Carbon Mitigation Impacts of Increased Softwood Lumber and Structural Panel Use for Nonresidential Construction in the United States

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Abstract

More wood use in the United States to construct low-rise nonresidential (NR) buildings would increase consumption and production of softwood (SW) lumber, engineered wood products, and structural and nonstructural wood panels. Using a consequential life-cycle analysis, we estimated the change in net CO_2 emissions that would be caused by increased use of SW lumber and structural panels in NR construction. Carbon (C) storage and emissions were projected over 50 years for baseline and increased wood use scenarios using the US Forest Products Module operating within the Global Forest Products Model (USFPM/GFPM) and the Southern region timber supply model (SRTS). Increased wood use in NR construction (C content of 428 million tons of carbon dioxide equivalent [tCO_2e]) could provide an emissions reduction of 870 million tCO_2e over 50 years or a net emissions reduction of 2.03 tCO_2e/tCO_2e of extra wood used in NR buildings over 50 years. The CO_2 savings varied for products provided in the South, North, and West because of differences in biological timber regrowth; market-induced changes in land use; differences in timber harvests, lumber, and structural panel production; and associated differences in C stored in forests, harvested wood products, logging slash, and manufacturing emissions. The US South provided the largest net change, –2.83 tCO_2e/tCO_2e of extra wood products, followed by the North and West with –1.89 and –0.60 tCO_2e/tCO_2e of extra wood, respectively. These results suggest strategies that result in increased use of wood in place of nonwood products in NR buildings would be effective in mitigating CO_2 emissions.

Forests and the wood products sector have important roles in greenhouse gas (GHG) mitigation by capturing atmospheric carbon dioxide (CO_2) during tree growth and substituting harvested wood products for more GHG-intensive nonwood building materials such as steel and concrete. A recent study found that there is a potential to increase the use of wood products in nonresidential (NR) construction (Adair et al. 2013). These volumes represent the incremental amounts of wood that could have been used in 2011 if 2011 had been a year with construction at historical average levels (Adair et al. 2013). Such increased demand could generate a carbon (C) mitigation benefit by substituting wood materials for nonwood materials such as steel and concrete, which emit more CO_2 during their manufacture. At the same time, increased wood demand would increase product prices, leading to increased timber prices that would encourage investment in tree plantations and intensified forest management that would increase both C sequestered in forests and C stored in harvested wood products (HWP). The objective of this study was to estimate the degree to which a projected expansion in US demand for softwood (SW) lumber and structural panels for construction of low-rise NR buildings would change C storage and C emissions. The study estimated change in C stored in forests, in logging slash left to decay in forests, and in HWP

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and also change in C emissions by substituting wood products manufacturing for nonwood products manufacturing. To make projections, this study integrated results from two forest sector models: the US Forest Products Module operating within the Global Forest Products Model (USPFM/GFPM; Buongiorno et al. 2003, Ince et al. 2011) and the Southern Region Timber Supply Model (SRTS; Abt et al. 2009), along with results from two other models that track C stored in HWP (Woodcarb II; Skog 2008) and C stored in logging slash left to decay in forest sites (a spreadsheet model, Nepal et al. 2014). The study used a consequential life-cycle analysis (CLCA) framework to estimate the degree to which increased demand for SW lumber and structural panels will change net CO_2 emissions. The processes and emissions within the system boundaries (Table 1) for the CLCA include the following:

1. processes and emissions to make wood and replacement nonwood products, including wood harvest and hauling,
2. forest growth and decay processes that result in net C sequestration and include the generation and decay of logging slash after harvesting,
3. processes to use and store C in HWP in long- and short-lived uses and disposal to and storage in landfills, and
4. market processes that stimulate investment in forests in the form of planting new forests.

The emissions from landfills in the form of CO_2 are implicitly included as the wood and paper C stocks in landfills decrease. The emissions from landfills in the form of CH_4 and any fossil emissions offset by burning biogenic methane generated from decaying wood and paper are considered part of an evaluation of landfill management systems and are not included. The emissions from burning a portion of wood discarded after its use life are implicitly included in our estimates because they do not appear as part of the wood and paper C deposited in landfills. Whereas the previous description gives our intended system boundary, caveats are discussed below because we use an estimate of average net reduction in manufacturing emissions when wood products displace nonwood products based on 21 studies reviewed by Sathre and O'Connor (2010) that have varying system boundaries.

