accounted for in secondary products, such as particleboard or wood pellets (Reed et al. 2012). An ALCA examines the situation in a moment of time or a "snapshot," while the basis for a CLCA could be a market decision. For example, the market decision could be to sell more sawdust to particleboard manufacturers at a higher price and buy cheap natural gas to fire the boilers to kiln-dry the lumber. Selling the sawdust affects the environmental profile of the energy generation process and the two downstream unit processes tied to the lumber product after sawing, drying, and planing. The resultant LCIA GHG impacts to produce a unit of planed dry lumber would be higher for selling sawdust than for burning it for energy onsite because burning natural gas as a substitute energy source would emit fossil CO_2.

Mill surveys are the main tool to collect primary data for establishing the carbon footprint of HWPs, along with site visits for developing the data- and time-intensive LCIs. Life-cycle mill surveys are comprehensive questionnaires that request detailed energy, material, and emission data on unit process operations, such as log transport, sawing, drying, and planing for lumber production. All of this information is needed to accurately define the equations used to perform LCAs. The Consortium for Research on Renewable Industrial Materials (CORRIM) developed research guidelines on conducting LCA research on wood products (CORRIM 2010), which have been supplanted by product category rules. Product category rules ensure that LCAs align with product standards to generate ecolabels, referred to as environmental product declarations (ISO 21930), which are important for the building industry (Bergman and Taylor 2011).

The USDA Forest Service Research and Development program and CORRIM have been collaborating to perform wood product LCIs and LCAs. CORRIM is the premier LCA organization for wood products in the United States and has developed a large repository of forestry and forest product LCIs and LCAs (Boyd et al. 1976, CORRIM 2005, Puettmann 2010). These analyses consistently show that many wood-based materials use less fossil fuels to produce than alternative materials (Bergman et al. 2014b, Puettmann and Wilson 2005, Puettmann et al. 2010). Most of these LCAs were performed on structural wood products in which the underlying LCI data were placed into on-demand whole-building LCA software tools like the Athena Impact Estimator for Buildings (Athena Sustainable Materials Institute 2019) and Tally® (Building Transparency et al. 2023) for use by building developers and architects. This information can also be used to develop SFs (also referred to as "displacement factors") for evaluating climate mitigation benefits of different wood utilization scenarios.

While most HWP estimation models capture the carbon storage and losses at different stages along the lifespan of individual products, they are not designed to capture the whole picture analysis of a product that includes embodied carbon emissions or substitution benefits as an LCA does. However, additional analyses are necessary to apply individual product-level LCAs to broader national forest-, regional-, state-level, or national HWP carbon assessment efforts to better understand the overall climate impact associated with existing HWPs (Cabiyo et al. 2021, Smyth et al. 2017). Research is underway to incorporate cradle-to-gate LCAs and SFs into a new Excel workbook tool being prepared in conjunction with the USDA's update to *Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory* (Murray et al., n.d.; Stockmann 2022).

The following case study provides an example of using LCAs to evaluate the embodied carbon emissions associated with state-level HWP production.

Case Study: Embodied Carbon of HWPs In California, 2019

The embodied carbon emissions of HWPs were estimated for 2019 for wood products harvested and manufactured in California and represents cumulative GHG emissions (CO_2, CH_4, and nitrous oxide [N_2O]) during the cradle-to-gate stages involving forest harvest and transportation of logs to mills), and manufacturing of HWPs. The estimates were derived from the harvest volumes in 2019 (UM-BBER 2021), and these volumes were converted to various timber product categories (e.g., sawtimber, pulpwood, fuelwood) based on the estimated ratio of timber products output to harvest volumes. The estimated timber products output volumes were further converted to primary product categories (e.g., lumber, panels) based on the ratio of primary product output per unit of timber product (Morgan et al. 2012). The embodied carbon emission of each type of wood product at mill gate was collected from published literature, especially wood product LCA studies by CORRIM and USDA Forest Service Forest Products Laboratory. These values are provided in the full assessment of this case study (Khatri et al. 2023). The embodied carbon emissions of HWP categories, such as nonstructural panels (particleboard, medium-density fiberboard, and hardboard) and other industrial products (laminated veneer lumber, laminal strand lumber, glued-laminated timber, cross-laminated timber, and I-joists), were estimated by multiplying the embodied carbon emissions (kilograms of CO_2 per cubic meter) of a unit of each HWP by the total volume (cubic meters) produced for each product category. The embodied carbon emissions of all HWP categories produced were then summed to estimate the total embodied carbon of all HWPs produced in California. In addition, the amount of carbon stored in each HWP category was calculated by multiplying their carbon content with the weight of the individual wood product and converting it to CO_2 equivalents (CO_2e).

In 2019, around 11 million m^3 of HWPs were produced in California, with an estimated total embodied carbon emissions of about 1 million tons (tCO_2e) (fig. 6), suggesting an average embodied emission of about 91 kg CO_2e/m^3. While lumber and fuelwood were the largest in total wood product by volume (41 and 29 percent, respectively), their contribution to total embodied carbon was relatively small (about 21 and 6 percent, respectively). In contrast, nonstructural panels comprised more than 50 percent of total embodied carbon (0.55 million tCO_2e) despite their smaller share of total wood product volume (12 percent). Nonstructural panels are a highly processed product, which contributes to their overall higher embodied emissions compared to other HWP categories. The carbon stored in the wood products harvested and manufactured in 2019 was 6.2 million tCO_2e, resulting in an average of

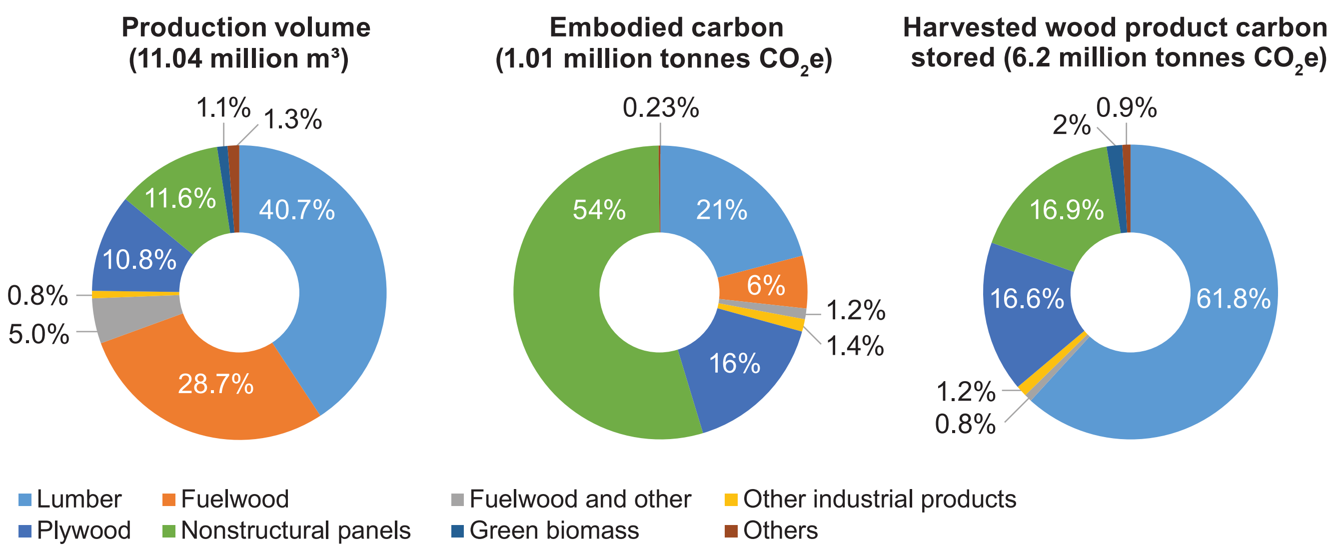

Figure 6—Production volumes, embodied carbon, and stored carbon of different wood products harvested and manufactured in California in 2019. The “others” category includes primary products that were less than 0.5 percent of the total volume, such as nonstructural panels, plywood, poles, and hardwood fuelwood.

564 kg CO_2e/m^3 of HWP production. Of all HWP categories, lumber, nonstructural panels, and plywood stored over 95 percent of the carbon. The analysis suggested that the cradle-to-gate embodied carbon emissions of all HWPs produced in California in 2019 reduced its overall HWP carbon storage benefits by about 16 percent.

Substitution (Displacement) Factors

To help identify tradeoffs with wood utilization scenarios, SFs are usually provided as a measure of avoided or increased GHG emissions if a wood product is used instead of another building material to provide the same function (Leskinen et al. 2018, Sathre and O'Connor 2010). Calculating SFs requires a formula in which the climate benefits of using wood products can be quantified. These factors are especially helpful for determining the climate mitigation potential of choosing to use one or more wood products instead of alternative building products. The equation to estimate the SF is the following (Leskinen et al. 2018, Sathre and O'Connor 2010):

$$SF = \frac{GHG_{nonwood} - GHG_{wood}}{QWU_{wood} - QWU_{nonwood}}$$

where $GHG_{nonwood}$ and GHG_{wood} are the LCA-estimated amounts of GHG emissions embodied in nonwood and wood alternatives, respectively. QWU_{wood} and $QWU_{nonwood}$ are the quantities of wood used in wood and nonwood alternatives (or less wood and more wood alternatives, respectively). QWU is usually expressed in terms of mass unit of carbon emissions per mass unit of carbon contained in additional wood. However, an SF is ultimately a unitless ratio when both the GHG emissions and wood use are expressed in mass units of carbon (Leskinen et al. 2018). Such an SF is estimated based on an integrated LCA study that ideally considers all fossil GHG emissions to the atmosphere from the wood and nonwood product systems during their different life-cycle stages, including production, use, cascading, and end of life (Leskinen et al. 2018, Sathre and O'Connor 2010). SFs for woody biomass energy in place of fossil fuel energy can be calculated in a similar fashion.

To assess potential substitution benefits for groups of individual products, it is necessary to compare SFs for mixed volumes of different HWPs and selected alternatives under different scenarios (Leskinen et al. 2018). From a synthesis of 21 studies, Sathre and O'Connor (2010) reported an average SF of 2.1 tons of carbon (tC) emission reduction per tC in wood product, with a wide range of SFs from −2.3 to 15 tC/tC. However, the system boundaries and products in studies differ, making the applicability of an average SF challenging; for example, some studies included forest ecosystem emissions. Leskinen et al. (2018) conducted a meta-analysis of 51 studies and estimated an average SF of 1.2 tC/tC across all products and life-cycle stages, with most values falling between of −0.7 to 5.1 tC/tC. An analysis conducted by Smyth et al. (2017) examined two scenarios for wood energy substitution and several wood-intensive and less wood-intensive scenarios using national consumption statistics to weigh SFs and assess overall substitution for the forest industry in Canada (fig. 7). An average SF of 0.47 and 0.89 tC/tC, respectively, for the two wood energy substitution scenarios was estimated. An average SF of 0.54 tC/tC was estimated for sawnwood and 0.45 tC/tC for panels in material substitution scenarios. Smyth et al. (2017) also reanalyzed Sathre and O'Connor's (2010) data using the same system boundaries for sawnwood and panels and found comparable results (0.51 and 0.14 tC/tC, respectively).

Most of these studies did not consider leakage or spillage effects associated with the increased use of wood in place of nonwood materials, suggesting that avoided emissions should be evaluated in conjunction with carbon stock changes in forests and HWPs, substitution impacts, and cross-sectoral fossil fuel use over time (Harmon 2019, Leskinen et al. 2018, Nepal et al. 2016). An SF cannot provide enough information for policy decisions regarding mitigation strategies at a state or national scale on its own. An SF is also limited in the context of policy decisions because they do not account for basic market forces like pricing; supply and demand; or cultural, technical, and capacity constraints (Howard et al. 2021). Furthermore, SFs are typically considered static over time, ignoring the likelihood of decarbonization of the nonwood sectors (Forster et al. 2021). If a jurisdiction is a net exporter, and if substitution occurs, the benefits will occur elsewhere, potentially leading to double counting. Nepal et al. (2016) considered these important dynamics, including price effects on forest land use and management and subsequent effects on forests and HWP carbon pools over time, in estimating an SF resulting from projected

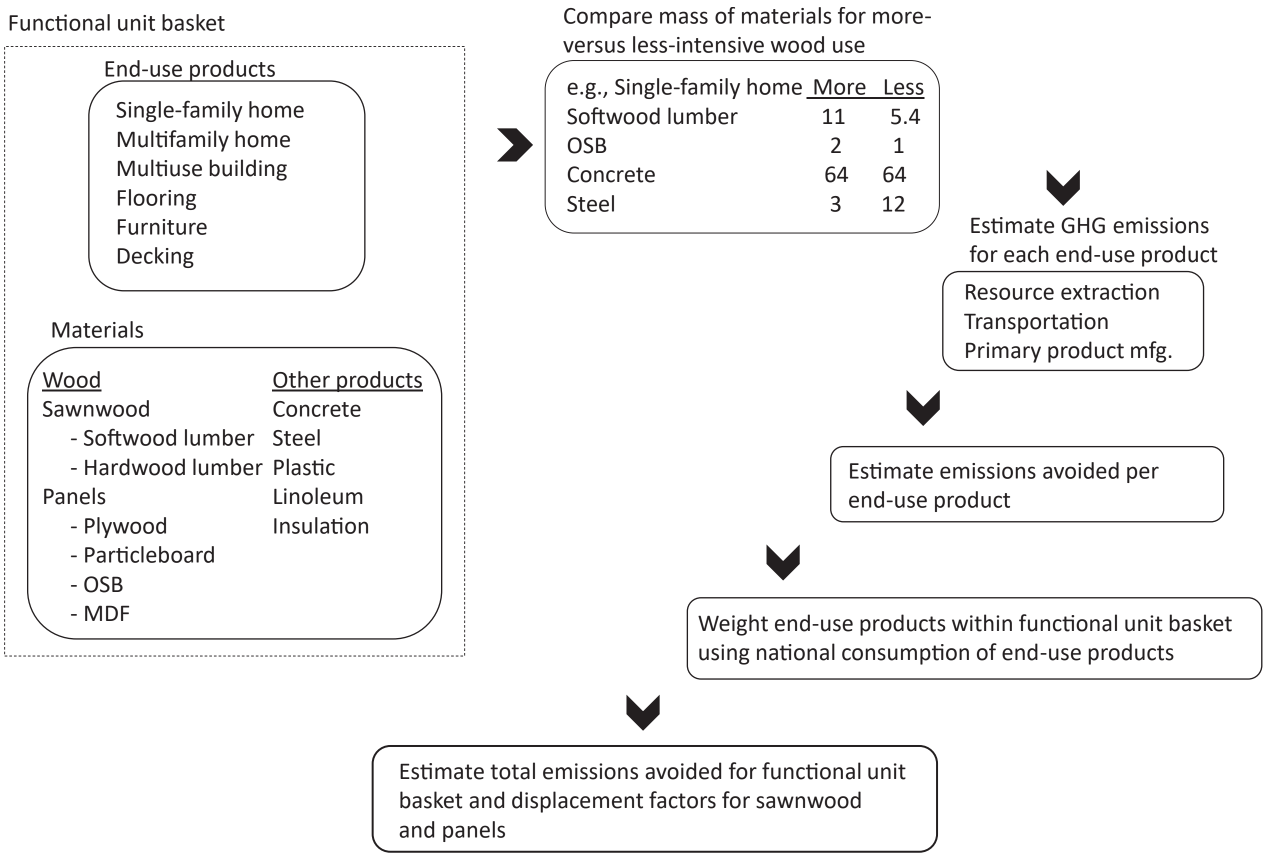

Figure 7—Smyth et al.'s (2017) overview of their method for calculating wood product substitution (displacement) factors at the Canadian national scale, including an example of substitution factors for a single-family home construction. GHG = greenhouse gas; MDF = medium-density fiberboard; OSB = oriented strand board.

increase in demand for softwood lumber and structural panels for use in low-rise nonresidential buildings in the United States over 50 years. Their estimated U.S. aggregate SF considering these dynamics was 2.03, suggesting 2.03 tons of CO_2 equivalent (tCO_2e) reduction per tCO_2e (tCO_2e /tCO_2e) of extra wood used in nonresidential buildings.

The use of SFs alone does not clarify whether or when substitution is occurring in real time. Nevertheless, it is important to identify realistic substitution opportunities. For example, the opportunity for additional wood use in U.S. single-family housing construction is very low because wood is already the preferred building material, although there is greater opportunity for increased wood use in the construction of low-rise nonresidential buildings, in multifamily buildings, such as condominiums, and in nonstructural elements, such as wood fiber insulation (Nepal et al. 2016, Xie et al. 2021). Understanding the demand for these buildings can help identify a more realistic potential for high-rise buildings and associated substitution benefits. These types of analyses are necessary for stimulating interest in and a robust market for mass timber, such as cross-laminated timber, or other engineered wood products, like glulam. Weighing tradeoffs between traditional and innovative wood uses, such as lumber and cross-laminated timber, and including changes in market demands requires the use of models that extend beyond the forest sector.

Understand the Relationship Between HWP Economics and Carbon Dynamics

Economic drivers, such as supply, demand, prices, and trade of alternative products influences HWP carbon dynamics (Brander et al. 2009; Daigneault et al. 2012; Nepal et al. 2014, 2016, 2019; Sedjo and Tian 2012). These economic factors influence whether carbon stays in forests or is removed for HWP production. The interactions between economic drivers are important to consider when evaluating policy changes that have upstream and downstream forest sector impacts, such as promoting demand for wood in specific end-uses (e.g., bioenergy or construction). Furthermore, because HWPs are globally traded commodities and industrial forest land can be an international investment, HWP production and consumption are influenced by global market fluctuations. Understanding how HWPs fit into global markets is critical for quantifying and analyzing carbon benefits because it is primarily markets and timber prices that influence large-scale or commercial harvests.

Policy changes are likely most effective when based on comprehensive analyses of timber markets that determine where and when harvest will take place and whether forests will receive the active management that increases the likelihood of remaining forest land. Most forest sector economic models are **partial equilibrium models** and assume that the forest sector is too small to affect other sectors of the economy. However, cross-sectoral economic impacts, such as reduced use of fossil fuels in construction materials, can be evaluated using **computational general equilibrium** economic models that can capture the potential direct and indirect market interactions and feedback effects across all economic sectors of a change in an economy (e.g., increased demand for wood use in construction). Several forest sector economic models (table 6) enable modeling of market processes that simulate change in forest investment (i.e., intensity of forest management or land-use change) in response to change in timber prices that result from change in demand for a forest product or from a reduction in timber supply (Daigneault and Favero 2021; Daigneault et al. 2012; Nepal and Skog 2014; Nepal et al. 2016, 2019; Sedjo and Tian 2012). These models link market dynamics, forest growth, and harvest. However, forest dynamics are often oversimplified because growth rates are applied at regional or global scales, disturbances beyond harvest (e.g., fire and insect outbreaks) are not well represented, and applicability to uneven-aged stands is difficult, if not impossible, with some models (Wear and Coulston 2019).

The models listed in table 6 are some of the most commonly used forest sector economic models in the United States. All models include international trade of wood products except for the Subregional Timber Supply Model (SRTS), which focuses on the Southern United States. All models also project forest carbon stock changes or changes to forest area. In addition to carbon stock changes, both the Timber Supply Model and Forest and Agriculture Sector Optimization Model with Greenhouse Gas Accounting (FASOM-GHG) model simulate competition for land between agricultural and forest demands. These models incorporate complex market components from U.S. regions and globally, including carbon flux or accounting, GHGs, and even land-use change; however, the forest growth models are generally simplified. Future forest management and policy questions around climate change may be best addressed if they consider variable economic scenarios as much as forest carbon flux scenarios because of market pressures on HWP supply and demand.

Studies simulating HWP market processes suggest that increases in demand for HWPs would result in increased product prices, which would also affect the upstream and downstream markets of HWP supply and value chains (Johnston and van Kooten 2016; Nepal et al. 2016, 2019). Higher prices paid to forest landowners could potentially provide economic incentives to invest in current or alternate forest management activities (including an incentive for forest plantations) (Daigneault and Favero 2021, Latta et al. 2016). Increased timber prices can also increase the value of forests, which can help prevent forest land conversion to other uses (Miner et al. 2014, Nepal et al. 2014). These market-induced indirect effects on forest management or land use that result in higher forest growth, and therefore higher forest carbon stocks, are termed "spillage" and were discussed in detail in "Additional Approaches for Evaluating Climate Change Mitigation Potential of HWPs" (Miner et al. 2014, Nepal and Skog 2014, Nepal et al. 2019).

Given some of the challenges described above and the simplification of growth and yield components in economic models, forest sector economic models could be used in conjunction with other models to gain an understanding of the full spectrum of impacts on, or from, the forest sector. A CLCA study, in conjunction with a forest sector economic model (table 6) and a model tracking carbon in forests ecosystem and HWP pools (table 2) can facilitate such evaluations. The reverse is valid as well; climate models, forest ecosystem carbon models, and HWP models could also be combined with forest sector economics models because market forces drive short- and long-term implications for forest carbon.

Table 6—Forest sector economic models in the United States that include carbon dynamics

Model name	Spatial scale	Accessibility	Description
Global Forest Products Model (GFPM) (Buongiorno 2015, Nepal et al. 2021)	Global	Free access online https://buongiorno.russell.wisc.edu/gfpm	Dynamic global partial market equilibrium model. Predicts production, consumption, trade, and prices for 180 countries using 14 types of harvested wood products. Model projections of production, consumption, and trade of forest products can be linked to global-scale harvested wood product carbon models (e.g., Johnston and Radeloff 2019, Nepal et al. 2022) to estimate harvested wood product carbon for individual countries or to WOODCARB II (Skog 2008) to estimate harvested wood product carbon for the United States.
United States Forest Products Model (USFPM/GFPM) (Ince et al. 2011)	Global with U.S. scales	Not publicly available	Dynamic global partial market equilibrium model. Predicts production, consumption, trade, and prices for 180 countries with greater product details for the United States. USFPM can run at a regional scale within the United States (three regions) using 20 types of harvested wood products. Model projections of U.S. production, consumption, and trade of forest products can be linked to WOODCARB II (Skog 2008) to estimate harvested wood product carbon for the United States (e.g., Nepal et al. 2012, 2016). The USFPM was used to investigate the forest sector effects of various socioeconomic and climate scenarios for the 2010 Resources Planning Act Assessment but has since been replaced with the Forest Resource Outlook Model.
FOrest Resource Outlook Model (FOROM) (Johnston et al. 2021)	Global with sub-U.S. regional scales	Not publicly available Currently used for the Resources Planning Act Assessment	Global recursive dynamic partial equilibrium model. Predicts production, consumption, and trade of 20 harvested wood products (primary and secondary products) across 6 regions of the United States as if they are separate scales within international trade. Using base year inputs, projects future market variables across different socioeconomic scenarios. Changes to forest stocks and area also predicted. Projects changes to stocks and area based on climate specifications and greenhouse gases. Model projections of production, harvested wood product, and trade of forest products can be linked to global-scale harvested wood product carbon models (e.g., Johnston and Radeloff 2019, Nepal et al. 2022) to estimate harvested wood product carbon for individual countries.
Timber Supply Model (Daigneault et al. 2012)	Global with sub-U.S. regional scales	Not publicly available	Global dynamic economic forest sector model with carbon accounting framework that predicts log supply and demand (for land—including agriculture) for 200 forest types, 16 different regions of the globe, and 5 U.S. regions. Demand for harvested wood products influenced by changes in income over time as well as price fluctuations for maintaining forests in different forest types. Estimates ecosystem carbon and harvested wood product carbon (and decay) from base year 2010 but not historical harvested wood product carbon.

Table 6 (continued)—Forest sector economic models in the United States that include carbon dynamics

Model name	Spatial scale	Accessibility	Description
Land Use and Resource Allocation Model (LURA) (Latta et al. 2016)	U.S. and subnational scales	Not publicly available	Partial spatial equilibrium model of U.S. forest sector using 22 harvested wood products. Demand influenced by energy prices and U.S. Energy Information Administration's annual energy outlook as baseline (gross domestic product, housing, diesel prices), a low economic growth scenario, and a high economic growth scenario. Tracks international harvested wood product trade, live tree carbon, and carbon flux at regional and ownership scales.
Forest and Agriculture Sector Optimization Model with Greenhouse Gas Accounting (FASOM-GHG) (Beach et al. 2010)	U.S. and subnational scales	Not publicly available	Dynamic nonlinear model that includes agriculture and forest sectors in the United States using 46 harvested wood products, including primary, secondary, and bioenergy feedstock. Model constraints: total production must equal total consumption and total land use remains constant. Model predicts competition between agriculture, livestock, and forest landownership over time and impacts on commodities produced. Simulates land-use change, change in production, and greenhouse gases (N_2O, CO_2, CH_4). International trade (imports and exports) included.
Subregional Timber Supply Model (SRTS) (Abt et al. 2009)	Southern United States	Not publicly available	Projects timber inventory, removals, prices, and demand for bioenergy for the Southern United States. Developed for subregional, ownership type and forest type supply analysis in the Southern United States using 6 harvested wood products. USDA Forest Service Forest Inventory and Analysis data inputs used for forest growth and harvest inventory.

Carbon Leakage and Spillage in HWP Markets

The magnitude of market-induced responses on forest management and land use may be limited in many regions because of differing biophysical, economic, and policy contexts. Forest sector economic models have been used to evaluate how increased price and demand for HWPs can counterintuitively lead to greater forest carbon stocks (i.e., spillage). Using the Timber Supply Model, an integrated dynamic global economic model of forest and land-use management, Daigneault et al. (2012) evaluated the forest carbon impacts of hypothetical U.S. bioenergy policies that promoted increasing the use of forest biomass. They suggested that expanded bioenergy use reduced net global carbon emissions because higher wood prices (from increased demand) led to new investments in intensified forest management/plantations (Daigneault et al. 2012). Similar findings were reported by Nepal et al. (2015), who used USFPM/GFPM to estimate net carbon impacts of expanded bioenergy use in the United States. In a scenario in which wood energy use was about sixfold higher than the baseline case, they found that an additional 5.6 million ha of pine forests would be planted by 2060. Carbon accumulated in these new pine forests offset about 80 percent of the reduced forest carbon caused by increased harvest for biomass over the period of 50 years.

In another study, Nepal et al. (2016) integrated the USFPM/GFPM and the Subregional Timber Supply Model (SRTS) to show market processes that simulate investment in forests in the form of planting new forests, mainly in the Southern United States. In their model, timber price increases due to increased demand for lumber and structural panels for use in nonresidential construction led to an additional 0.60 million ha of forest land (relative to reference scenario) in the Southern United States by 2060. The examples of Nepal et al. (2015, 2016) examined individual scenarios, such as increased bioenergy, or increased demand for wood in nonresidential construction. To date, these examples have not been directly compared to gain an understanding of which scenario would have a greater overall climate benefit. Additionally, these analyses do not consider how direct climate impacts and disturbances beyond harvest would affect the forest carbon dynamics associated with increased acreage of forest land. However, these models are useful for exploring how markets influence forest carbon dynamics and some of the unintended consequences of changing timber prices.

Forest sector economic models can help bridge HWP carbon estimates by providing information on potential unintended carbon leakage or spillage. Economic models can provide such information by providing estimates of price-induced changes in timber harvests, land use, production, consumption, and trade of wood products across different regions due to change in a policy affecting forest products demand in a particular region (Murray et al. 2004, Nepal et al. 2013, UNECE and FAO 2021, Wear and Murray 2004). Former estimates of leakage and spillage have ranged from −10 to 100 percent, depending on contextual factors (Pan et al. 2020). For example, estimated leakage potential for policies and programs that constrained harvest in the United States ranged from 71 to 88 percent because of increased harvests in non-set-aside/unrestricted forests (Daigneault et al. 2012, Nepal et al. 2013, Sohngen and Brown 2008). Leakage within the forest sector due to harvest shifts, also called "cross-sectoral leakage," (Harmon 2019) may occur if wood is substituted for nonwood materials. Harmon (2019) suggested that if wood replaced more carbon-intensive concrete or steel in the construction sector, the avoided use of fossil fuels may end up being used in another sector.

The interplay between timber supply, wood demand, product prices, land-use change, and subsequent potential for leakage and spillage are important considerations for climate mitigation strategies. For example, in California most HWP demand is being met by imported products (Christensen et al., n.d.). Targeting forest fuels management by increasing harvest could potentially reduce wildfire risk and improve the stability of the forest ecosystem. Increasing harvest may also meet more of the wood demand in California; however, forest fuel removals would likely include a large proportion of smaller stems (smaller than sawlog size). Increasing harvest volume to meet demand could influence forest ecosystem and HWP carbon dynamics in regions that export wood to California (e.g., Oregon and British Columbia). Alternatively, if exporting regions seek to decrease harvest to increase forest ecosystem carbon stocks, this could lead to increased harvests elsewhere or land-use changes within those regions (subject to local and regional constraints on land-use change). The forest ecosystem and HWP markets are complex, and these are just a few examples of pathways worth studying in the future. Finding ways to model these interactions, as well as others, is critical to developing effective climate mitigation policy.

Timing and Tradeoff Considerations at Different Spatial and Temporal Scales

Even with high-quality data, addressing different temporal and spatial scales are complicated for any carbon model. This is particularly true for projection scenarios because there are more uncertainties the further that projections are modeled into the future. Carbon stocks in a forest stand can fluctuate greatly as it cycles through regeneration, growth, maturation, and harvest over time. At a landscape scale, however, carbon stocks can be quite stable if tree growth, mortality, and harvest have reached a relative equilibrium (McKinley et al. 2011). Similarly, future projections of forest and HWP carbon stocks and emissions under alternative management scenarios may change over several decades, and the resulting GHG warming potential at different points in time could affect the outcome of analyses as GHGs accumulate and persist in the atmosphere, are reabsorbed, break down, and decay (Ganguly et al. 2020, Nepal et al. 2014). Temporal and spatial scales are key to interpreting contextual relevance between different carbon impact analyses.

The magnitude of climate mitigation from HWPs depends on model choice and the spatial and temporal extent of the analysis (Salazar and Bergman 2013, Smyth et al. 2020, Xu et al. 2018). The amount of time needed for a forest to sequester carbon removed during harvest (i.e., reach **equilibrium state**) or achieve parity with a baseline can range from a few years to more than centuries depending on a number of factors, such as species, tree size, percentage of trees removed, site productivity, disturbance regimes, prior land use and management, type of products that were produced (e.g., sawlog or pulpwood), assumptions about wood products replacing nonwood products (e.g., steel, concrete, plastic, textile), or the amount and types of fossil fuel technology (e.g., coal, natural gas, or oil) that wood energy replaced (Lamers et al. 2013; McKechnie et al. 2011; Nepal et al. 2014, 2016; UNECE and FAO 2021; Zanchi et al. 2012).

One analysis considered potential effects of a wood energy consumption scenario that was about five times higher than a reference scenario on forest land use, biomass regrowth on forest land, carbon storage in HWPs, and carbon stored in logging slash left to decay in the forest (Nepal et al. 2016). They evaluated the carbon impacts of this increased wood energy consumption in the United States using a CLCA framework. At a U.S. aggregate scale, they suggested that an estimated 78 percent of the additional emissions created by burning wood for heat and power generation would be offset by forest ecosystem carbon storage over 50 years through biological and market-induced forest regrowth (and inventory). However, within this timeframe, projected forest ecosystem carbon offsets differed regionally. The Northern, Western, and Southern United States (as defined in Ince et al. 2011) were reported to offset 16, 50, and 95 percent, respectively, of increased emissions primarily because of regional differences in timber harvests, biological regrowth, market-induced changes in forest land use, and quantity of wood products manufactured. Nepal et al. (2016) highlights that even within the United States, there is spatial variation when analyzing climate mitigation scenarios using the same temporal scale. Another important takeaway from this study is that projections were made over a period of 50 years. Had the authors examined the same scenario over a shorter period (e.g., 25 years), it is possible that projected carbon storage would not offset the increased timber harvest or that offsets would be negligible depending on the region. From a policy perspective, temporal scale becomes increasingly important for analysis because model outputs change depending on the projected time horizon. Researchers are currently using both static and dynamic LCA approaches to investigate the cradle-to-grave embodied carbon associated with all wood products harvested and manufactured in California over a time horizon of 500 years with longer lifespans of HWPs (Khatri et al. 2023).

Studies conducted in Canada demonstrate that the magnitude of climate mitigation benefits varied over time and space within and among scenarios (Smyth et al. 2020, Xu et al. 2018). For example, in Smyth et al. (2020), scenarios that reduced harvest levels below existing levels showed stronger climate mitigation potential because of greater forest carbon stocks in the short term (10 years). However, in the long term (50 years), the mitigation potential decreased because of increased wildfire, and sink strength weakened as forests aged (changes in wildfire dynamics were not assumed for all scenarios). Short-term climate mitigation benefits were greatest in scenarios where there was increased production of longer lived HWPs without changes to harvest levels, and HWPs were assumed to have substitution benefits. Scenarios in which bioenergy use increased resulted in greater substitution benefits in the short term as fossil fuels were replaced; however, benefits decreased over time as renewable energy use increased.

In the examples above, maximizing climate mitigation potential varied temporally (i.e., short, mid, or long term) and spatially (i.e., domestically or globally), resulting in different climate mitigation expectations. To develop climate mitigation policy at a state or national level, comparing scenarios with several combinations of forest management, wood utilization, and substitution may be most useful (e.g., Smyth et al. 2020). Ideally, these analyses would incorporate inherited HWP carbon stocks and emissions from past management. Overall, if only domestic wood consumption and substitution benefits were considered, climate mitigation potential was not as powerful as when global wood consumption and substitution benefits were considered. Therefore, consideration of a time horizon, carbon emissions, and sequestration are important aspects of HWP carbon estimation and LCAs. The studies described above demonstrate how examining current carbon estimation objectives at smaller geographical scales might inhibit global mitigation goals. Furthermore, studies that used long-term carbon perspectives suggested greater mitigation potential than short-term perspectives; however, caveats about long-term perspectives have also been noted. Although there is no set rule regarding the appropriate timeframe to consider, the chosen frame for determining the net carbon impact of HWPs should be long enough to allow consideration of carbon change associated with establishing, growing, or regenerating the forest, including disturbances, land-use change and other management activities, and all emissions associated with all harvested wood products (Malmsheimer et al. 2011, Nepal and Skog 2014).

Evaluating Climate Mitigation from HWPs: Cascaded Wood Products

Studies have shown that extending the life of HWPs through cascaded products has climate mitigation potential (Dugan et al. 2021, Smyth et al. 2017, Suominen et al. 2017). Cascaded products were first introduced as a tool for achieving sustainability in resource use with applications for product design (Sirkin and ten Houten 1994). The goal was to enhance resource use efficiency by sequentially reusing the same resource unit for multiple applications followed by final disposal (Höglmeier et al. 2014). Frequently, the final step of a cascaded product network is energy generation (Risse et al. 2017). Odegard et al. (2012) defined cascading through three different lenses that are not mutually exclusive:

- **Cascading in time—**Sequential use of biomass, such as traditional paper recycling.
- **Cascading in value—**Prioritizing the highest value over the life cycle, such as repurposing waste wood for particleboard over bioenergy.
- **Cascading in function—**Optimizing coproduction of multiple product pathways to maximize total functional use, such as converting sawtimber to lumber, wood pulp, and mill residues, each with several resulting products. This is traditionally how HWPs are manufactured, as mills generally try to utilize as much harvested wood, and in the most efficient way possible.

Recovery options of lumber products could include reuse as lumber, reprocessing as particleboard, pulping to form paper products, or burning for energy recovery (Bergman et al. 2013, Gustavsson and Sathre 2006). Similarly, multiple cascading steps, such as reusing the lumber, followed by reprocessing to particleboard, followed by burning for energy, can result in additional carbon benefits. However, there are also many other product pathways, each with a unique network of energy uses and carbon balances (fig. 8). Even though cascaded products can extend the lifetime of wood products, a complete LCA would be needed to account for the embodied carbon to determine the net environmental benefits. Processing and transportation efficiency is important, as well as the number of end uses for HWPs and the choices of products to manufacture (Puettmann et al. 2010).

Cascading has been shown to offer benefits when used with relatively simple product networks. In some cases, like cascading products in which mill residues are used at sawmills (common practice in the Western United States.), the benefits of cascaded products are likely to be much greater because of the limited transportation and processing requirements of onsite residues. When considering complex scenarios involving more cascading stages and product types that include both energy and material benefits, the specific advantages are less clear (Höglmeier et al. 2014). For example, if secondary or tertiary products require long transportation distances or large amounts of fossil fuel energy to produce, the carbon emissions may outweigh the additional carbon storage. This underscores the importance of case-by-case evaluations of cascaded product networks. Garcia et al. (2020) indicated that when considering multiple products and recycling steps, both the biogenic carbon footprint and discard pathways must be considered. These factors are influenced by the timing of forest regrowth in the area that provided the primary wood. Gustavsson and Sathre (2006) quantified the following mechanisms by which cascading of wood products can affect energy and carbon balances:

- **Direct cascade effects—**Due to different properties and logistics of virgin wood versus recovered materials.
- **Substitution effects—**Due to the reduced demand for nonwood materials when wood is cascaded.
- **Land-use effects—**Due to alternative possible land uses when less timber harvest is needed because of wood cascading.