Table 1.—Carbon (C) stock changes and emissions included/not included when estimating changes in net CO_2 emissions owing to increased wood use for nonresidential construction.

Category	Included/not included
C stock	
Forest C aboveground and belowground	Included
Logging slash C	Included
Wood products C from domestic harvest (products in use and products in landfills)	Included
Wood products C from exports (products in use and products in landfills)	Included
Wood products C from imports (products in use and products in landfills)	Not included
Forest soil carbon	Not included
C emissions	
From fossil energy use in manufacturing wood products	Included
From wood energy in manufacturing wood products	Included
From fossil energy use in manufacturing nonwood material (steel and concrete)	Included
From harvesting operations, planting and hauling to mills (e.g., fossil fuel emissions)	Included
From wood and paper decay in landfills in the form of CH_4	Not included

The final outcome of this study is estimates of net CO_2 emissions reduction per ton of CO_2 equivalent (tCO_2e) of additional wood products that are used in NR construction compared with a base case without increased NR wood use, at various times over a period of 50 years.

NR buildings can be divided into those with six or fewer stories (low-rise), and those with seven or more stories (high-rise). Low-rise NR buildings were the focus of this study because they hold near-term potential for increasing wood use. Adair et al. (2013) estimated that in 2011, low-rise buildings accounted for 72 percent of all nonresidential buildings. That study estimated the potential for using more wood in NR buildings by building type, geographical area, and wood product. The estimated increase includes wood that could be used (under International Building Code standards) if (1) all major building applications (framing, columns, stairs, siding, soffit, fascia, exterior trim, sheathing, and underlayment) in concrete- or steel-framed buildings were built with wood, and (2) all major building applications in wood-framed buildings that are not using wood were converted to wood. A substantial portion of the increase would come from shifting to wood wall framing that in 2011 was used in only 12 percent of NR low-rise buildings. The largest use of wood wall framing was in hotels and religious buildings (54% and 34% use, respectively) and the lowest was for stores, industrial buildings, and public buildings (7% use each). Stores, schools, public buildings, and offices have the most potential for increased wood use (a combined 74% of the total increase), whereas increased use in industrial buildings is limited by building codes and is negligible.

Although environmental benefits of using wood in construction have been studied (e.g., Buchanan and Levine 1999, Börjesson and Gustavsson 2000, Lippke et al. 2004, Perez-Garcia et al. 2005, Gustavsson et al. 2006, Upton et al. 2008, Sathre and O'Connor 2010, Bergman et al. 2014), there has not been evaluation of use of wood in place of nonwood materials in construction that was done by using nationwide forest carbon and forest sector models. Examining CO_2 mitigation benefits of increased wood use in NR construction is important in the United States because there is little opportunity to increase wood use in residential construction. The US residential sector already provides about 90 percent of the C and energy benefits that could be provided by use of wood materials in US construction (Upton et al. 2008). Up to now, the use of wood framing in NR construction has been limited for higher story buildings because of perceived building code restrictions. However, a reevaluation of the International Building Code suggests that many more five- or six-story buildings could use wood framing with only minor design changes (Adair et al. 2013). As a result, more wood could be used for low-rise NR building construction in the United States (Adair et al. 2013). Adair et al. indicate a possibility of using an additional 5,369 thousand m^3 of SW lumber and 2,338 thousand m^3 of structural panels (a total of 7,707 thousand m^3 of wood) in construction of low-rise NR buildings in the United States. Wood-framed buildings up to eight stories are already becoming more common in Scandinavia (Mahapatra and Gustavsson 2009).

For this analysis, we use a time dynamic CLCA, where we explicitly track change in C fluxes over time caused by a change in production and use of wood products. Some analyses implicitly estimate the effect of a change in wood use and production where all changes in C fluxes over an indefinite time period are added together to get one net change in C flux. Such net estimates made over an indefinite time period may give the impression that CO_2 emissions from wood energy used in making wood products are offset immediately by forest regrowth on land. However, many studies have shown that regrowth may take a few to many decades to recover forest C (e.g., Zanchi et al. 2012, Agostini et al. 2013). Therefore, the short-term net GHG emissions from wood use can be different from the long-term net emissions. This study estimated the difference, over time, in actual forest sector C loss and recovery between the baseline and high (higher) wood use scenarios over a period of 50 years. An increased harvest to meet increased wood demand can change forest sector C sequestration not only from harvest and changes in regrowth but also from changes in forest investments (e.g., increased plantation area). Such investment effects leading to increased C sequestration have been simulated by several studies (e.g., Abt et al. 2012, Daigneault et al. 2012, Nepal et al. 2014). This study incorporated both biological and market outcomes of increased wood use in evaluating net CO_2 mitigation effects.