Market and policy forces can also result in cascaded products competing against one another, which can alter the potential carbon benefits. For example, reclaiming demolition lumber for energy generation may be preferred to reprocessing for wood composite manufacture (i.e., furniture) because of cost or government policies around bioenergy, resulting in tradeoffs between energy substitution benefits and longer carbon storage in HWPs (Jarre et al. 2020). Numerous studies also mention

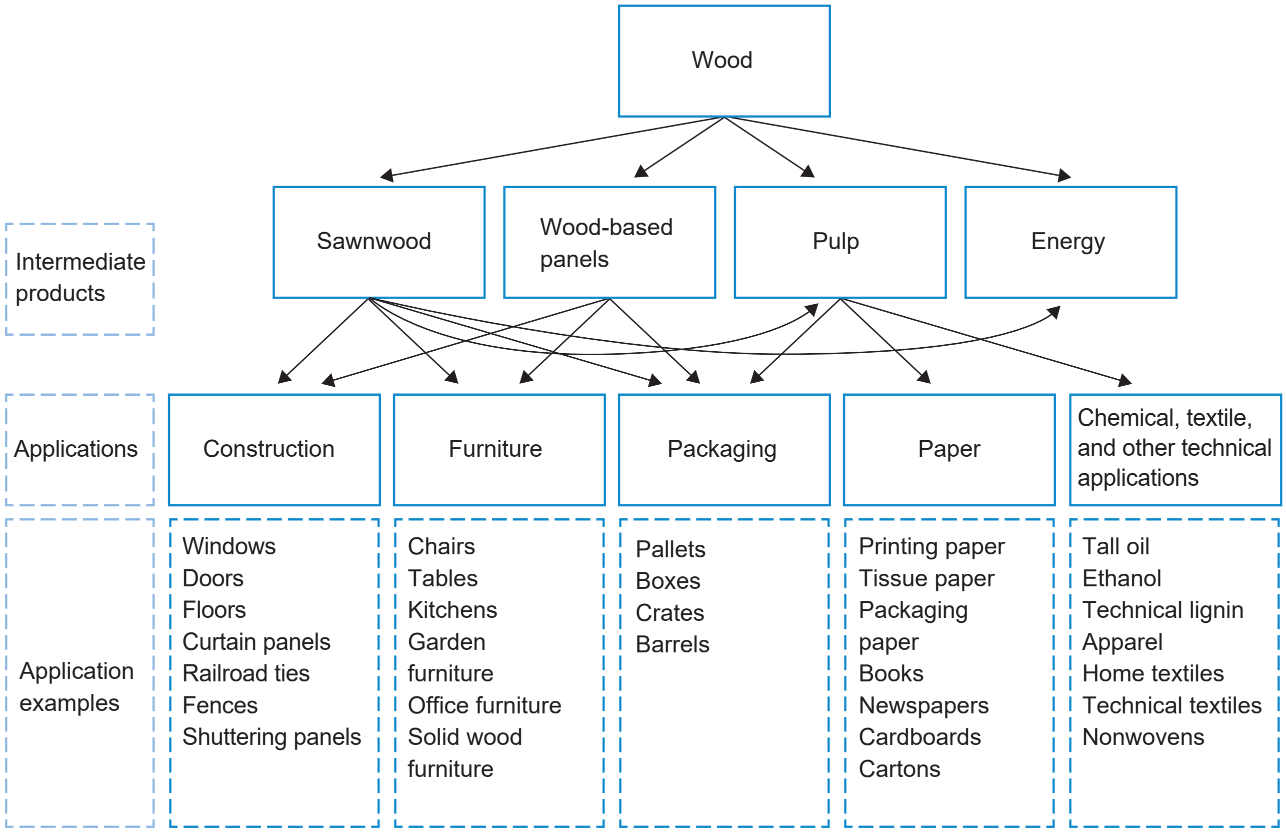

Figure 8—Cascaded product networks adapted from Leskinen et al. (2018).

land-use change and land-use accounting as a key consideration when evaluating tradeoffs from wood cascading scenarios (Gustavsson and Sathre 2006, Risse et al. 2017). Other research has suggested that cascaded wood scenarios lead to less resource extraction from the natural environment compared to the use of freshly harvested wood alone to provide the same products (Risse et al. 2017, Tashkiri et al. 2019).

Ultimately, in a market with no failures, the cascading strategies that are most effective and efficient will be adopted most widely. However, there will be tradeoffs when balancing total economic costs and GHG emissions for cascaded product strategies (Tashkiri et al. 2019). If wood use increases for mass timber construction, or products such as biofuels, biochemicals, and wood composites, a greater amount of wood will also enter waste streams and could potentially be repurposed for cascaded products. Successful cascaded product networks will likely depend on efficient and cost-effective technologies, sustainable forest management practices, well-defined markets, short transportation distances, a forest industry infrastructure capable of producing different types of product, and policy measures to promote the use of products with low-carbon footprints.

Complete LCAs for cascaded products are difficult to provide, given the differences in product types, product networks, temporal and spatial boundaries, and functional units. Compared to single product uses, developing carbon estimation models for cascaded products is more complex if a discarded product reenters the use stream as a different product. The number of possible products, the potential use of waste disposal streams for multiple pathways simultaneously, and the potential variability in decay rates, as well as the eventual fate of products (i.e., disposal ratios), complicate the ability to account for cascaded product streams. However, some HWP carbon models do have the capability to build in cascaded products (e.g., MitigAna). Developing separate and rigorous LCA models for each cascaded product pathway is a challenge. This is especially true when considering that a complete LCA framework must factor in land-use change and forest harvest practices among other variables. The HWP carbon estimation models evaluated in this synthesis address only the simplest form of cascaded products through primary product ratios and traditional recycling.

HWP Carbon Estimation Synthesis Summary Points: Q&A for Natural Resource Managers

This synthesis highlighted methods, required datasets, and models for different approaches to estimating carbon storage and emissions associated with HWPs. We also examined examples, limitations, and options for modeling tradeoffs. In addition to improving parameters for existing HWP carbon inventory model estimates, future decision-making pathways about alternative forest management and wood utilization scenarios could incorporate LCAs and integrated forest sector economic models. LCAs are critical for understanding the comprehensive carbon emission impacts of HWP use and can assess tradeoffs when specific products are compared to nonwood products or alternative HWP uses for consumers. However, LCAs are complex, requiring knowledge of material flows and energy inputs for timber harvests and for manufacturing, consumption, and disposition of wood products, among other factors. Carbon models for HWPs paired with forest ecosystem models can elucidate tradeoffs between forest management and disturbance effects on carbon in forests and use of harvested wood for energy or long-lived wood products. Forest sector economic models can incorporate market forces that ultimately affect forest ecosystem and HWP carbon pools under different management and utilization scenarios. There are limitations and caveats to every approach, and no single model can address all aspects of the climate mitigation potential of different wood utilization scenarios. The following are some key issues to consider before choosing an approach to quantify carbon in HWPs for evaluating the HWP climate mitigation potential under different forest management and product use scenarios:

1. **Which model(s) can be or has been parameterized for a selected system boundary?** This synthesis reviewed both USFS HWP-C v1 and subsequent variants as well as ANSE-based models, focusing mainly on NFCMARS-HWP. We selected and compared these two models because they have been used in the United States and Canada at national forest, state, regional, or national scales. USFS HWP-C v1 provides data templates based on U.S. regions defined in Smith et al. (2006). HWP-C vR (Groom and Tase 2022) provides the original regional templates as well as detailed datasets for California, Oregon, and Washington. The first author (T.K. Lucey) is currently parameterizing HWP-C vR for regions of New England. Several ANSE-based models have been calibrated for use at national and provincial levels in Canada, as well as for some U.S. states, including South Carolina, Vermont, Pennsylvania, Maryland, Wisconsin, and Minnesota (Government of Canada 2023). These models function similarly; however, outputs have yet to be compared using the same inputs for a given locale. ANSE-based models are currently being calibrated for California and Oregon (American Forests 2021), and a comparison to a variant of USFS HWP-C v1, such as HWP-C vR, could be possible in the future.
2. **Are recent data available for a selected model? What are the sources for these data?** Both USFS HWP-C v1 (including its variants) and NFCMARS-HWP and other ANSE-based models generally rely on the same datasets or similar datasets. This synthesis establishes common terminology for required data inputs:
   a. **Harvest quantity—**USFS HWP-C v1 and NFCMARS-HWP require annual harvest quantified by volume or mass. USFS HWP-C v1 uses historical records from public data sources. NFCMARS-HWP uses annual harvest volumes from empirical data combined with simulated harvest volumes from the forest ecosystem model, CBM-CFS3.
   b. **Timber and primary product ratios—**Both models use timber and primary product ratios to allocate harvest to products. USFS HWP-C v1 also uses end-use product ratios. Detailed industry and utilization data were needed to develop these ratios and would be needed to update them, with the level of detail varying depending on the scale of analysis.
   c. **Volume to carbon conversion factors—** NFCMARS-HWP uses harvest quantities from CBM-CFS3 that are already in units of carbon,

therefore volume to carbon conversion factors are applied to harvest data before using NFCMARS-HWP. USFS HWP-C v1 converts primary product volumes to carbon using factors provided in Smith et al. (2006). These factors are based on a cubic foot per unit of primary product (e.g., thousand board feet or thousand square feet), a pounds-per-cubic-foot conversion factor to provide the weight of the product in pounds, a factor for the fraction of the product that is wood fiber, and a factor for the fraction of carbon in wood (0.5). The source information or methods to develop the primary product volume to cubic feet, cubic foot volume to weight conversions, and fraction of product that is wood fiber are not documented in Smith et al. (2006). It is likely that the factors from Smith et al. (2006) reflect the different wood densities for the average national distribution of species used for each primary product from around the time of that publication. These factors may need to be updated periodically to reflect changes in species mix.

d. **End-use product ratios—**USFS HWP-C v1 and its variants use national statistics to develop end-use product ratios (McKeever 2009, McKeever and Howard 2011). These datasets have only been updated through 2009. The USDA Forest Service Forest Products Laboratory is currently updating these datasets. Some variants of ANSE-based models, such as NFCMARS-HWP, estimate HWP carbon storage and emissions based on primary product use and do not require end-use product ratios. However, other ANSE-based models do use end-use product ratios. End-use product ratios in variants of ANSE-based models are location and data dependent; for example, Xie et al. (2021) used forest sector economics data from Forest Economics Advisors, LLC for MitigAna.

e. **Loss factor—**Loss factors apply to HWPs as they are put into end uses, usually because of construction wood waste. USFS HWP-C v1 applies an 8-percent loss factor to solid wood products (Skog and Nicholson 2000), but literature has suggested there is also a loss factor for paper products. This default of 8 percent in the original model can be changed in subsequent variants of USFS HWP-C v1. HWP-C vR also allows users to specify a loss factor for both solid wood and paper products, with a similar modification anticipated in USFS HWP-C v2. ANSE-based models that allocate HWPs to end uses, such as BC-HWPv1, employ the use of loss factors (Franklin Associates 1998) for lumber (7.5 percent) and panels and plywood (4 percent).

f. **Product half-lives—**Both models use product half-lives for HWPs in use to determine the quantity and rate at which products are discarded. NFCMARS-HWP uses IPCC default half-lives for primary products, while USFS HWP-C v1 and its variants use end-use product half-lives from a variety of sources.

g. **Discard disposition ratios—**Both models also use discard disposition ratios to allocate HWPs to various discard pathways. The quantity of HWPs entering landfills and dumps comes from national datasets that changed through time in USFS HWP-C v1. Updated national data (US EPA 2020b) are available for updating discard disposition ratios. Regional or local data may also exist. Data sources to determine the discard fate of HWPs at the end of their useful life are documented in the waste section of the national inventory report (ECCC 2021). USFS HWP-C v1 can accommodate waste incineration for energy production, but no datasets have been available to include in any analyses using this model at this time. Subnational data on waste incineration for energy or fuel may be available from state agencies (e.g., CEC 2022); however, it may be difficult to attribute the portion of waste incineration tied to HWPs from these data. Currently no data exist to employ this pathway, but waste incineration for energy is possible in NFCMARS-HWP.

h. **Discard product half-lives—**Both models use discard product half-lives to determine when and how much carbon is lost as HWPs decay in landfills and dumps. USFS HWP-C v1 relies on IPCC defaults, whereas NFCMARS-HWP includes regionally specific discard product half-lives.

i. **Landfill fixed carbon ratio—**Both models apply discard half-lives to the portion of wood waste subject to decay in landfills as determined by fixed carbon ratios for wood and paper not subject to decay. USFS HWP-C v1 and its variants use older values from Freed and Mintz (2003) and Barlaz (1998), but more recent studies have updated these values (Rüter et al. 2019; Wang and Barlaz 2016; Wang et al. 2011, 2013; Ximenes et al. 2018). Updated values for solid wood and paper summarized in the IPCC's refinement (Rüter et al. 2019) can be used immediately in variants of USFS HWP-C v1; the first author is updating HWP-C vR to accommodate updated values for disaggregated solid wood and paper categories. NFCMARS-HWP currently uses the updated IPCC values (Rüter et al. 2019).

3. **Are all the components of trees removed from the forest ecosystem being tracked?** It is important to understand exactly which tree components are included in HWP estimation models. Treatment of bark, residential/personal fuelwood, logging residues, and other noncommercial biomass can differ between HWP carbon models. Large quantities of these materials may be used for energy production. Although wood burned for energy does not affect the carbon storage associated with HWPs, it does affect the overall climate impact associated with HWPs and can represent substitution benefits. Tree components that are often treated differently or omitted across carbon models include the following:

   a. **Bark—**USFS HWP-C v1 does not estimate carbon from bark utilization because the model follows the IPCC tier 3 production approach and uses commercial harvest volume underbark. Bark removed through commercial harvest is accounted for as an emission from forest ecosystem pools. Methods to include bark utilization are being explored for HWP-C vR, and emissions from using bark for energy are being incorporated into a new workbook tool being prepared in conjunction with the USDA's update to *Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory* (Murray et al., n.d.; Stockmann 2022). Harvest data inputs for CBM-CFS3 forest ecosystem model includes bark, which results in NFCMARS-HWP including it as well. NFCMARS-HWP assumes that bark is used and combusted in the first year of harvest for energy production as hog fuel, but these emissions are not explicitly separate from other fuelwood products (ECCC 2021, Head et al. 2021).

   b. **Firewood or fuelwood—**In the United States, there are limited data about personal fuelwood, or firewood. USDA Forest Service cut-and-sold data include estimates for these product categories, and they are applied to USFS HWP-C v1. However, illegal firewood removal from NFS lands and unreported firewood removal from other public or private lands may result in an underestimation of firewood removals. NFCMARS-HWP provides an option to quantify residential firewood and incorporates these data and parameters from Environment and Climate Change Canada (ECCC 2021, Head et al. 2021).

   c. **Logging residue—**In USFS HWP-C v1, logging residue utilization is accounted for only to the extent that it is reflected in commercial harvest volumes—by definition, this material is typically left in the forest for disposal. In Canada, users can choose to estimate additional utilization of logging residues removed from the forest in the forest ecosystem model (CBM-CFS3) by applying a parameter value for the portion or volume of residue recovered (Smyth et al. 2013). This additional material can then enter ANSE-based HWP carbon models. In the United States, if logging residues potentially available for utilization could be determined, scenarios of additional logging residue utilization could be evaluated through models such as USFS HWP-C v1 with adjustments to input data (e.g., harvest volumes, timber and primary product ratios), or the development of a new pathway for logging residues in the model.

   d. **Other noncommercial material—**Other noncommercial material from precommercial mechanical thinning or fuel reduction activities is also often left in the forest to decay or be disposed of through burning onsite. Because it is noncommercial material and therefore not removed from the site as a product and reflected in harvest volumes, it is excluded in U.S. and Canadian HWP carbon estimation (e.g., USFS HWP-C v1 and NFCMARS-HWP). Exploration of the use of this material in either model could occur either by adding these quantities to harvest volume or roundwood estimates and adjusting timber and primary product ratios according to their possible use or by creating a novel pathway for this material.

4. **Is it possible to estimate landfill CH_4 emissions associated with HWPs?** Landfill CH_4 emissions are attributed to the waste sector rather than the forest sector. For this reason, these emissions have been excluded from U.S. HWP carbon estimation models. However, understanding landfill CH_4 emission associated with discarded HWPs is important to understand their overall climate impact and possible mitigation opportunities. Consequently, NFCMARS-HWP provides an option to calculate landfill CH_4 emissions, including those that are associated with different landfill technologies, such as CH_4 flaring or energy capture. These technologies can reduce the overall CH_4 emissions and may include energy substitution benefits. BC-HWP v1 also includes CH_4 emissions and landfill gas management based on US EPA WARM v-15 (US EPA 2020b). CH_4 capture from landfills is a topic of growing public interest, so producing estimates for both CH_4 emissions and capture are important aspects of future HWP modeling. The first author is currently developing datasets and modifying HWP-C vR to estimate landfill CH_4 emissions and capture in California.

5. **Are all discard pathways for wood represented?** Traditional recycling and cascaded utilization are pathways that can extend the life of HWPs and potentially delay end-of-life emissions. However, traditional recycling is represented in a simplified manner in USFS HWP-C v1 and its variants. Wood and paper are recycled once, retired based on the half-life of 2.6 years for paper, and then are allocated directly to the emissions without energy capture category rather than cycled through the discard pathways (as are regular products in use). Traditional recycling is not currently represented in NFCMARS-HWP; however, ANSE-based models have included this pathway. NFCMARS-HWP and USFS HWP-C v1 and subsequent variants do not allow for reprocessing of material into alternative end uses through cascaded utilization. However, other ANSE-based models, such as MitigAna (Xie et al. 2021) have incorporated pathways for cascaded utilization. Research is underway to improve recycling pathways in USFS HWP-C v1 and its variants. Waste incineration associated with energy production is possible in NFCMARS-HWP and USFS HWP-C v1 and its variants. However, data for waste incineration associated with energy production is limited, and this pathway has not yet been used in either model.

6. **Which IPCC approach is applied and how may that affect resulting HWP carbon estimates?** The IPCC production approach is useful because it directly ties HWPs to a particular land base, but may not accurately reflect the climate impacts from a region's actual wood consumption if most demand is met from imported wood products or large amounts are exported. Although the end-of-life emissions from disposal associated with imported HWPs are not included in the IPCC production approach, carbon stocks and losses associated with imported HWPs are likely accounted for in the originating country or jurisdiction.

   Depending on which IPCC approach is used, estimates may emphasize storage as in the production approach, or emissions as in the simple decay approach. USFS HWP-C v1 uses the IPCC production approach and NFCMARS-HWP uses the simple decay approach. ANSE-based models can accommodate any IPCC approach. The level of detail in outputs from either model are sufficient to summarize results to accommodate either approach, as both exclude imports but include exports. HWP-C vR has been modified to summarize results for both approaches. If there are large export volumes, it may be important to consider utilization, discard, and decay parameters where wood is consumed to more accurately estimate HWP carbon stocks and losses. While NFCMARS-HWP and other ANSE-based models can accommodate different assumptions for exported wood, USFS HWP-C v1 and its variants currently do not.

7. **Can the model be used to evaluate different forest management and wood utilization scenarios?** There is growing interest in understanding the climate impacts and tradeoffs in forest ecosystem and HWP carbon under different management and utilization scenarios. The models function similarly, with some minor differences. NFCMARS-HWP was designed to function as both an inventory of HWP carbon resulting from past and current harvests and a scenario-building model. Users can run the forest ecosystem model (CBM-CFS3) for different management scenarios, which will provide HWP outputs that are then input into NFCMARS-HWP (or other ANSE-based models) to project future HWP carbon storage and emissions.

   USFS HWP-C v1 was designed to use annual historical harvest data to understand current HWP carbon stocks and stock changes. USFS HWP-C v1 has not been used to project future HWP carbon stocks and stock changes from different forest management and wood utilization scenarios. It would be possible to create future projections by using harvest volume outputs from planning projections or forest ecosystem models that run hypothetical management scenarios as inputs for USFS HWP-C v1. However, many forest ecosystem models provide removals in terms of biomass or carbon rather than merchantable volume, so conversions from total removals to merchantable volumes could be challenging. Future wood utilization scenarios could also be accommodated by adjusting timber and primary product ratios as well as other parameters. Users cannot edit most parameters in USFS HWP-C v1, but subsequent variants allow users to change values for several parameters (e.g., end-use product ratios, product half-lives, discard disposition ratios, discard product half-lives), which could accommodate various wood utilization scenarios.

8. **How are global warming implications of using HWP instead of other nonwood building materials or energy sources measured?** The HWP carbon estimation models described in this synthesis do not estimate potential substitution benefits resulting from current or future wood utilization practices. Quantifying these benefits require separate analyses that employ the use of LCAs, resulting substitution factors, and efforts to scale results to national or subnational product mixes (e.g., "Case Study: Embodied Carbon of HWPs in California, 2019"). Research is underway to incorporate cradle-to-gate life-cycle analyses and SFs into a new workbook tool being prepared in conjunction with the USDA's update to *Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry: Methods for Entity-Scale Inventory* (Murray et al., n.d.; Stockmann 2022). However, moving from potential substitution benefits to actual benefits, by addressing whether benefits are additional to what would have occurred otherwise or did not result in leakage impacts on other entities through complementary analysis, is also a challenge. When possible, evaluations of substitution benefits should clearly state the spatial scale and timeframes, as these will change estimated benefits. Benefits may change if the focus is on utilization in a local geography compared to a larger geography or shorter versus longer timeframes.

9. **Are unintended consequences from different forest management and wood utilization scenarios taken into account or measured through a model or accounting approach?** Potential leakage should be an important consideration for changing harvest levels or when evaluating increased usage of wood compared to alternative nonwood building materials. A change in one region's harvest levels might lead to increased forest ecosystem carbon stocks in one location; however, it might lead to additional harvest and reduced forest ecosystem carbon stocks elsewhere. However, reduced demand for wood products could also lead to reduced investment in retaining forest land. Conversely, increased timber prices could lead to greater harvesting and reduced forest carbon stocks, but it could also incentivize investments in forest land that increase overall forest carbon stocks (i.e., spillage). HWP carbon estimation models and LCAs do not have a way to measure leakage or potential for leakage. This type of analysis typically requires the use of forest sector economic models. However, forest dynamics, disturbances, and

effects on growth and yield are often oversimplified in these models. Limited empirical research about leakage and spillage exists, making it difficult to incorporate into current models. Empirical examples of leakage occurring would improve efforts to estimate leakage in model scenarios even if they are highly uncertain.

10. **Are climate change impacts accounted for?** Innumerable and immeasurable variables associated with climate change may influence the future use and availability of timber to supply HWPs because there are variables that might influence forest growing conditions from decreased (or increased) precipitation or higher concentrations of atmospheric CO_2 to higher rates of disturbances throughout the world. The HWP carbon estimation models discussed in this synthesis do not address impacts from climate change. These impacts are reflected in scenarios only to the extent that forest ecosystem modeling incorporates them and couples results with the use of an HWP carbon estimation model.

Conclusion

HWP carbon estimation requires a great level of detail about harvest, utilization, disposal, and decay. This is particularly true for use of HWP carbon models at the subnational level; although these data are developed in some regions, gaps remain. Several datasets currently used as inputs for HWP carbon modeling in the United States or Canada, such as primary product volume to carbon conversions, end-use product ratios, loss factors for products going into end uses, product half-lives, discard ratios, discard half-lives, and landfill fixed carbon ratios, may need to be updated or explored further to refine values or provide more regionally or locally specific values. Improved data about traditional recycling and cascaded utilization, both for paper and solid wood products, as well as ways to more completely incorporate these pathways into HWP carbon models could improve HWP analyses. Improved understanding of quantities and utilization in export markets can help refine U.S. estimates. Lastly, comparisons of estimates from different HWP carbon models using similar inputs and system boundaries would help to better understand the similarities and differences between models and underlying datasets.

Beyond improvements to current HWP carbon models, future research could also consider the impact that economic markets, climate change, and forest management policies have on forest ecosystem and HWP carbon pools. Forest ecosystems are under pressure from land-use change, wildfire, insects, disease, and other consequences from climate change. Evaluating alternative forest management scenarios under changing climate and disturbance regimes may require the use of ecosystem models that can incorporate these factors. Subsequent wood utilization scenarios can be created to address potential HWP carbon storage and emissions associated with the resulting harvest from different scenarios.

Wood products provide renewable building materials and can store carbon for decades or longer in use, as well as in landfills after their useful service life. Wood can be used in place of more carbon-intensive materials or energy sources. Some regions may place greater emphasis on, or have greater ecosystem capital or infrastructure and capacity for, manufacturing long-lived products, while others may be more focused on the use of wood for electrical energy generation. For example, Washington's and Oregon's wood products industries are being augmented by products, such as cross-laminated timber, that have strong growth potential and are being manufactured at newly designed facilities. California is placing emphasis on the use of low-carbon transportation fuels. The future growth of the bioeconomy could also play a role for potential biobased materials. As these products become part of the mainstream economy, their carbon benefits could be significant; however, consideration of shifting GHG emissions tradeoffs will be important to assess their impact.

Forests and wood products are an integral part of climate change solutions, but analyses benefit by considering external influences, such as economics and global atmospheric changes. Current carbon estimation approaches, LCAs, and economic models can increase the understanding of some of the climate benefits associated with HWPs. There is still room to improve forest ecosystem and HWP carbon modeling packages to be more integrated, make space for newer building technologies, and help facilitate future forest management and HWP production across the globe.

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U.S. Standard Equivalents

When you know:	Multiply by:	To find:
Hectares (ha)	2.47	Acres
Cubic meters (m^3)	35.3	Cubic feet
Kilograms (kg)	0.0011	Tons
Teragrams (Tg)	0.000001102	Tons
Tonnes (t) or megagrams (Mg)	1.102	Tons

References

Abt, R.C.; Cubbage, F.W.; Abt, K.L. 2009. Projecting southern timber supply for multiple products by subregion. Forest Products Journal. 59(7/8): 7–16.

American Forests. 2021. Effects of forest management and wood utilization on carbon sequestration and storage in California. California State Board of Forestry and Fire Protection, virtual public meeting, March 22. https://bof.fire.ca.gov/media/eaqlr4vx/full-16-presentation_-american-forests_effects-of-forest-managment-and-wood-utilization-on-carbon-storage-and-sequestration.pdf.

Anderson, N.; Young, J.; Stockmann, K.; Skog, K.; Healey, S.; Loeffler, D.; Jones, J.G.; Morrison, J. 2013. Regional and forest-level estimates of carbon stored in harvested wood products from the United States Forest Service Northern Region, 1906–2010. Gen. Tech. Rep. RMRS-GTR-311. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 114 p. https://doi.org/10.2737/RMRS-GTR-311.

Athena Sustainable Materials Institute. 2019. User manual and transparency document: Impact Estimator for Buildings. Version 5. Ottawa, ON: Athena Sustainable Materials Institute. https://calculatelca.com/wp-content/uploads/2019/05/IE4B_v5.4_User_Guide_May_2019.pdf.

Barlaz, M.A. 1998. Carbon storage during biodegradation of municipal solid waste components in laboratory-scale landfills. Global Biogeochemical Cycles. 12(2): 373–380. https://doi.org/10.1029/98GB00350.

Barlaz, M.A. 2022. Personal communication. Professor, North Carolina State University, Fitts-Woolard Hall, 915 Partners Way, Raleigh, NC 27606. barlaz@ncsu.edu.

Bates, L.; Jones, B.; Marland, E.; Marland, G.; Ruseva, T.; Kowalczyk, T.; Hoyle, J. 2017. Accounting for harvested wood products in a forest offset program: lessons from California. Journal of Forest Economics. 27(1): 50–59. https://doi.org/10.1016/j.jfe.2017.02.004.

Beach, R.H.; Adams, D.; Alig, R.; Baker, J.; Latta, G.S.; White, E. 2010. Model documentation for the forest and agricultural sector optimization model with greenhouse gases (FASOMGHG). Research Triangle Park, NC: RTI International. 78 p.

Benson, R.E.; Johnston, C.M. 1976. Logging residues under different stand and harvesting conditions, Rocky Mountains. Res. Pap. INT-RP-181. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 15 p.

Bergman, R.D.; Bowe, S.A. 2008. Environmental impact of producing hardwood lumber using life-cycle inventory. Wood and Fiber Science. 40(3): 448–458.

Bergman, R.; Bowe S.A. 2010. Environmental impact of manufacturing softwood lumber determined by life-cycle inventory. Wood and Fiber Science. 42(Suppl. 1): 67–78.

Bergman, R.; Bowe, S. 2012. Life-cycle inventory of manufacturing hardwood lumber in the Southeastern United States. Wood and Fiber Science. 44(1): 71–84.

Bergman, R.; Falk, R.; Salazar, J.; Gu, H.; Napier, T.; Meil, J.M. 2013. Life-cycle energy and GHG emissions for new and recovered softwood framing lumber and hardwood flooring considering end-of-life scenarios. Res. Pap. FPL-RP-672. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 33 p. https://doi.org/10.2737/FPL-RP-672.

Bergman, R.; Oneil, E.; Puettmann, M.; Eastin, I.; Ganguly, I. 2014a. Updating of U.S. wood product life-cycle assessment data for environmental product declarations. In: Salenikovich, A., ed. Proceedings, 2014 World Conference on Timber Engineering. Quebec City, QC: World Conference on Timber Engineering. 8 p.

Bergman, R.; Puettmann, M.; Taylor, A.; Skog, K.E. 2014b. The carbon impacts of wood products. Forest Products Journal. 64(7/8): 220–231. https://doi.org/10.13073/FPJ-D-14-00047.

Bergman, R.; Taylor, A. 2011. Environmental product declarations of wood products: an application of life cycle information about forest products. Forest Products Journal. 61(3): 192–201. https://doi.org/10.13073/0015-7473-61.3.192.

Birdsey, R.A.; Lewis, G.M. 2003. Carbon in U.S. forests and wood products, 1987–1997: state-by-state estimates. Gen. Tech. Rep. NE-310. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station. 42 p. https://doi.org/10.2737/NE-GTR-310.

Boyd C.W.; Koch, P.; McKean, H.B.; Morschauser, C.R.; Preston, S.B.; Wangaard. F.F. 1976. Wood for structural and architectural purposes. Wood and Fiber. 8(1): 3–72. https://www.fs.usda.gov/treesearch/pubs/7976.

Brander, M.; Tipper, R.; Hutchison, C.; Davis, G. 2009. Consequential and attributional approaches to LCA: a guide to policy makers with specific reference to greenhouse gas LCA of biofuels. Tech. Pap. TP090403A. London, United Kingdom: Econometrica Press. 14 p.

Bruckner, T.; Bashmakov, I.A.; Mulugetta, Y.; Chum, H.; de la Vega Navarro, A.; Edmonds, J.; Faaij, A.; Fungtammasan, B.; Garg, A.; Hertwich, E.; Honnery, D.; Infield, D.; Kainuma, M.; Khennas, S.; Kim, S.; Nimir, H.B.; Riahi, K.; Strachan, N.; Wiser, R.; Zhang, X. 2014: Energy systems. In: Edenhofer, O.; Pichs-Madruga, R.; Sokona, Y.; Farahani, E.; Kadner, S.; Seyboth, K.; Adler, A.; Baum, I.; Brunner, S.; Eickemeier, P.; Kriemann, B.; Savolainen, J.; Schlömer, S.; von Stechow, C.; Zwickel, T.; Minx, J.C., eds. Climate change 2014: mitigation of climate change. Contribution of working group III to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York: Cambridge University Press: 511–597. Chapter 7.

Brunet-Navarro, P.; Jochheim, H.; Muys, B. 2016. Modelling carbon stocks and fluxes in the wood product sector: a comparative review. Global Change Biology. 22(7): 2555–2569. https://doi.org/10.1111/gcb.13235.

Building Transparency; KT Innovations; thinkstep; Autodesk. ©2023. Tally®. https://choosetally.com. (January 31, 2023).

Buongiorno, J.; Zhu, S. 2015. Technical change in forest sector models: the global forest products model approach. Scandinavian Journal of Forest Research. 30(1): 30–48. https://doi.org/10.1080/02827581.2014.957238.

Butler, E.; Stockmann, K.; Anderson, N.; Skog, K.; Healey, S.; Loeffler, D.; Jones, J.G.; Morrison, J.; Young, J. 2014. Estimates of carbon stored in harvested wood products from United States Forest Service Pacific Northwest Region, 1909–2012. 28 p. Unpublished report. On file with: Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, 800 East Beckwith Avenue, Missoula, MT 59801-5801. https://www.fs.usda.gov/rm/pubs_other/rmrs_2014_butler_e001.pdf.

Cabiyo, B.; Fried, J.S.; Collins, B.M.; Stewart, W.; Wong, J.; Sanchez, D.L. 2021. Innovative wood use can enable carbon-beneficial forest management in California. Proceedings of the National Academy of Sciences of the United States of America. 118(49): e2019073118. https://doi.org/10.1073/pnas.2019073118.

California Air Resources Board [CARB]. 2018a. An inventory of ecosystem carbon in California's natural and working lands. 2018 ed. Sacramento, CA: California Air Resources Board. 63 p. https://ww3.arb.ca.gov/cc/inventory/pubs/nwl_inventory.pdf.

California Air Resources Board [CARB]. 2018b. Technical support document for the natural and working lands inventory. Sacramento, CA: California Air Resources Board, Air Quality Planning and Science Division. 95 p. https://ww3.arb.ca.gov/cc/inventory/pubs/nwl_inventory_technical.pdf.

California Department of Resources Recycling and Recovery [CalRecycle]. 2018. 2018 disposal-facility-based characterization of solid waste in California. DRRR-2020-1666. Sacramento, CA: California Environmental Protection Agency. 176 p. https://www2.calrecycle.ca.gov/Publications/Details/1666. (January 16, 2023).

California Energy Commission [CEC]. 2022. Municipal solid waste power plants. https://www.energy.ca.gov/data-reports/california-power-generation-and-power-sources/biomass/municipal-solid-waste-power. (January 16, 2023).

Canadian Council of Forest Ministers [CCFM]. 2022. National forestry database, harvest. http://nfdp.ccfm.org/en/data/harvest.php. (January 16, 2023).

Christensen, G.A.; Gray, A.N.; Kuegler, O.; Tase, N.A.; Rosenberg. M. 2021. AB 1504 California forest ecosystem and harvested wood product carbon inventory: 2019 reporting period data update. Sacramento, CA: California Department of Forestry and Fire Protection and California Board of Forestry and Fire Protection; U.S. Department of Agriculture Forest Service agreement 18-CO-11052021-214, and California Department of Forestry and Fire Protection agreement 8CA04056. 447 p. https://bof.fire.ca.gov/media/beddx5bp/6-final_forest_ecosys_hwp_c_2019_feb2021_all_ada.pdf.

Christensen, G.A.; Gray, A.N.; Kuegler, O.; Tase, N.A.; Rosenberg. M.; Loeffler, D.; Anderson, N.; Stockmann, K.; Morgan, T.A. 2019a. AB 1504 California forest ecosystem and harvested wood product carbon inventory: 2017 reporting period. [Missoula, MT: University of Montana]; final report; U.S. Department of Agriculture Forest Service agreements 18-CO-11052021-214 and 17-CO-11261979-086, California Department of Forestry and Fire Protection agreement 8CA04056 and 8CA03714. 539 p. https://bof.fire.ca.gov/media/8026/4-final_1504_forest_ecosys_hwp_c_2017_13feb19_full.pdf.

Christensen, G.A.; Gray, A.N.; Kuegler, O.; Yost, A.C. 2019b. Oregon forest ecosystem carbon inventory: 2001–2016. Final report. Salem, OR: Oregon Department of Forestry. Agreement 18-C-CO-11261979-019. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 1220 SW 3rd Avenue, Suite 1410, Portland, OR 97204. 347 p. https://www.oregon.gov/odf/forestbenefits/Documents/Forest Carbon Study/OR-Forest-Ecosystem-Carbon-2001-2016-Report-FINAL.pdf.

Christensen, G.; Tase, N.; Gray, A.; Kuegler, O.; Drummond, J.; Dymond, C.; Kurz, W.; Dillon, T.; Morgan, T.; Evans, S.; Scott, S.; Henly, R. [N.d.]. Pacific coast region temperate forest carbon dynamics: a regional forest and harvested wood products carbon assessment of British Columbia, California, Oregon, and Washington, 2001–2019. Manuscript in preparation. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 1220 SW 3rd Avenue, Suite 1440, Portland, OR 97204.

Consortium for Research on Renewable Industrial Materials [CORRIM]. 2005. Documenting the environmental performance of wood building materials. Wood and Fiber Science. 37 (Spec. issue). 155 p. https://corrim.org/wp-content/uploads/2017/12/Documenting-the-Environmental-Performance-of-Wood-Building-Materials.pdf.

Consortium for Research on Renewable Industrial Materials [CORRIM]. 2010. Introduction to special issue: extending the findings on the environmental performance of wood building materials. Wood Fiber Science. 42 (Spec. issue). 164 p. https://wfs.swst.org/index.php/wfs/article/view/1986/1986. (August 21, 2020).

Daigneault, A.; Favero, A. 2021. Global forest management, carbon sequestration and bioenergy supply under alternative shared socioeconomic pathways. Land Use Policy. 103: 105302. https://doi.org/10.1016/j.landusepol.2021.105302.

Daigneault, A.; Sohngen, B.; Sedjo, R. 2012. Economic approach to access the forest carbon implications of biomass energy. Environmental Science and Technology. 46(11): 5664–5671. https://doi.org/10.1021/es2030142.

De la Cruz, F.B.; Barlaz, M.A. 2010. Estimation of waste component-specific landfill decay rates using laboratory-scale decomposition data. Environmental Science and Technology. 44(12): 4722–4728. https://doi.org/10.1021/es100240r.

Denny, A. 2021. Personal communication. Local energy and environment program lead, U.S. Environmental Protection Agency, 1200 Pennsylvania Avenue NW, Washington, DC 20004. denny.andrea@epa.gov.

Dillon, T; Morgan, T.A. 2023. Pacific coast temperate forest regional timber product flow analysis. Gen. Tech. Rep. PNW-GTR-1017. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. https://doi.org/10.2737/PNW-GTR-1017.

Domke, G. 2021. Personal communication. Team leader and research forester, U.S. Department of Agriculture, Forest Service, Northern Research Station, St. Paul, MN 55108-1034. grant.m.domke@usda.gov.

Domke, G.M.; Walters, B.F.; Nowak, D.J.; Smith, J.E.; Nichols, M.C.; Ogle, S.M.; Coulston, J.W.; Wirth, T.C. 2021. Greenhouse gas emissions and removals from forest land, woodlands, and urban trees in the United States, 1990–2019. Res. Update FS–307. Madison, WI: U.S. Department of Agriculture, Forest Service, Northern Research Station. 5 p. [plus 2 appendices]. https://doi.org/10.2737/FS-RU-307.

Donahue, T.; Morgan, T.A.; Dillon, T. 2021. Oregon sawmill energy consumption and associated emissions, 2017. Missoula, MT: University of Montana, Bureau of Business and Economic Research. 25 p. https://www.oregon.gov/odf/forestbenefits/Documents/or-sawmill-energy-consumption-associated-emissions-2017.pdf.

Dugan, A.J.; Birdsey, R.; Mascorro, V.S.; Magnan, M.; Smyth, C.E.; Olguin, M.; Kurz, W.A. 2018. A systems approach to assess climate change mitigation options in landscapes of the United States forest sector. Carbon Balance and Management. 13(1): 1–14. https://doi.org/10.1186/s13021-018-0100-x.

Dugan, A.J.; Lichstein, J.W.; Steele, A.; Metsaranta, J.M.; Bick, S.; Hollinger, D.Y. 2021. Opportunities for forest sector emissions reductions: a state-level analysis. Ecological Applications. 31(5): e02327. https://doi.org/10.1002/eap.2327.

Dymond, C.C. 2012. Forest carbon in North America: annual storage and emissions from British Columbia's harvest, 1965–2065. Carbon Balance and Management. 7(1): 1–20. https://doi.org/10.1186/1750-0680-7-8.

Eggers, T. 2002. The impacts of manufacturing and utilisation of wood products on the European carbon budget. Internal Rep. 9. Joensuu, Finland: European Forest Institute. 89 p. https://efi.int/sites/default/files/files/publication-bank/2018/ir_09.pdf.

Eleazer, W.E.; Odel, W.S., III; Wang, Y.S.; Barlaz, M.A. 1997. Biodegradability of municipal solid waste components in laboratory-scale landfills. Environmental Science and Technology. 31(3): 911–917. https://doi.org/10.1021/es9606788.