Methods

The study used a CLCA framework to evaluate net CO_2 mitigation benefits of projected increase in use of SW lumber and structural panels (SW plywood/veneer and oriented strand board [OSB]) for US NR construction. The information needed for the CLCA was provided by 50-year projections from four different models of the US forest sector including the USFPM/GFPM (Buongiorno et al. 2003, Ince et al. 2011), the SRTS (Abt et al. 2009), Woodcarb II (Skog 2008), and a spreadsheet model of logging slash accumulation and decay (Nepal et al. 2014). The estimates of increased wood use derived from Adair et al. (2013) are used as inputs to the forest sector models. In this section we describe the models and their use in this study.

Economic and biological models and inputs

The USFPM is a partial equilibrium model of US timber markets that operates within a broader Global Forest Products Model (GFPM). The USFPM/GFPM provides forecasts of forest product and timber market equilibrium prices and quantities and includes timber inventory, timber harvests, forest product production, consumption, and net trade. The estimated market equilibrium timber harvest quantities and prices for a given year in the USFPM/GFPM are ones that maximize total producer and consumer surplus of the entire forest sector, based on the optimization approach of Samuelson (1947) to regional market modeling (Buongiorno et al. 2003, Ince et al. 2011). The USFPM/GFPM models supply, over time, for four categories of timber (sawtimber and nonsawtimber by SW and HW species group) by three US regions (North, South, and West) as a function of timber price and species group inventory using a Cobb-Douglas functional form and associated price and inventory elasticities.

The USFPM/GFPM relies on exogenous projections of macroeconomic data as demand or supply drivers such as US gross domestic products (GDP), US housing starts, currency exchange rates, and US timberland area. The national demands for primary products (e.g., SW lumber, plywood, paperboard) are modeled as a function of product price, GDP, and other drivers using a Cobb-Douglas functional form. Manufacturing costs are assigned to convert timber to timber products and converting timber products to primary wood products. Demand for timber is derived from national demands for primary forest products (lumber, panels, paper, paperboard) that use timber products (sawlogs, pulpwood) as input. To provide the supply of sawlogs and pulpwood, the supply of timber (sawtimber, nonsawtimber) is converted by region (North, South, and West) into amounts of sawlogs, pulpwood, and fuelwood that are used for lumber, panels, pulp, and energy. More details about the USFPM/GFPM model can be found in Ince et al. (2011) and Buongiorno et al. (2003). For example, the key parameters used in the study such as price elasticities of wood product demand, timber supply, and input–output coefficients can be found in tables 18 and 19, 7, and 14, respectively, in Ince et al. (2011). Similarly, the parameters of forest growth response to growing-stock density can be found in table 3 in Nepal et al. (2012b), and the logging slash decay factors are reported in table S2 in Nepal et al. (2014).

The Southern regional timber supply model (SRTS) uses an exogenously specified timber demand trajectory to drive changes in southern timber inventory and timber markets (Abt et al. 2009). It uses the USDA Forest Service forest inventory and analysis (FIA) data (USDA Forest Service 2014) on timber inventory, growth, removals, and area by forest type, private ownership category, species group, and age class for FIA survey units (multicounty areas). A constant elasticity supply function with specified price and inventory elasticities is used to equilibrate each year with either an exogenous demand shift (demand mode) or a harvest quantity (harvest mode). For this analysis, we used harvest mode with the harvest as specified in USFPM. The model then uses a goal program to assign timber harvest quantities to owners and management types based on ''last year.'' The harvest is then passed to the biological accounting module to update available inventory for the next period's equilibrium calculation.