Environment and Climate Change Canada [ECCC]. 2021. National inventory report 1990–2019: greenhouse gas sources and sinks in Canada. Ottawa, Ontario. 207 p. https://publications.gc.ca/site/eng/9.506002/publication.html. (January 16, 2023).

Forster, E.; Healey, J.; Dymond, C.; Styles, D. 2021. Commercial afforestation can deliver effective climate change mitigation under multiple decarbonisation pathways. Nature Communications. 12(1): 3831. https://doi.org/10.1038/s41467-021-24084-x.

Franklin Associates. 1998. Characterization of building-related construction and demolition debris in the United States. EPA530-R-98-010. [Prairie Village, KS: Franklin Associates]; Contract 68-W4-0006, work assignment R11026. Submitted to: U.S. Environmental Protection Agency, Municipal and Industrial Solid Waste Division, Office of Solid Waste. 94 p.

Freed, R. 2021. Personal communication. Former senior vice president, ICF International, Inc., 1725 Eye St. NW, Suite 1000, Washington, DC 20006. rfreed@icfconsulting.com.

Freed, R.; Mintz, C. 2003. Open-dump and landfill timeline spreadsheet. Unpublished report. On file with: ICF International, Inc., 1725 Eye Street NW, Suite 1000, Washington, DC 20006.

Freed, R.; Skog, K.; Mintz, C.; Glick, N. 2004. Carbon storage due to disposal of biogenic materials in U.S. landfills. In: Proceedings of the third annual conference on carbon sequestration. Alexandria, VA: U.S. Department of Energy. 15 p.

Ganguly, I.; Pierobon, F.; Sonne Hall, E. 2020. Global warming mitigating role of wood products from Washington state's private forests. Forests. 11(2): 194. https://doi.org/10.3390/f11020194.

Garcia, R.; Alvarenga, R.A.F; Huysveld, S.; Dewulf, J.; Allacker, K. 2020. Accounting for biogenic carbon and end-of-life allocation in life cycle assessment of multi-output wood cascade systems. Journal of Cleaner Production. 275: 122795. https://doi.org/10.1016/j.jclepro.2020.122795.

Gaudreault, C.; Samson, R.; Stuart, P.R. 2010. Energy decision making in a pulp and paper mill: selection of LCA system boundary. The International Journal of Life Cycle Assessment. 15: 198–211. https://doi.org/10.1007/s11367-009-0125-1.

Government of Canada. 2023. Abstract Network Simulation Engine (ANSE). https://natural-resources.canada.ca/climate-change/climate-change-impacts-forests/carbon-accounting/forest-carbon-accounting-tools/abstract-network-simulation-engine-anse/24901. (March 23, 2023).

Groom, J.D.; Tase, N.A. 2022. Harvested wood products carbon model, version R documentation. [Place of publication unknown: Publisher unknown]; California Department of Forestry and Fire Protection agreement 3540 0000277546; Oregon Department of Forestry agreement ODF-1136A-20. https://jeremygroom.github.io/HWP-vR-Documentation. (January 16, 2023).

Gustavsson, L.; Sathre, R. 2006. Variability in energy and carbon dioxide balances of wood and concrete building materials. Building and Environment. 41(7): 940–951. https://doi.org/10.1016/j.buildenv.2005.04.008.

Harmon, M.E. 2019. Have product substitution carbon benefits been overestimated? A sensitivity analysis of key assumptions. Environmental Research Letters. 14(6): 065008. https://doi.org/10.1088/1748-9326/ab1e95.

Head, M.; Magnan, M.; Kurz, W.A. Levasseur, A.; Beauregard, R.; Margni, M. 2021. Temporally-differentiated biogenic carbon accounting of wood building product life cycles. SN Applied Sciences. 3: 1–17. https://doi.org/10.1007/s42452-020-03979-2.

Heath, L.S.; Maltby, V.; Miner, R.; Skog, K.E.; Smith, J.E.; Unwin, J.; Upton, B. 2010. Greenhouse gas and carbon profile of the U.S. forest products industry value chain. Environmental Science and Technology. 44(10): 3999–4005. https://doi.org/10.1021/es902723x.

Höglmeier, K.; Steubing, B.; Weber-Blaschke G.; Richter, K. 2015. LCA-based optimization of wood utilization under special consideration of a cascading use of wood. Journal of Environmental Management. 152: 158–170. https://doi.org/10.1016/j.jenvman.2015.01.018.

Höglmeier, K.; Weber-Blaschke, G.; Richter, K. 2014. Utilization of recovered wood in cascades versus utilization of primary wood—a comparison with life cycle assessment using system expansion. International Journal of Life Cycle Assessment. 19: 1755–1766. https://doi.org/10.1007/s11367-014-0774-6.

Hoover, C.M.; Beukema, S.J.; Robinson, D.C.E.; Kellock, K.M.; Abraham, D.A. 2014. PRESTO: online calculation of carbon in harvested wood products. Gen. Tech. Rep. GTR-NRS-141. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 20 p. https://doi.org/10.2737/NRS-GTR-141.

Hoover, C.M.; Rebain, S.A. 2011. Forest carbon estimation using the Forest Vegetation Simulator: seven things you need to know. Gen. Tech. Rep. NRS-77. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 16 p. https://doi.org/10.2737/NRS-GTR-77.

Howard, C.; Dymond, C.C.; Griess, V.C; Tolkien-Spurr, D.; van Kooten, G.C. 2021. Wood product carbon substitution benefits: a critical review of assumptions. Carbon Balance and Management. 16(9): 1–11. https://doi.org/10.1186/s13021-021-00171-w.

Ince, P.J.; Kramp, A.D.; Skog, K.E.; Spelter, H.N.; Wear, D.N. 2011. U.S. forest products module: a technical document supporting the Forest Service 2010 RPA assessment. Res. Pap. FPL-RP-662. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 61 p. https://doi.org/10.2737/FPL-RP-662.

Ince, P.J.; Kramp, A.D.; Skog, K.E.; Spelter, H.N.; Wear, D.N. 2011. U.S. forest products module: a technical document supporting the Forest Service 2010 RPA Assessment. Res. Pap. FPL-RP-662. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 61 p. https://doi.org/10.2737/FPL-RP-662.

Intergovernmental Panel on Climate Change [IPCC]. 1997. Revised 1996 IPCC guidelines for national greenhouse inventories. Houghton, J.T.; Meira Filho, L.G.; Lim, B.; Tréanton, K.; Mamaty, I.; Bonduki, Y.; Griggs, D.J.; Callander, B.A., eds. IPCC/Organisation for Economic Co-operation and Development/International Energy Agency. Paris, France: Task Force on National Greenhouse Gas Inventories. 3 vol. https://www.ipcc-nggip.iges.or.jp/public/gl/invs1.html. (July 13, 2023).

Intergovernmental Panel on Climate Change [IPCC]. 2000. Good practice guidance and uncertainty management in national greenhouse gas inventories. Penman, J.; Kruger, D.; Galbally, I.; Hiraishi, T.; Nyenzi, B.; Emmanul, S.; Buendia, L.; Hoppaus, R.; Martinsen, T.; Meijer, J.; Miwa, K.; Tanabe, K., eds. IPCC/Organisation for Economic Co-operation and Development/International Energy Agency/Global Environmental Strategies. Hayama, Japan: Institute for Global Environmental Strategies. https://www.ipcc.ch/publication/good-practice-guidance-and-uncertainty-management-in-national-greenhouse-gas-inventories. (July 13, 2023).

Intergovernmental Panel on Climate Change [IPCC]. 2003. Good practice guidance for land use, land-use change and forestry. Penman, J.; Gytarsky, M.; Hiraishi, T.; Krug, T.; Kruger, D.; Pipatti, R.; Buendia, L.; Miwa, K.; Ngara, T.; Tanabe, K.; Wagner, F., eds. IPCC/Organisation for Economic Co-operation and Development/International Energy Agency/Global Environmental Strategies. Hayama, Japan: Institute for Global Environmental Strategies. https://www.ipcc-nggip.iges.or.jp/public/gpglulucf/gpglulucf_files/GPG_LULUCF_FULL.pdf.

Intergovernmental Panel on Climate Change [IPCC]. 2007. Climate change 2007: working group I: the physical science basis. Solomon, S.; Qin, D.; Manning, M.; Chen, Z.; Marquis, M.; Averyt, K.B.; Tignor, M.; Miller, H.L., eds. Contribution of working group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. https://archive.ipcc.ch/publications_and_data/ar4/wg1/en/ch2s2-10-2.html. (January 16, 2023).

Intergovernmental Panel on Climate Change [IPCC]. 2014. 2013 revised supplementary methods and good practice guidance arising from the Kyoto Protocol. Hiraishi, T.; Krug, T.; Tanabe, K.; Srivastava, N.; Baasansuren, J.; Fukuda, M.; Troxler, T.G., eds. Geneva, Switzerland: Task Force on National Greenhouse Gas Inventories. 268 p. https://www.ipcc.ch/site/assets/uploads/2018/03/KP_Supplement_Entire_Report.pdf.

International Organization for Standardization [ISO]. 2006a. Environmental management—life-cycle assessment—principles and framework. ISO 14040:2006. Geneva, Switzerland: International Organization for Standardization Technical Committee. 20 p. https://www.iso.org/standard/37456.html. (July 13, 2023).

International Organization for Standardization [ISO]. 2006b. Environmental management—life-cycle assessment—requirements and guidelines. ISO 14044:2006. Geneva, Switzerland: International Organization for Standardization Technical Committee. 46 p. https://www.iso.org/standard/38498.html. (July 13, 2023).

International Organization for Standardization [ISO]. 2018. Greenhouse gases—carbon footprint of products—requirements and guidelines for quantification and communication. ISO/TS 14067:2018. Geneva, Switzerland: International Organization for Standardization Technical Committee. 58 p. https://www.iso.org/standard/71206.html. (July 13, 2023).

Jarre, M.; Petit-Boix, A.; Priefer, C.; Meyer, R.; Leipold, S. 2020. Transforming the biobased sector towards a circular economy—What can we learn from wood cascading? Forest Policy and Economics. 110: 101872. https://doi.org/10.1016/j.forpol.2019.01.017.

Jasinevičius, G.; Lindner, M.; Cienciala, E.; Tykkyläinen, M. 2018. Carbon accounting in harvested wood products: assessment using material flow analysis resulting in larger pools compared to the IPCC default method. Journal of Industrial Ecology. 22(1): 121–131. https://doi.org/10.1111/jiec.12538.

Jenkins, J.C.; Chojnacky, D.C.; Heath, L.S.; Birdsey, R.A. 2003. National-scale biomass estimators for United States tree species. Forest Science. 49(1): 12–35.

Jeong, S. 2016. Verification of methodologies and estimation of IPCC model parameters for solid waste landfills. Seoul: Seoul National University. 141 p. Ph.D. dissertation.

Johnston, C.M.T.; Guo, J.; Prestemon, J.P. 2021. The Forest Resource Outlook Model (FOROM): a technical document supporting the Forest Service 2020 RPA Assessment. Gen. Tech. Rep. SRS-254. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 19 p. https://doi.org/10.2737/SRS-GTR-254.

Johnston, C.M.T.; Radeloff, V.C. 2019. Global mitigation potential of carbon stored in harvested wood products. Proceedings of the National Academy of Sciences. 116(29): 14526–14531. https://doi.org/10.1073/pnas.1904231116.

Johnston, C.M.T.; van Kooten, G.C. 2016. Global trade impacts of increasing Europe's bioenergy demand. Journal of Forest Economics. 23: 27–44. https://doi.org/10.1016/j.jfe.2015.11.001.

Jones, G.; Loeffler, D.; Calkin, D.; Chung, W. 2010. Forest treatment residues for thermal energy compared with disposal by onsite burning: emissions and energy return. Biomass and Bioenergy. 34(5): 737–746. https://doi.org/10.1016/j.biombioe.2010.01.016.

Khatri, P.; Nepal, P.; Sahoo, K.; Bergman, R.; Nicholls, D.; Gray, A. 2023. California's harvested wood products: a time-dependent assessment of life cycle greenhouse gas emissions. Science of the Total Environment. 886: 163918. https://doi.org/10.1016/j.scitotenv.2023.163918.

Kizha, A.R.; Han, H.S. 2016. Predicting aboveground biomass in second growth coast redwood: comparing localized with generic allometric models. Forests. 7(5): 1–13. https://doi.org/10.3390/f7050096.

Kurz, W.A.; Dymond, C.C.; White, T.M.; Stinson, G.; Shaw, C.H.; Rampley, G.J.; Smyth, C.; Simpson, B.N.; Neilson, E.T.; Trofymow, J.A.; Metsaranta, J. 2009. CBM-CFS3: a model of carbon-dynamics in forestry and land-use change implementing IPCC standards. Ecological Modelling. 220(4): 480–504. https://doi.org/10.1016/j.ecolmodel.2008.10.018.

Lamers, P.; Junginger, M.; Dymond, C.C.; Faaij, A. 2013. Damaged forests provide an opportunity to mitigate climate change. GCB Bioenergy. 6(1): 44–60. https://doi.org/10.1111/gcbb.12055.

Latta, G.S.; Adams, D.M.; Bell, K.P.; Kline, J.D. 2016. Evaluating land-use and private forest management responses to a potential forest carbon offset sales program in western Oregon (USA). Forest Policy and Economics. 65: 1–8. https://doi.org/10.1016/j.forpol.2016.01.004.

Leskinen, P.; Cardellini, G.; González-García, S.; Hurmekoski, E.; Sathre, R.; Seppälä, J.; Smyth, C.; Stern, T.; Verkerk, P.J. 2018. Substitution effects of wood-based products in climate change mitigation. From Science to Policy 7. Joensuu, Finland: European Forest Institute. 27 p. https://doi.org/10.36333/fs07.

Lippke, B.; Wilson, J.; Perez-Garcia, J.; Bowyer, J.; Meil, J. 2004. CORRIM: Life-cycle environmental performance of renewable building materials. Forest Products Journal. 54(6): 8–19.

Lippke, B.; Oneil, E.; Harrison, R.; Skog, K.; Gustavsson, L.; Sathre, R. 2011. Life cycle impacts of forest management and wood utilization on carbon mitigation: knowns and unknowns. Carbon Management. 2(3): 303–333. https://doi.org/10.4155/cmt.11.24.

Loeffler, D.; Anderson, N.; Stockmann, K.; Morgan, T.A.; Tase, N.A. 2019. Harvested wood products. In: Christensen, G.A.; Gray, A.N.; Kuegler, O.; Tase, N.A.; Rosenberg, M.; Loeffler, D.; Anderson, N.; Stockmann, K.; Morgan, T.A. AB 1504 California forest ecosystem and harvested wood product carbon inventory: 2017 reporting period. Final report. [Missoula, MT: University of Montana]. U.S. Department of Agriculture Forest Service agreements 18-CO-11052021-214 and 17-CO-11261979-086, California Department of Forestry and Fire Protection agreement 8CA04056 and 8CA03714. 539 p. Chapters 5 and 6. https://bof.fire.ca.gov/media/8026/4-final_1504_forest_ecosys_hwp_c_2017_13feb19_full.pdf.

Loeffler, D.; Anderson, N.; Stockmann, K.; Skog, K.; Healey, S.; Jones, J.G.; Morrison, J.; Young, J. 2014. Estimates of carbon stored in harvested wood products from United States Forest Service Southern Region, 1911–2012. Unpublished report. Missoula, MT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 27 p. https://www.fs.usda.gov/treesearch/pubs/46645. (January 16, 2023).

Magnani, F.; Dewar, R.C.; Borghetti, M. 2009. Leakage and spillover effects of forest management on carbon storage: theoretical insights from a simple model. Tellus B. Chemical and Physical Meteorology.61(2): 385–393. https://doi.org/10.1111/j.1600-0889.2008.00386.x.

Malmsheimer R.W.; Bowyer, J.L.; Fried, S.J.; Gee, E.; Izlar, R.L.; Miner, R.A.; Munn, I.A.; Oneil, E.; Stewart, W.C. 2011. Managing forests because carbon matters: integrating energy, products, and land management policy. Journal of Forestry. 109(7): S7–S51.

Manley, B.; Evison, D.C. 2017. Quantifying the carbon in harvested wood products from logs exported from New Zealand. New Zealand Journal of Forestry. 62(3): 36–44.

Marcille, K.C.; Morgan, T.A.; McIver, C.P.; Christensen, G.A. 2020. California's forest products industry and timber harvest, 2016. Gen. Tech. Rep. PNW-GTR-994. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 58 p.

Maritato, B.; Plaggmier, J.; Quealy, J.; Anderson, M.; Frisch, S.; Hao, S.; Sanford, J.; Stromberg, Z.; Timmer, K.; Alvey, E.; Buchholz, T.; Rudeen, C.; Schmidt., D.; Bui, J. 2021. Lake Tahoe greenhouse gas inventory update. Final report. Stateline, NV: Tahoe Regional Planning Agency. 106 p.

Marland, E.; Marland, G. 2003. The treatment of long-lived, carbon-containing products in inventories of carbon dioxide to the atmosphere. Environmental Science and Policy. 6(2): 139–152. https://doi.org/10.1016/S1462-9011(03)00003-0.

Marland, E.S.; Stellar, K.; Marland, G. 2010. A distributed approach to accounting for carbon in wood products. Mitigation and Adaptation Strategies for Global Change. 15: 71–91. https://doi.org/10.1007/s11027-009-9205-6.

McKechnie, J.; Colombo, S.; Chen, J.; Mabee, W.; MacLean, H.L. 2011. Forest bioenergy or forest carbon? Assessing trade-offs in greenhouse gas mitigation with wood-based fuels. Environmental Science and Technology. 45(2): 789–795. https://doi.org/10.1021/es1024004.

McKeever, D.B. 2002. Domestic market activity in solid wood products in the United States, 1950–1998. Gen. Tech. Rep. PNW-GTR-524. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 76 p. https://doi.org/10.2737/PNW-GTR-524.

McKeever, D.B. 2009. Estimated annual timber products consumption in major end uses in the United States, 1950–2006. Gen. Tech. Rep. FPL-GTR-181. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 47 p. https://doi.org/10.2737/FPL-GTR-181.

McKeever, D.B.; Howard, J.L. 2011. Solid wood timber products consumption in major end uses in the United States, 1950–2009: a technical document supporting the Forest Service 2010 RPA assessment. Gen. Tech. Rep. FPL-GTR-199. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 39 p. https://doi.org/10.2737/FPL-GTR-199.

McKinley, D.C.; Ryan, M.G.; Birdsey, R.A.; Giardina, C.P.; Harmon, M.E.; Heath, L.S.; Houghton, R.A.; Jackson, R.B.; Morrison, J.F.; Murray, B.C.; Pataki, D.E.; Skog, K.E. 2011. A synthesis of current knowledge on forests and carbon storage in the United States. Ecological Applications. 21(6): 1902–1924. https://doi.org/10.1890/10-0697.1.

Melosi, M.V. 1982. Garbage in the cities: refuse reform and the environment: 1880–1980. College Station, TX: Texas A&M University Press. 268 p.

Melosi, M.V. 2000. The sanitary city: urban infrastructure in America from Colonial times to the present. Baltimore, MD: Johns Hopkins University Press. 600 p.

Micales, J.A.; Skog, K.E. 1997. The decomposition of forest products in landfills. International Biodeterioration and Biodegradation. 39(2–3): 145–158. https://doi.org/10.1016/S0964-8305(97)83389-6.

Miner, R.A.; Abt, R.C.; Bowyer, J.L.; Buford, M.A.; Malmsheimer, R.W.; O'Laughlin, J.; Oneil, E.E.; Sedjo, R.A.; Skog, K.E. 2014. Forest carbon accounting considerations in U.S. bioenergy policy. Journal of Forestry. 112(6): 591–606.

Morgan, T.A.; Brandt, J.P.; Songster, K.E.; Keegan, C.E.; Christensen, G.A. 2012. California's forest products industry and timber harvest, 2006. Gen. Tech. Rep. PNW-GTR-866. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 48 p. https://doi.org/10.2737/PNW-GTR-866.

Morgan, T.A.; Donahue, T. 2019. Energy use and emissions by the California sawmill sector, 2016. Final report. Missoula, MT: University of Montana, Bureau of Business and Economic Research. 24 p. https://bof.fire.ca.gov/media/9689/ca-sawmill-energy-report_final_20_9-16-19_v1_ada.pdf.

Morgan, T.A.; Donahue, T.S.; Dillon, T.; Yost, A.; Groom, J. 2021. Oregon harvested wood products carbon inventory, 1906–2018. Missoula, MT: University of Montana, Bureau of Business and Economic Research. 56 p. https://www.oregon.gov/odf/forestbenefits/Documents/oregon-harvested-wood-products-carbon-inventory-report.pdf.

Morgan, T.A.; Spoelma, T.P. 2008. California logging utilization: 2004. Western Journal of Applied Forestry. 23(1): 12–18. https://doi.org/10.1093/wjaf/23.1.12.

Murray, B.C.; McCarl, B.A.; Lee, H.C. 2004. Estimating leakage from forest carbon sequestration programs. Land Economics. 80(1): 109–124. https://doi.org/10.2307/3147147.

Murray, L.T.; Woodall, C.; Lister, A.; Stockmann, K.; Gu, H.; Urbanski, S.; Riley, K.; Greenfield E., et al. [N.d.]. Quantifying greenhouse gas sources and sinks in managed forest systems. In: Hansen, W.; Itle, C., eds. Quantifying greenhouse gas fluxes in agriculture and forestry: methods for entity-scale inventory. Manuscript in preparation. On file with: U.S. Department of Agriculture Forest Service, Washington Office, Inventory, Monitoring, and Assessment Research,1400 Independence Avenue, SW Washington, DC 20250. Chapter 5.

Negro, F.; Bergman, R. 2019. Carbon stored by furnishing wood-based products: an Italian case study. Maderas. Ciencia y tecnología. 21(1): 65–76. https://doi.org/10.4067/S0718-221X2019005000106.

Nepal, P.; Abt, K.L.; Skog, K.E.; Prestemon, J.P.; Abt, R.C. 2019. Projected market competition for wood biomass between traditional products and energy: a simulated interaction of U.S. regional, national, and global forest product markets. Forest Science. 65(1): 14–26. https://doi.org/10.1093/forsci/fxy031.

Nepal, P.; Buongiorno, J.; Johnston, C.M.T.; Prestemon, J.; Guo, J.-G. 2021. Global forest products trade model. In: van Kooten, G.C.; Voss, L., eds. International trade in forest products: lumber trade disputes, models and examples. Wallingford UK: CABI: 110–141. https://doi.org/10.1079/9781789248234.0110.

Nepal, P.; Ince, P.J.; Skog, K.E.; Chang, S.J. 2013. Forest carbon benefits, costs and leakage effects of carbon reserve scenarios in the United States. Journal of Forest Economics. 19(3): 286–306. https://doi.org/10.1016/j.jfe.2013.06.001.

Nepal, P.; Prestemon, J.P.; Joyce, L.A.; Skog, K.E. 2022. Global forest products markets and forest sector carbon impacts of projected sea level rise. Global Environmental Change. 77: 102611. https://doi.org/10.1016/j.gloenvcha.2022.102611.

Nepal, P.; Skog, K.E. 2014. Estimating net greenhouse gas emissions from wood energy use: issues and the current state of knowledge. In: Aguilar, F.X., ed. Wood energy in developed economics: resource management, economics and policy. New York: Routledge: 223–252. Chapter 8.

Nepal, P.; Skog, K.E.; McKeever, D.B.; Bergman, R.D.; Abt, K.L.; Abt, R.C. 2016. Carbon mitigation impacts of increased softwood lumber and structural panel use for nonresidential construction in the United States. Forest Products Journal. 66(1–2): 77–87. https://doi.org/10.13073/FPJ-D-15-00019.

Nepal, P.; Wear, D.N.; Skog, K.E. 2014. Net change in carbon emissions with increased wood energy use in the United States. Global Change Biology Bioenergy. 7: 820–835. https://doi.org/10.1111/gcbb.12193.

Nichols, M.C. 2021. Personal communication. Science delivery specialist, U.S. Department of Agriculture, Forest Service, Northern Research Station, 271 Mast Road, Durham, NH 03824-4600. michael.c.nichols@usda.gov.

Nichols, M.C.; Morgan, T.A.; Christensen, G.; Tase, N. 2020. Estimated carbon stored in harvested wood products in Washington, USA: 1906–2018. Draft report. 30 p. Olympia, WA: Washington Department of Natural Resources. https://www.dnr.wa.gov/publications/em_wa_carbon_121420.pdf.

O'Connor, J.; Bowick, M. 2014. Advancing sustainable design with life cycle assessment. Ottawa, Ontario: Athena Sustainable Materials Institute. http://www.athenasmi.org/wp-content/uploads/2015/09/Advancing_Sustainable_Design_with_LCA.pdf.

Odegard, I.; Croezen, H.; Bergsma, G. 2012. Cascading of biomass: 13 solutions for a sustainable bio-based economy. Delft, Netherlands: CE Delft. https://cedelft.eu/wp-content/uploads/sites/2/2021/04/CE_Delft_2665_Cascading_of_Biomass_def.pdf.

Oneil, E.; Hall, E.S.; Ganguly, I.; Han, H.; Mahalle, L.; Malmsheimer, R.; McDonald, A.; Oliver, C.D.; Puettmann, K.; Lippke, B.; Puettman, M.; Quesada, H.; Rials, T.; Sinha, A.; Stewart, W.; Sumathipala, K.; Taylor, A.; Vlosky, R.; Volk, T. 2018. Letter to the Joint Interim Committee on Carbon Reduction, Oregon State Legislature. https://www.oregon.gov/ODF/ForestBenefits/Documents/Forest Carbon Study/Letter-092418-Forest-Carbon-Accounting-Scientist-JICCR.pdf.

Pan, W.; Kim, M.K.; Ning, Z.; Yang, H. 2020. Carbon leakage in energy/forest sectors and climate policy implications using meta-analysis. Forest Policy and Economics. 115: 102161. https://doi.org/10.1016/j.forpol.2020.102161.

Panwar, P.; Chauhan, S.; Kaushal, R.; Das, D.K.; Arora, G.; Chaturvedi, O.P.; Tewari, S. 2017. Carbon sequestration potential of poplar-based agroforestry using the CO2FIX model in the Indo-Gangetic region of India. Tropical Ecology. 58(2): 1–9.

Pingoud, K.; Skog, K.E.; Martino, D.L.; Tonosaki, M.; Xiaquan, Z. 2006. Harvested wood products. In: Eggleston, S.; Buendia, L.; Miwa, K.; Ngara, T.; Tanabe, K., eds. IPCC guidelines for national greenhouse gas inventories, Volume 4: Agriculture, forestry, and other land use. Hayama, Japan: Institute for Global Environmental Strategies. 33 p. Chapter 12. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_12_Ch12_HWP.pdf.

Pipatti, R.; Svardal, P.; Alves, J.W.S.; Gao, Q.; López-Cabrera, C.; Mareckova, K.; Oonk, H.; Scheehle, E.; Sharma, C.; Smith, A.; Yamada, M. 2006. Solid waste disposal. In: Eggleston, S.; Buendia, L.; Miwa, K.; Ngara, T.; Tanabe, K., eds. IPCC guidelines for national greenhouse gas inventories. Volume 5: Waste. Hayama, Japan: Institute for Global Environmental Strategies. 40 p. Chapter 3. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_3_Ch3_SWDS.pdf.

Puettmann M.E.; Bergman, R.D.; Hubbard, S.S.; Johnson, L.; Lippke, B.; Wagner, F. 2010. Cradle-to-gate life-cycle inventories of U.S. wood products production—CORRIM phase I and phase II products. Wood and Fiber Science. 42 (Spec. issue): 15–28.

Puettmann, M.E.; Wilson, J.B. 2005. Life-cycle analysis of wood products: cradle-to-grave LCI of residential wood building materials. Wood and Fiber Science. 37 (Spec. issue): 18–29.

Reed, D.; Bergman, R.; Taylor, A.; Harper, D.; Jones, P.; Knowles, C.; Puettmann, M. 2012. Cradle-to-gate life-cycle inventory and impact assessment of wood fuel pellet manufacturing from hardwood flooring residues in the Southeastern United States. Forest Products Journal. 62(4): 280–288. https://doi.org/10.13073/FPJ-D-12-00015.1.

Risse, M.; Weber-Blaschke, G.; Richter, K. 2017. Resource efficiency of multifunctional wood cascade chains using LCA and exergy analysis, exemplified by a case study for Germany. Resources, Conservation and Recycling. 126: 141–152. https://doi.org/10.1016/j.resconrec.2017.07.045.

Row, C.; Phelps, R.B. 1991. Carbon cycle impacts of future forest products utilization and recycling trends. In: Agriculture in a world of change: proceedings of outlook ’91, 67th annual outlook conference. Washington, DC: U.S. Department of Agriculture. 786 p.

Row, C.; Phelps, R.B. 1996. Wood carbon flows and storage after timber harvest. In: Sampson, R.N.; Hair, D., eds. Forests and global change: forest management opportunities for mitigating carbon emissions. Washington, DC: American Forests: 27–58. Vol. 2.

Rüter, S.; William, R.; Lundblad, M.; Sato, A., Hassan, R.A. 2019. Harvested wood products. In: Calvo Buendia, E.; Tanabe, K.; Kranjc, A.; Baasansuren, J.; Fukuda, M.; Ngarize S.; Osako, A.; Pyrozhenko, Y.; Shermanau, P.; Federici, S., eds. 2019 refinement to the 2006 IPCC guidelines for national greenhouse gas inventories. Volume 4: Agriculture, forestry, and other land use. Geneva, Switzerland: Intergovernmental Panel on Climate Change, Task Force on National Greenhouse Gas Inventories. 49 p. Chapter 12.

Sahoo, K.; Bergman, R.; Alanya-Rosenbaum, S.; Gu, H.; Liang, G. 2019. Life cycle assessment of forest-based products: a review. Sustainability. 11(17): 4722. https://doi.org/10.3390/su11174722.

Salazar, J.; Bergman, R. 2013. Temporal considerations of carbon sequestration in LCA. In: Proceedings from the LCA XIII conference. Orlando, FL. 136–142.

Sathre, R.; O'Connor, J. 2010. Meta-analysis of greenhouse gas displacement factors of wood product substitution. Environmental Science and Policy. 13(2): 104–114. https://doi.org/10.1016/j.envsci.2009.12.005.

Sato, A.; Nojiri, Y. 2019. Assessing the contribution of harvested wood products under greenhouse gas estimation: accounting under the Paris Agreement and the potential for double-counting among the choice of approaches. Carbon Balance and Management. 14: 15.

Schelhaas, M.J.; van Esch, P.W.; Groen, T.A.; de Jong, B.H.J.; Kanninen, M.; Liski, J.; Masera, O.; Mohren, G.M.J.; Nabuurs, G.J.; Palosua, T.; Pedroni, L.; Vallejo, A.; Vilén, T. 2004. CO2Fix V3.1: a modelling framework for quantifying carbon sequestration in forest ecosystems. Wageningen, Netherlands: Wageningen Universiteit, Alterra-Centrum Ecosystemen. 122 p. http://edepot.wur.nl/43524. (January 23, 2023).

Sedjo, R.; Tian, X. 2012. Does wood bioenergy increase carbon stocks in forests? Journal of Forestry. 110(6): 304–311. https://doi.org/10.5849/jof.11-073.

Simmons, E.; Berg, E.; Morgan, T. 2016. Timber Products Output (TPO): forest inventory, timber harvest, mill and logging residue-essential feedstock information needed to characterize the NARA supply chain. Missoula, MT: University of Montana, Bureau of Business and Economic Research. 37 p.

Simmons, E.A.; Marcille, K.C.; Lettman, G.J.; Morgan, T.A.; Smith, D.C.; Rymniak, L.A.; Christensen, G.A. 2021. Oregon's forest products industry and timber harvest 2017 with trends through 2018. Gen. Tech. Rep. PNW-GTR-997. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 63 p.

Sirkin, T.; ten Houten, M. 1994. The cascade chain: a theory and tool for achieving resource sustainability with applications for product design. Resources, Conservation, and Recycling. 10(3): 213–276. https://doi.org/10.1016/0921-3449(94)90016-7.

Skog, K.E. 2008. Sequestration of carbon in harvested wood products for the United States. Forest Products Journal. 58(6): 56–72.

Skog, K.E.; McKinley, D.C.; Birdsey, R.A.; Hines, S.J.; Woodall, C.W.; Reinhardt, E.D.; Vose, J.M. 2014. Managing carbon. In: Peterson, D.L.; Vose, J.M.; Patel-Weynand, T., eds. Climate change and United States forests. Advances in Global Change Research, vol. 57. Dordrecht, Netherlands: Springer: 151–182. https://doi.org/10.1007/978-94-007-7515-2_7. Chapter 7.

Skog, K.E.; Nicholson, G.A. 1998. Carbon cycling through wood products: the role of wood and paper products in carbon sequestration. Forest Products Journal. 48(7/8): 75–83.

Skog, K.E.; Nicholson, G.A. 2000. Carbon sequestration in wood and paper products. In: Joyce, L.A.; Birdsey, R., tech. eds. 2000. The impact of climate change on America's forests. Gen. Tech. Rep. RMRS-GTR-59. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 79–88.

Smith, J.E.; Heath, L.S.; Skog, K.E.; Birdsey, R.A. 2006. Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States. Gen. Tech. Rep. NE-343. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station. 216 p. https://doi.org/10.2737/NE-GTR-343.

Smyth, C.; Rampley, G.; Lemprière, T.C.; Schwab, O.; Kurz, W.A. 2017. Estimating product and energy substitution benefits in national-scale mitigation analyses for Canada. GCB Bioenergy. 9: 1071–1084. https://doi.org/10.1111/gcbb.12389.

Smyth, C.; Stinson, G.; Neilson, E.; Lemprière, T.; Hafer, M.; Rampley, G.; Kurz, W. 2013. Quantifying the biophysical climate change mitigation potential of Canada's forest sector. Biogeosciences Discussions. 11(13): 3515–3529. https://doi.org/10.5194/bg-11-3515-2014.

Smyth, C.E.; Xu, Z.; Lemprière, T.C.; Kurz, W.A. 2020. Climate change mitigation in British Columbia's forest sector: GHG reductions, costs, and environmental impacts. Carbon Balance and Management. 15: 21. https://doi.org/10.1186/s13021-020-00155-2.

Sohngen, B.; Brown, S. 2008. Extending timber rotations: carbon and cost implications. Climate Policy. 8(5): 435–451. https://doi.org/10.3763/cpol.2007.0396.

Stockmann, K.D. 2021. Personal communication. Economist, National Technology Development Program, 5785 Highway 10 W, Missoula, MT 59808. keith.stockmann@usda.gov.

Stockmann, K.D. 2022. Personal communication. Economist, National Technology Development Program, 5785 Highway 10 W, Missoula, MT 59808. keith.stockmann@usda.gov.

Stockmann, K.D.; Anderson, N.M.; Skog, K.E.; Healy, S.P.; Loeffler, D.R.; Jones, G.; Morrison, J.F. 2012. Estimates of carbon stored in harvested wood products from the United States Forest Service Northern Region, 1906–2010. Carbon Balance and Management. 7(1): 1–16. https://doi.org/10.1186/1750-0680-7-1.

Stockmann, K.; Anderson, N.; Young, J.; Skog, K.; Healey, S.; Loeffler, D.; Butler, E.; Jones, J.G.; Morrison, J. 2014. Estimates of carbon stored in harvested wood products from United States Forest Service Northern Region, 1906–2012. 27 p. Unpublished report. On file with: Department of Agriculture, Forest Service, Rocky Mountain Research Station, 800 East Beckwith Avenue, Missoula, MT 59801-5801.

Suominen, T.; Kunttu, J.; Jasinevičius, G.; Tuomasjukka, D.; Lindner, M. 2017. Trade-offs in sustainability impacts of introducing cascade use of wood. Scandinavian Journal of Forest Research. 32(7): 588–597. https://doi.org/10.1080/02827581.2017.1342859.

Taskhiri, M.S.; Jeswani, H.; Geldermann, J.; Azapagic, A. 2019. Optimising cascaded utilization of wood resources considering economic and environmental aspects. Computers and Chemical Engineering. 124: 302–316. https://doi.org/10.1016/j.compchemeng.2019.01.004.

Thomassen, M.A.; Dalgaard, R.; Heijungs, R.; de Boer, I. 2008. Attributional and consequential LCA of milk production. The International Journal of Life Cycle Assessment. 13: 339–349. https://doi.org/10.1007/s11367-008-0007-y.

United Nations Economic Commission for Europe and Food and Agriculture Organization of the United Nations [UNECE and FAO]. 2021. Forest sector outlook study: 2020–2040. Geneva, Switzerland. 90 p. https://unece.org/sites/default/files/2021-11/SP-51-2021-11_0.pdf.

United Nations Food and Agriculture Statistics [UN FAOSTAT]. 2022. Forestry production and trade [Database]. https://www.fao.org/faostat/en/#data/FO. (January 31, 2023).

United Nations Framework Convention on Climate Change [UNFCCC]. 1998. Kyoto protocol to the United Nations framework convention on climate change adopted at COP3 in Kyoto, Japan, on December 11, 1997.

United Nations Framework Convention on Climate Change [UNFCCC]. 2003. Estimation, reporting and accounting of harvested wood products. Tech. Pap. FCCC/TP/2003/7. Bonn, Germany. 44 p.

United Nations Framework Convention on Climate Change [UNFCCC]. 2015. The Paris Agreement. https://unfccc.int/process-and-meetings/the-paris-agreement. (August 15, 2023).

United Nations Framework Convention on Climate Change [UNFCCC]. 2023. What is the United Nations framework convention on climate change? https://unfccc.int/process-and-meetings/what-is-the-united-nations-framework-convention-on-climate-change. (February 14, 2022).

University of Montana, Bureau of Business and Economic Research [UM-BBER]. 2021. Harvest by county tool. https://bber.umt.edu/FIR/H_harvest.asp. (January 31, 2023).

U.S. Department of Agriculture, Forest Service [USDA FS]. 1982. An analysis of the timber situation in the United States, 1952–2030. Forest Resource Rep. 23. Washington, DC. 520 p. https://www.fs.usda.gov/research/docs/rpa/pre-1989/timber-situation-1979.pdf.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012. Timber Products Output (TPO) reports. Knoxville, TN: Southern Research Station. https://www.fia.fs.usda.gov/program-features/tpo. (March 1, 2021).

U.S. Department of Agriculture, Forest Service [USDA FS]. 2022. Forest products cut and sold from the national forests and grasslands. https://www.fs.usda.gov/forestmanagement/products/cut-sold/index.shtml. (January 17, 2023).

U.S. Energy Information Administration [US EIA]. 2021a. Table 10.2a Renewable energy consumption: residential and commercial sectors. Total energy. https://www.eia.gov/totalenergy/data/browser/index.php?tbl=T10.02A#/?f=M&start=200001. (January 17, 2023).

U.S. Energy Information Administration [US EIA]. 2021b. State energy data system (SEDS): 1960–2019. State Profiles and Energy Estimates. https://www.eia.gov/state/seds/seds-data-complete.php?sid=US. (January 17, 2023).

U.S. Environmental Protection Agency [US EPA]. 2005. Municipal solid waste in the United States: 2005 facts and figures. EPA530-R-06-011. Washington, DC. 164 p. https://archive.epa.gov/epawaste/nonhaz/municipal/web/pdf/mswchar05.pdf.

U.S. Environmental Protection Agency [US EPA]. 2007. Municipal solid waste generation, recycling, and disposal in the United States: facts and figures for 2006. EPA-530-F-07-030. Washington, DC. 11 p. https://archive.epa.gov/epawaste/nonhaz/municipal/web/pdf/msw06.pdf.

U.S. Environmental Protection Agency [US EPA]. 2017. User's guide for estimating emissions and sinks from land use, land-use change, and forestry using the state inventory tool. [Reston, VA: ICF International, Inc.]. 24 p. https://www.epa.gov/sites/default/files/2017-04/documents/land-use_change_and_forestry_users_guide.pdf.

U.S. Environmental Protection Agency [US EPA]. 2020a. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2018. Washington, DC. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2018. (January 17, 2023).

U.S. Environmental Protection Agency [US EPA]. 2020b. Waste Reduction Model (WARM), Version 15. Updated November 2020. https://www.epa.gov/warm/versions-waste-reduction-model#15. (January 17, 2023).

U.S. Environmental Protection Agency [US EPA]. 2021. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2019. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2019. (January 17, 2023).

Wang, X.; Barlaz, M.A. 2016. Decomposition and carbon storage of hardwood and softwood branches in laboratory-scale landfills. Science of the Total Environment. 557–558: 355–362. https://doi.org/10.1016/j.scitotenv.2016.03.091.

Wang, X.; De la Cruz, F.B.; Ximenes, F.; Barlaz, M.A. 2015. Decomposition and carbon storage of selected paper products in laboratory-scale landfills. Science of the Total Environment. 532: 70–79. https://doi.org/10.1016/j.scitotenv.2015.05.132.

Wang, X.; Padgett, J.M.; De la Cruz, F.B.; Barlaz M.A. 2011. Wood biodegradation in laboratory-scale landfills. Environmental Science and Technology. 45(16): 6864–6871. https://doi.org/10.1021/es201241g.

Wang, X.; Padgett, J.M.; Powell, J.S.; Barlaz, M.A. 2013. Decomposition of forest products buried in landfills. Waste Management. 33(11): 2267–2276. https://doi.org/10.1016/j.wasman.2013.07.009.

Wear, D.N.; Coulston, J.W. 2019. Specifying forest sector models for forest carbon projections. Journal of Forest Economics. 34(1–2): 73–97. https://doi.org/10.1561/112.00000443.

Wear, D.N.; Murray, B.C. 2004. Federal timber restrictions, interregional spillovers, and the impact on U.S. softwood markets. Journal of Environmental Economics and Management. 47(2): 307–330. https://doi.org/10.1016/S0095-0696(03)00081-0.

Xie, S.H. 2015. Modelling the decay patterns and harvested wood product pools of residential houses. Vancouver, BC: University of British Columbia. 127 p. M.A.S. thesis.

Xie, S.H. 2020. Forest sector mitigation consequences of cascading uses of British Columbia's postconsumer wood products. Vancouver, BC: University of British Columbia. 239 p. Ph.D. dissertation.

Xie, S.H.; Kurz, W.A.; McFarlane, P.N. 2021. Inward- versus outward-focused bioeconomy strategies for British Columbia's forest products industry: a harvested wood products carbon storage and emission perspective. Carbon Balance and Management. 16: 30 (2021). https://doi.org/10.1186/s13021-021-00193-4.

Ximenes, F.A.; Cowie, A.L.; Barlaz, M.A. 2018. The decay of engineered wood products and paper excavated from landfills in Australia. Waste Management. 74: 312–322. https://doi.org/10.1016/j.wasman.2017.11.035.

Xu, Z.; Smyth, C.E.; Lemprière, T.C.; Rampley, G.J.; Kurz, W.A. 2018. Climate change mitigation strategies in the forest sector: biophysical impacts and economic implications in British Columbia, Canada. Mitigation and Adaptation Strategies for Global Change. 23: 257–290. https://doi.org/10.1007/s11027-016-9735-7.

Zanchi, G.; Pena, N.; Bird, N. 2012. Is wood bioenergy carbon neutral? A comparative assessment of emissions from consumption of woody bioenergy and fossil fuel. GCB Bioenergy. 4(6): 761–772. https://doi.org/10.1111/j.1757-1707.2011.01149.x.

APPENDIX 1
Acronyms and Abbreviations Used in This Report

Acronym	Definition
ALCA	Attributional life-cycle assessment
ANSE	Abstract Network Simulation Engine
CARB	California Air Resources Board
CBM-CFS3	Carbon Budget Model of the Canadian Forest Sector
CBM-FHWP	Carbon Budget Model Framework for Harvested Wood Products
CLCA	Consequential life-cycle assessment
FAO	Food and Agriculture Organization
FIA	(U.S. Department of Agriculture Forest Service) Forest Inventory and Analysis
FVS	Forest Vegetation Simulator
GFPM	Global Forest Products Model
GHG	Greenhouse gas
HWP	Harvested wood product
IPCC	International Panel on Climate Change
ISO	International Organization of Standardization
LCA	Life-cycle assessment
LCI	Life-cycle inventory
LCIA	Life-cycle impact assessment
NFS	National Forest System
NGHGI	National greenhouse gas inventory
PRESTO	Product Estimation Tool Online
SF	Substitution factor
SWDS	Solid waste disposal site
TPO	Timber Products Output
UNECE	United Nations Economic Commission for Europe
UN FAOSTAT	United Nations Food and Agriculture Organization Corporate
USDA	U.S. Department of Agriculture
USFPM	United States Forest Products Model (module run with GFPM)
USFS	U.S. (Department of Agriculture) Forest Service
US EPA	U.S. Environmental Protection Agency
WARM	(U.S. Environmental Protection Agency's) Waste Reduction Model

APPENDIX 2

Overview of Harvested Wood Product Carbon Estimation Methods from General Technical Report NE-343

U.S. Department of Agriculture Forest Service General Technical Report (GTR) NE-343 (Smith et al. 2006) is not a harvested wood product (HWP) carbon estimation model. However, it is one of the first documents to describe U.S. HWP and forest carbon quantification methods following International Panel on Climate Change tier 3 approaches. Smith et al. (2006) provided several datasets used in various U.S. HWP carbon estimation tools and models. The document describes two methods (land based and product based) for estimating HWP carbon storage and emissions for up to 100 years after harvest for 51 different forest types across 10 regions of the United States (summarized below). Skog (2008), another guiding document for HWP carbon estimation approaches in the United States, contains methods and datasets updated from Smith et al. (2006) to establish quantification methods for cumulative HWP carbon stocks and stock changes from harvests dating back to 1900 for the U.S. national greenhouse gas inventory in the WOODCARB model and its variants. Methods in Skog (2008) also provide the basis for the original Excel model used for National Forest System (NFS) HWP carbon inventories composing the Northern Region (e.g., Anderson et al. 2013, Stockmann et al. 2012) and later coded into the USFS HWP-C v1 model for use in all NFS regional HWP carbon inventories (e.g., Stockmann et al. 2014).

Land-based estimates use Forest Inventory and Analysis (FIA) growing-stock harvest volumes determined by regional forest type stand look-up tables (per acre). Growing-stock volume is converted to softwood and hardwood sawlog and pulpwood categories using predefined merchantability specifications derived from FIA data. Volumes are then allocated to carbon using the specific gravities of softwood and hardwood species. However, not all harvested growing stock becomes harvested roundwood. Some portion of the growing stock remains in the forest as logging residue and is determined by regional ratios (e.g., Timber Products Output data). Regional ratios are based on allometric equations (Jenkins et al. 2003) and FIA data. The ratios are then applied to quantities of softwood and hardwood sawlog and pulpwood of harvested growing-stock carbon associated with harvested roundwood, bark, and fuelwood. Another set of regional look-up tables distinguished by forest type and roundwood category (sawlog/pulpwood) provides ratios to allocate the amount of carbon in the roundwood growing-stock volume to four HWP carbon dispositions, or subpools (products in use, in solid waste disposal sites, emitted with energy capture, and emitted without energy capture), for up to 100 years after harvest. These ratios provide a means to track the harvested carbon through time and provide estimates of future carbon storage in these pools up to 100 years after harvest. Carbon associated with fuelwood and bark on roundwood are assumed to be emitted the same year as harvest. Thus, the carbon is added to the two emitted categories at the time of harvest; all fuelwood and a portion of bark on roundwood are emitted with energy capture. Although the details are not specified, ratios in these regional look-up tables incorporate primary product ratios, end-use product ratios, end-use product half-lives, discarded disposition half-lives (including recycling), and landfill fixed carbon ratios. Data sources for several of these parameters provided in appendix D of Smith et al. (2006) are older than those used in Skog (2008) and Stockmann (2012).

Land-based estimation methods rely on determining growing-stock volume from a table in Smith et al. (2006) (using FIA data from the mid-2000s) and does not require harvest data. This HWP carbon estimation method simply requires an acreage and general forest type for potential harvest operations. Additionally, while this synthesis focuses on HWP carbon models, the land-based HWP carbon estimation method described in Smith et al. (2006) can be applied without the use of a model or with modeled harvest volumes (i.e., no harvest data is required). This approach could help evaluate HWP carbon storage and losses associated with different forest management scenarios to weigh tradeoffs; however, underlying datasets may need to be updated. The information provided from the land-based method could be used to understand trends in carbon stocks rather than provide more precise estimates, such as those required for a greenhouse gas inventory.

The second method Smith et al. (2006) describes is the product-based method. This method starts carbon estimation calculations with one of two options: (1) a known amount of industrial roundwood volume delivered to mills, or (2) the primary product output from mills. Industrial roundwood volumes in units of dry tons can be directly converted to carbon and allocated to HWP carbon pools and tracked through time using the regional look-up tables in Smith et al. (2006) described above. If using primary product output from mills, these values can be converted from primary products to carbon using ratios in Smith et al. (2006) and then allocated to HWP carbon pools. These carbon pools can be tracked through time up to 100 years after harvest using look-up tables described in Smith et al. (2006). Units for conversion ratios by primary product differ for each primary product. However, the document does not specify the methods or data sources used to derive the primary product volume to carbon conversions, and it is unclear how species mix is considered in those conversions.

As with the land-based method, if an industrial roundwood volume is derived from other sources, such as a field timber cruise, the subsequent calculations can be applied to evaluate associated HWP carbon storage and losses without the need for a model. Similarly, the approach can be applied to volumes derived from other models, such as those evaluating different forest management scenarios, and therefore help evaluate tradeoffs between different scenarios. Lastly, although the regional look-up tables have a variety of values for specific parameters and calculations required for tracking the fate of HWP carbon through time, Smith et al. (2006) provides detailed instructions for the computational steps required along the way that result in the final look-up tables.

APPENDIX 3

Assumptions, Data Sources, and Discrepancies for End-Use Product Half-Lives Used in USFS HWP-C v1

Original documentation for USFS HWP-C v1 and related models (e.g., Anderson et al. 2013; Stockmann et al. 2012, 2014) did not specify a data source for end-use product half-lives; however, personal communication with an expert (Stockmann 2021) confirms that these values are from Skog and Nicholson (2000). These values are also summarized in table D3 of Smith et al. (2006), which cited Skog and Nicholson (1998), Row and Phelps (1996), and a personal communication with someone at the USDA Forest Service Forest Products Laboratory (Smith et al.'s [2006] table D3 cites Skog and Nicholson [1998], which is identical to Skog and Nicholson [2000]). Smith et al. (2006) provides no additional information about the personal communication. However, not all end-use product half-lives in USFS HWP-C v1 are published. Some half-lives used for one product category were assumed to be sufficient for others that lacked data. For example, "manufactured products" are assumed to have a 12-year half-life in Skog and Nicholson (1998, 2000), but this half-life was also applied to the "other industrial uses" end-use product category in USFS HWP-C v1 because there were no data for this particular end-use product category at the time. USFS HWP-C v1 uses an end-use product half-life of 100 years for single-family homes from Smith et al. (2006), whereas values provided in Skog (2008) and Dymond (2012) are slightly lower. Dymond (2012) provides a range of values of 78 to 350 years. USFS HWP-C v1 uses a half-life of 2.6 years for paper from Smith et al. (2006), which is the only value unique to Smith et al. (2006) and differs from the values in Skog and Nicholson (2000), Skog (2008), and in International Panel on Climate Change publications (IPCC 2003, 2014; Pingoud et al. 2006). In general, end-use product half-lives in USFS HWP-C v1 were taken from as few sources as possible to ensure consistency (Stockmann 2021). Table A3-1 identifies data sources, assumptions where data were missing for end-use product half-lives used in USFS HWP-C v1 and subsequent versions, and differences in values with more recent publications.

Table A3-1—Assumptions and data sources for end-use product half-lives used in the USFS HWP-C v1 model and subsequent versions

End-use product	Half-life	Comments
	YEARS	
New housing, multifamily	70	Value from Skog and Nicholson (2000). Value of 150 years for "multifamily homes" in Row and Phelps (1996), cited in Smith et al. (2006). Values range from about 47 to 59 years before calibrating the estimate with U.S. housing data and 48 to 51 years after calibration in Skog (2008). Value of 75 years, with a range of 48 to 350 years, for "multifamily homes and commercial buildings" in Dymond (2012).
New housing, single family	100	Value from Skog and Nicholson (2000), corresponds to post-1980 value. Value of 200 years for "single-family homes" in Row and Phelps (1996), cited in Smith et al. (2006). Values range from about 75 to 95 years before calibrating the estimate with U.S. housing data and 78 to 84 years after calibration in Skog (2008). Value of 90 years, with a range of 78 to 350 years, in Dymond (2012).
New housing, manufactured housing	12	No value for this category in Skog and Nicholson (2000). Appears to correspond to value for "manufacturing" in Skog and Nicholson (2000). Also corresponds to value for "mobile homes" in Smith et al. (2006) rather than the value of 20 years for "mobile homes" in Skog and Nicholson (2000). Smith et al. (2006) cites a "personal communication" with someone at the USDA Forest Service Forest Products Laboratory, but it is unclear which half-lives are attributed to that correspondence. Value of 12 years for "mobile homes" and "manufactures" in Row and Phelps (1996), cited in Smith et al. (2006). Value of 30 years, with a range of 5 to 50 years, for "residential upkeep and moveable homes" in Dymond (2012).
New nonresidential, buildings	67	Corresponds to value for "nonresidential construction" in Skog and Nicholson (2000) and Row and Phelps (1996), cited in Smith et al. (2006).
New nonresidential, other	67	Corresponds to value for "nonresidential construction" in Skog and Nicholson (2000) and Row and Phelps (1996), cited in Smith et al. (2006).
Residential remodel and repair	30	No value for this category in Skog and Nicholson (2000). Could correspond to value of 30 years for "residential maintenance and repair" in Row and Phelps (1996), cited in Smith et al. (2006). Smith et al. (2006) cites a "personal communication" with someone at the USDA Forest Service Forest Products Laboratory, but it is unclear which half-lives are attributed to that correspondence. Values range from about 24 to 31 years before calibration with U.S. housing data and 23 to 25 years after calibration in Skog (2008). Value of 30 years, with a range of 5 to 50 years, for "residential upkeep and moveable homes" in Dymond (2012).

Table A3-1 (continued)—Assumptions and data sources for end-use product half-lives used in the USFS HWP-C v1 model and subsequent versions

End-use product	Half-life	Comments
Manufacturing, furniture	30	Corresponds to value in Skog and Nicholson (2000). Value of 38 years, with a range of 19 to 38 years, for “furniture and other manufactured products” in Dymond (2012).
Manufacturing, other	12	Corresponds to value in Skog and Nicholson (2000) and for “manufactures” in Row and Phelps (1996), cited in Smith et al. (2006).
Packaging and shipping	6	Corresponds to value for “pallets” in Skog and Nicholson (2000) and for “shipping” in Row and Phelps (1996) Value of 2 years, with a range of 1 to 2 years, for “shipping” in Dymond (2012).
Other	12	No value for this category in Skog and Nicholson (2000). It does correspond to their value for “manufacturing.” Could correspond to a value of 30 years for “other solid wood uses” in Row and Phelps (1996). Smith et al. (2006) cites a “personal communication” with someone at the USDA Forest Service Forest Products Laboratory, but it is unclear which half-lives are attributed to that correspondence. Value of 38 years for “solidwood in all other end-uses” in Skog (2008) and for “other” (remainder of wood consumption) in Dymond (2012), with a range of 19 to 38 years in Dymond (2012).
Rail and railcar	12	No value for this category in Skog and Nicholson (2000). Corresponds to their value for “manufacturing.”
Other industrial products	12	No value for this category in Skog and Nicholson (2000). Corresponds to their value for “manufacturing.” Smith et al. (2006) cites a “personal communication” with someone at the USDA Forest Service Forest Products Laboratory, but it is unclear which half-lives are attributed to that correspondence.
Wood pulp	2.6	Value is from Smith et al. (2006), but it is unclear from where this originates. Could correspond to a value of 6 years for “printing and writing paper” and 1 year for “newsprint, tissue paper, and packaging paper and board” in Row and Phelps (1996), also cited in Smith et al. (2006). Smith et al. (2006) cites a “personal communication” with someone at the USDA Forest Service Forest Products Laboratory, but it is unclear which half-lives are attributed to that correspondence. Value is 6 years for “free sheet” or 1 year for “all other” in Skog and Nicholson (2000), which comprise the Intergovernmental Panel on Climate Change (IPCC 2003) default for the U.S. The “estimated value” is 2.53 years, and the “mean value distribution” is 2.59 years in Skog (2008). The Intergovernmental Panel on Climate Change (IPCC 2006a, 2014) default is 2 years.
Poles, pilings, mine props, posts, nonsawtimber, railroad ties, coop bolts, acids and other distilled products (e.g., wood alcohol), float logs, trap float, miscellaneous convertible/converted products (e.g., bags, envelopes), naval stores (e.g., pine tar, pitch), cull logs, small roundwood, convertible green biomass, convertible dry biomass, specialty products[a]	12	No values for these end-use products in Skog and Nicholson (2000) but could correspond to their value for “other uses for lumber and panels.” Could correspond to the value for “manufactures” in Row and Phelps (1996). Smith et al. (2006) cites a “personal communication” with someone at the USDA Forest Service Forest Products Laboratory, but it is unclear which half-lives are attributed to that correspondence.

[a]All individual end-uses in USFS HWP-C v1, lumped here for brevity.

APPENDIX 4
Fixed Carbon in Landfills: Partial Sensitivity Analysis for HWP-C vR Using 1906–2018 Oregon Harvest Data

When harvested wood products (HWPs) are discarded using HWP-C vR, they are aggregated into either "solid wood" or "paper." For the landfill discard category, these products are additionally divided by degradable and nondegradable ratios (also referred to as "fixed" or "inert" carbon ratio) within a landfill. Landfills create anaerobic decay environments, and wood products contain at least some lignin, leading to a slower rate of decomposition than in aerobic waste environments (Wang et al. 2013). The fixed portion of carbon in wood is treated as permanent carbon storage in the landfill. Decomposition rates can differ among landfills and within a single landfill because the rate of decay is most strongly influenced by the waste characteristics and moisture content of a landfill (De la Cruz and Barlaz 2010). The decaying portion of wood products emits carbon dioxide (CO_2), methane (CH_4), and other greenhouse gases (GHGs). CH_4 is the second most common GHG in the atmosphere and is 25 times more potent than CO_2 (IPCC 2007); however, CH_4 can also be captured for energy production or flared, producing CO_2.

Studies in the last decade examined decomposition of solid wood and paper samples at two commercial facilities on the east coast of the United States (Wang et al. 2013). Wang et al. (2013) sampled all the products listed in table A4-1, except wood flooring in a landfill environment. Solid wood samples were taken from a commercial lumberyard and corrugated containers were purchased new. A caveat with this approach is that these samples were new and untreated (i.e., they did not go into use before disposal). This may affect the rate of decomposition and result in a longer half-life for products than those that have been weathered or reused and recycled before being buried in a landfill. These samples were removed 1.5 and 2.5 years after being buried. Wang et al. (2013) compared the decomposition of these samples to the findings in their previous lab study (Wang et al. 2011) and found more degradation in newspaper and cardboard boxes in the field than the lab. Copy paper exhibited the most degradation in both the lab and field. Their findings for both studies consistently demonstrated that products with more lignin had less degradation. These findings were similar to conclusions drawn from a previous study that estimated carbon storage and emission (CO_2 and CH_4) potential for various forest products (Micales and Skog 1997). The estimates from Micales and Skog (1997) were also used for landfill carbon storage and emission estimates as part of International Panel on Climate Change's (IPCC's) national GHG guidelines, using the original WOODCARB model (Freed et al. 2004).

Under landfill conditions, Wang et al. (2013) observed that the carbon conversion of solid wood products is much lower than the 50 percent degradable organic carbon that earlier IPCC (2006) guidance recommended; in other words, a lower proportion of solid wood decays. Solid wood has a higher proportion of lignin than paper; however, within categories of paper, there are substantial differences in proportions of lignin (Wang et al. 2013). These differences in lignin content affect the CH_4 potential of HWPs because some will contain greater proportions of degradable material than others, and furthermore, some will decompose more quickly and more completely than others in a landfill (i.e., have shorter half-lives), contributing to higher CH_4 emissions. There is ongoing research between the lead author and the California department of Forestry and Fire Protection (CAL FIRE) to develop methods to incorporate disaggregated fixed carbon storage proportions for wood and paper product categories.

Imperfect empirical data exist about the decomposition rate of wood products in landfills. However, since the development of USFS HWP-C v1 model (e.g., Stockmann et al. 2012) and its variants, a few recent studies about wood products in landfills, such as those described above, have been conducted. There have also been updates to the US EPA WARM v-15 (2020b) fixed carbon ratios estimates for wood and paper. Given the limitations of previous fixed carbon ratio estimates, the purpose of this appendix is to perform a limited sensitivity analysis for fixed carbon ratios in landfills using HWP-C vR to see how landfill carbon emissions, carbon storage, and overall solid waste disposal site (SWDS) estimates are affected using Oregon harvest data as an example.

Table A4-1—Literature sources and fixed carbon ratio values for harvested wood products discarded in landfills

Product	Micales and Skog (1997)[a]	Barlaz (1998)	Freed and Mintz (2003)[b]	US EPA (2020b)[c]
Solid wood	0.97 to 1	0.77[d]	0.77[d]	n.d.
Paper	n.d.	n.d.	0.44 (average)[e]	n.d.
Branches	0.97 to 1	0.77[c]	0.77[d]	0.77
Newspaper	0.84	0.85	0.8	0.84
Corrugated containers	0.69	0.55	0.55	0.55
Office paper	0.62	0.125	0.12	0.12
Coated paper	0.83	0.99	0.68	0.74
Dimensional lumber	n.d.	n.d.	n.d.	0.88
Medium-density fiberboard	n.d.	n.d.	n.d.	0.84
Wood flooring	n.d.	n.d.	n.d.	0.95

[a] Estimates were not confirmed by field studies (estimates are rounded to nearest hundredth).

[b] Freed and Mintz (2003) is a memo about revised input data for the WOODCARB model (not publicly available). Freed and Mintz relied on Barlaz (1998) and others for initial calculations. ICF International, Inc. prepared the memo and submitted it to the U.S. Environmental Protection Agency in 2003, and landfill fixed carbon estimates in the memo were additionally cited in Dymond (2012), Skog (2008), and Smith et al. (2006).

[c] Data used to inform US EPA WARM v-15 (US EPA 2020b), exhibit 6-5, "Adjusted Carbon Storage as Proportion of Initial Carbon" (Barlaz 1998, 2022; Levis et al. 2013; Wang et al. 2011, 2013).

[d] The estimated fixed carbon ratio for solid wood and branches was 0.77, which was based on data from branches in a lab environment that mimicked a landfill. Branches were used as a proxy for all solid wood, including medium-density fiberboard. (Barlaz 1998).

[e] Displayed as an "average" in Freed and Mintz (2003) memo but does not display calculations for the average.

n.d. = no data.

Stockmann et al. (2012) cited Smith et al. (2006) and Skog (2008) for a landfill fixed carbon ratio of 0.44 for all paper and 0.77 for all solid wood, which differs from IPCC's recommendation as well as Micales and Skog's (1997) estimates used for national greenhouse gas inventories for HWPs, using the original WOODCARB model (USFS HWP-C cites Smith et al. [2006] for these figures, but they are originally from a collaboration between Freed and Mintz [2003] and Barlaz [1998]). However, these landfill fixed carbon ratios came from an unpublished spreadsheet that is not publicly available (Freed and Mintz 2003). Part of the spreadsheet was reproduced in a memorandum for the U.S. Environmental Protection Agency as an update for the original WOODCARB national greehouse gas inventory (Freed et al. 2004). The memo is referenced in table A4-1 as Freed and Mintz (2003). Table A4-1 lists Freed and Mintz's (2003) fixed carbon ratios for a handful of products that were derived from calculations by the authors at that time, in collaboration with Barlaz's (1998) work (Freed 2021).

More recently, research collaborations between Barlaz and Wang et al. (2011, 2013) informed US EPA WARM v-15 (US EPA 2020b) estimates for fixed carbon ratios in specific wood and paper products disposed of in landfills. The fixed carbon ratios in US EPA WARM v-15 were used to test the sensitivity of fixed carbon (table A4-1). There are limitations to applying US EPA WARM v-15 estimates to USFS HWP-C v1 and its current variants. US EPA WARM v-15 contains more recent estimates of fixed carbon ratios for specific wood and paper products, whereas USFS HWP-C v1 and its current variants aggregate all paper into one category and all solid wood into another for landfills. There are limited empirical data about the fixed carbon ratios of specific wood products in landfills because surveys about solid waste disposal sites have focused on overall waste streams rather than disposal of specific products (Ximenes et al. 2018). Another limitation of US EPA WARM v-15 fixed carbon ratios is that they originate from a national-scale model but are being applied at a state level in this context. The original fixed carbon ratio inputs were also not created for state-level use and were part of a lab-scale experiment (Freed and Mintz 2003). Running USFS HWP-C v1 at state scales (rather than national forest scales) was a more recent focus of the application of this model (e.g., CA USFS HWP-C in Loeffler et al. [2019] and Morgan et al. [2021]; HWP-C vR in Groom and Tase [2022]).

For this limited analysis, we ran the HWP-C vR model using US EPA WARM v-15 fixed carbon ratios in place of the fixed ratios in Smith et al. (2006)'s table D5. Because HWP-C vR applies one fixed carbon ratio to all paper and one fixed ratio to all solid wood in landfills, we applied US EPA WARM v-15 estimates for various product types to those same two categories. For example, the reference fixed carbon ratio for paper (0.44), was replaced with US EPA WARM v-15 fixed carbon ratio for newspaper (0.84). This was repeated for the other three paper product category estimates (corrugated containers, office paper, and coated paper) while holding the reference fixed carbon ratio for wood (0.77) constant. These same steps were then repeated holding the reference paper fixed carbon ratio (0.44) constant, while US EPA WARM v-15 estimates for dimensional lumber, medium-density fiberboard, and wood flooring replaced the original solid wood fixed

Table A4-2—Carbon outputs in 2018 from the HWP-C vR model using Oregon harvest inputs from 1908–2018

Model run	Product type	Landfill fixed carbon ratio	SWDS carbon storage	Difference from baseline[a]	Emit without energy	Difference from baseline[a]
			MMTC	PERCENT	MMTC	PERCENT
1	Paper	0.44	149.56	Reference	308.22	Reference
1	Solid wood	0.77	149.56	Reference	308.22	Reference
2	Newspaper	0.84	164.35	9	293.44	–5
2	Solid wood	0.77	164.35	9	293.44	–5
3	Corrugated containers	0.55	153.62	3	304.16	–1
3	Solid wood	0.77	153.62	3	304.16	–1
4	Office paper	0.12	137.73	–9	320.06	4
4	Solid wood	0.77	137.73	–9	320.06	4
5	Coated paper	0.74	160.65	7	297.13	–4
5	Solid wood	0.77	160.65	7	297.13	–4
6	Dimensional lumber	0.88	154.31	3	303.47	–2
6	Paper	0.44	154.31	3	303.47	–2
7	MDF	0.84	152.58	2	305.20	–1
7	Paper	0.44	152.58	2	305.20	–1
8	Wood flooring	0.95	157.33	5	155.56	–3
8	Paper	0.44	157.33	5	155.56	–3
9	Average[b] fixed carbon ratio for all paper	0.56	201.09	6	298.51	–3
9	Average[b] fixed carbon ratio for all solid wood	0.89	201.09	6	298.51	–3

[a] Difference is rounded to the nearest percentage.

[b] Averages refer to the average of all paper or all wood fixed carbon ratios from EPA WARM v-15.

SWDS = solid waste disposal site; MMTC = million metric tons of carbon; MDF = medium density fiberboard.

carbon ratio. We then compared model outputs to outputs with current parameters used for Oregon (referred to as "reference" in table A4-2).

Table A4-2 displays the outputs from altering fixed carbon ratios in landfills in 2018 using Oregon data. Carbon in products in use and carbon emitted with energy capture were omitted because the outputs did not differ from outputs with reference parameters (i.e., no change with altered fixed carbon ratios). Table A4-2 also displays the difference (percentage of change) between altered fixed carbon ratios and reference fixed carbon ratios. Finally, new averages of paper (0.56) and wood (0.89) fixed carbon ratios from US EPA WARM v-15 were run through the model.

The most notable difference between reference SWDS carbon outputs and outputs with US EPA WARM v-15 fixed carbon ratios were within paper categories. Changing the fixed carbon ratio for paper from 0.44 (reference) to 0.84 (newspaper) and coated paper (0.74) increased SWDS carbon storage by 7–9 percent and decreased emissions without energy capture by 4–5 percent. Conversely, changing paper's fixed carbon ratio to 0.12 (office paper) decreased SWDS carbon storage by 9 percent and increased emissions without energy capture by 4 percent.

Compared to paper, solid wood SWDS carbon storage was more stable when fixed carbon ratios were altered. Changing fixed carbon ratios for solid wood from 0.77 (reference) to 0.88, 0.84, and 0.95 marginally increased SWDS carbon by 2–5 percent. Emissions without energy capture decreased by 1–3 percent. Averaging US EPA WARM v-15 fixed carbon ratios resulted in increasing SWDS carbon storage by 6 percent and decreasing emissions without energy capture by 3 percent.

Monte Carlo uncertainty analyses have been performed for USFS HWP-C v1 and its variants, which include a generic range of uncertainty of ±15 percent for landfill fixed carbon ratios for wood and paper. However, we did not perform an uncertainty analysis for this limited sensitivity analysis examining fixed carbon ratios in SWDS. This brief analysis is meant only to serve as an example of the level of detail in HWP carbon estimation models like USFS HWP-C v1 and its variants that could still be improved and updated with more recently published data sources, such as US EPA WARM v-15.

A full sensitivity analysis has yet to be completed for USFS HWP-C v1 and its variants (Loeffler et al. 2014). The lead author is currently updating HWP-C vR to include product-specific fixed carbon ratios for discarded products. The differences in lignin content among products and the turnover rate of specific products likely contributed to the differences in outputs in table A4-2. Product-specific categories for the fixed carbon ratio of paper could be more influential on landfill carbon estimates than solid wood because paper products have a higher turnover rate, short half-lives, and widely varying fixed carbon ratios for specific products. Differences in solid wood fixed carbon ratios influenced landfill carbon storage and losses but to a lesser degree likely because solid wood products have slower rates of decomposition and longer half-lives.

There are many caveats to this brief analysis, but it has demonstrated that there are more recent estimates that could be applied to the landfill fixed carbon ratios used in U.S. HWP carbon estimation models. Using more recent product-specific fixed carbon ratio estimates from US EPA WARM v-15 model outputs may improve landfill parameters.

There are limitations to testing the sensitivity of landfills as described above; swapping out the old estimates with product-specific estimates has the same caveat as using the older estimates from Barlaz (1998)—GHG and global warming potential vary between wood and paper products. There are limited product-specific data for landfills, and more empirical data are needed at the state level to improve fixed carbon ratios, half-lives, and disposition ratios for individual products. Using US EPA WARM v-15 fixed carbon ratios for the broad categories of solid wood and paper is nonetheless useful because it demonstrates the sensitivity of landfill carbon estimates. More research is also needed to estimate CH_4 emissions associated with landfilled products.

Glossary

BARK EXPANSION FACTOR—A bark factor is the ratio of inside bark diameter to outside bark diameter. Although the bark factor can vary, the measurement is usually assumed constant at diameter at breast height. A bark expansion factor is a method of incorporating an estimation for bark removed from a tree (typically to incorporate into a calculation or a model).

BIOGENIC CARBON—Carbon that is stored (during carbon sequestration) or released (during decay) from biological materials such as plant material or soil. Biogenic carbon storage and release are estimated and then reported for the forest sector, including carbon stored in harvested wood products.

CARBON FOOTPRINT—Expressed as carbon dioxide equivalent (CO_2e), this represents the total greenhouse gas (GHG) emissions caused by an organization, structure, material, or individual etc.

CASCADED UTILIZATION—Using harvested wood products destined for recycling or discard pathways as raw material inputs instead of virgin material.

COMPUTATIONAL GENERAL EQUILIBRIUM—Economic models that can capture the potential direct and indirect market interactions and feedback effects across all economic sectors of a change in an economy.

CRADLE-TO-GATE—A term in reference to a life-cycle assessment, which is an analysis of raw material extraction through factory output (see "life-cycle assessment").

CRADLE-TO-GRAVE—A term in reference to an attributional life-cycle assessment, which is an analysis of raw material extraction to waste disposal.

DISCARD DISPOSITION RATIOS—After harvested wood products are discarded, these ratios allocate carbon to various discard pathways.

DISCARDED PRODUCT DECAY HALF-LIVES—The number of years it takes for half of the amount of discarded material to decay. These half-lives are applied to the degradable fraction of carbon in solid wood and paper products disposed of in solid waste disposal sites (dumps and landfills) to determine the rate at which carbon is emitted to the atmosphere from these solid waste disposal sites.

EMBODIED EMISSIONS—Fossil fuel emissions associated with harvest, transport, and manufacturing of harvested wood products.

END-USE PRODUCT RATIOS—These ratios are applied to primary product categories (e.g., lumber, panels, paper, and paperboard) to determine the amount of harvested material going to end-use products.

END-USE PRODUCTS—Finished harvest wood products; for example, new single-family or multifamily housing, furniture, new manufactured housing, residential repair products, new nonresidential buildings, and other construction products.

EQUILIBRIUM STATE—The amount of time needed for a forest to sequester carbon removed during harvest or achieve parity with a baseline.

FOREST SECTOR—Forest ecosystems and the carbon removed from them.

HARVESTED WOOD PRODUCT (HWP)—Sometimes also referred to as "forest product." This synthesis uses the IPCC (2006a) term "harvested wood product," to capture all forest biomass, including bark, that is removed from forests for use or consumption.

LANDFILL FIXED CARBON RATIOS—These ratios establish the proportion of harvested wood product carbon that is not subject to decay (also referred to as nondegradable, or fixed, carbon).

LEAKAGE—Occurs when a policy or management activity leads to an unintentional shift in forest use, degradation, or reversal of intended forest management (IPCC 2007); for example, forest conservation in one location leading to forest harvest in another location to meet timber supply demand.

LIFE-CYCLE ASSESSMENT (LCA)—Also known as a life-cycle analysis, this is used to gain a more complete picture of the environmental impacts, climate costs, benefits, and mitigation potential of any product or process, including harvested wood products. They extend beyond the forest-generated ("biogenic") carbon to evaluate the energy, materials, and environmental impacts needed to produce and use wood products for every stage along the supply chain, including forest regeneration and management, harvest, transportation, manufacturing, use, maintenance, reuse and recycling, and final disposal.

MILL RESIDUES—The proportion of harvest volume or sawlogs (bark is excluded from sawlogs) that are removed from the forest but not directly utilized for solid wood products.

MILLING EFFICIENCY RATIO—The portion of roundwood (used for harvested wood commodities or primary products) that becomes a product, with the remainder being mill residues.

PARTIAL EQUILIBRIUM MODELS—A class of economic models that provide the detailed description of the effects of a given policy, technology, or other external factors on an economic sector that is directly affected (e.g., forests and the forest products sector). Many forest sector economic models are partial equilibrium models and assume that forest sector is too small to affect other sectors of the economy.

PULPWOOD—Industrial and nonindustrial harvested roundwood for processing or manufacturing for wood pulp products. This product category can sometimes include bolts and chips if it is mixed with fiber byproducts.

PRIMARY PRODUCT CLASSES—This class of harvested wood products includes lumber, plywood, panels, wood pulp, and fuelwood; also referred to as "semi-finished" products.

PRIMARY PRODUCT RATIOS—These ratios are applied to timber product classes to determine the primary product classes.

PRODUCT HALF-LIVES—The number of years it takes for half of the amount of harvested wood product in use to be discarded. These half-lives are applied to the products in use carbon pools to determine the annual rate and amount of carbon that is discarded.

PRODUCTS IN USE—Harvested wood products that are currently functional as a finished product and not yet discarded.

ROUNDWOOD—Harvested wood portions of trees (e.g., logs) used for commodities or primary products that become products such as sawnwood, wood-based panels, and paper or paperboard; does not include bark (UNFCCC 2003).

SOLID WASTE DISPOSAL SITE (SWDS)—One term to describe two discard pathways, dumps and landfills, both of which continue storing harvested wood product carbon after those products have been discarded. A landfill is a regulated, covered waste collection facility, whereas a dump is uncovered, excavated area where waste is deposited. Dumps are technically no longer legal in the United States, but were historically used for waste collection.

SPILLAGE—Similar to leakage, and sometimes referred to as "spillover," these are unintended consequences of changing markets, policy, or management decision that lead to intensified forest management or additional carbon stocks.

SUBSTITUTION FACTOR (SF)—A measure of avoided or increased greenhouse gas emissions when a wood product is used instead of another building material to provide the same function.

SYSTEM BOUNDARY—Describes which components of a model (or system) are included and excluded.

TIMBER PRODUCT CLASSES—Harvested wood product classes, such as sawlogs, nonsawtimber, pulpwood, or fuelwood.

TIMBER PRODUCT RATIOS—Ratios based on, and in some cases extrapolated from, USDA Forest Service data or state-level utilization data.

UNDERBARK VOLUME—Bark is excluded because the starting point for harvested wood product carbon estimation is commercial harvest volume once bark has been removed.

WOOD ENERGY SUBSTITUTION—Residues, discarded harvested wood products, and captured emissions from landfills are burned for energy, thereby avoiding burning fossil fuel emissions for energy production.

WOOD PULP—Products created from processing pulpwood, such as paper and paperboard products.

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