Timberland area is projected as a function of pine sawtimber and nonsawtimber price, agricultural rents, and county population forecasts based on an econometric model estimated by Hardie et al. (2000). The current study assumes a constant agriculture land rent with the loss of rural land to urbanization based on county-level population forecasts by Prestemon and Abt (2002). Total timberland area increases in response to an increase in SW sawtimber and nonsawtimber price. This increased area is allocated to pine plantation, natural pine, mixed (natural pine, mixed pine, upland hardwoods), and lowland hardwoods by assuming that pine plantations are three times as responsive as natural pine and mixed, and lowland hardwoods are assumed half as responsive as natural pine and mixed. Regressions for pine plantations are based on quadratic age equations that are estimated separately by ownership category, subregion (southeast vs. south central), and physiographic region with covariates for state. Natural types are estimated based on quadratic age regressions estimated separately by physiographic and subregion with covariates for ownership and state. A growth calibration is applied by shifting the estimated equation so that the mean of the equation values matches the mean of the observed values in the survey unit. For further details on the SRTS model see Abt et al. (2009).

In USFPM/GFPM sawtimber is defined as the growing-stock portion of trees above 27.94 cm (11 in.) in diameter at breast height (dbh) for hardwoods and trees above 22.86 cm (9 in.) dbh for softwoods. Nonsawtimber is defined as trees between 12.70 and 22.86 cm (5 to 9 in.) dbh for the softwoods and between 12.70 and 27.94 cm (5 to 11 in.) dbh for hardwoods and includes all of the non–growing-stock portion of both poletimber and sawtimber trees. For consistency, the timber product definition files required to run the SRTS model were modified according to the USFPM/GFPM timber merchantability specifications.

The WOODCARB II model (Skog 2008) was used to obtain estimates of C stored in HWP that use USFPM/GFPM projected production, supply, and trade quantities of various wood and paper products. In addition, a spreadsheet model of logging slash decay was used to project changes in logging slash C that are based on first-order exponential decay and assume half-life of logging slash of 16.5 years (Nepal et al. 2014).

The SRTS and the USFPM/GFPM models are used to make projections through an iterative procedure described in ''Integrating the USPFM/GFPM and the SRTS model.''

Scenarios

The study used two scenarios, referred to as the baseline and high wood use. For both scenarios, US consumption of forest products is driven by projected growth trend in US real GDP and other variables, such as recent historical growth rates for advertising expenditures in print media and electronic media that shift demands for newsprint and printing and writing paper (Ince et al. 2011). Consumption of structural lumber and wood panels is driven by projected growth in US real GDP and US single-family housing starts. The consumption of forest products is also driven by projected growth in global currency exchange rates. Both scenarios used the 2012 USDA Economic Research Service (ERS) global projections for GDP, currency exchange rates for all countries to 2030 (USDA ERS 2012), and a revised US housing projection (Ince and Nepal 2012). The GDP growth rate was extended to 2060 by continuation of projected average growth rates (about 2.5% per year for the United States), and the exchange rates were held constant from 2030 to 2060. Both scenarios assume a rebound in housing, with average single-family housing starts increasing to the long-run historical trendline at around 1.1 million per year by 2020, and then following a slowly increasing trendline to 2060 (Ince and Nepal 2012).

For the high wood use scenario, demands for US SW lumber and structural panels (SW plywood/veneer and OSB) were increased from the baseline scenario level until a targeted degree of increase in wood use in NR construction was achieved. Note that OSB largely uses SW in the South and HW in the North. Adair et al. (2013) estimate a possibility of using an additional 5,369 thousand m^3 of SW lumber and 2,338 thousand m^3 of structural panels (a total of 7,707 thousand m^3 of wood) for construction of low-rise NR buildings in the United States. Accordingly, for the high wood use scenario, the total SW lumber and structural panel demands were increased iteratively by shifting the demand curves in the baseline scenario until the periodic (5 yr) projections of cumulative consumption of SW lumber or panels in NR uses matched the target (increased) NR consumption within about ± 10 percent. The target increases in product demand were assumed to occur between years 2010 to 2015 and remain at the increased level through 2060.

Because the USFPM/GFPM does not distinguish between consumption for residential and NR construction, separate demand equations were estimated for SW lumber, OSB, and SW plywood/veneer, and a side calculation was done to calculate the portion of total SW lumber and structural panel demands that would be used for NR construction based on estimated price elasticity of NR demand. The following equation indicates how the NR consumption (qNR) in period 1 is estimated based on total consumption (qT) in period 1, NR consumption in period 0, the price shift (e.g., for SW lumber or structural panels) between period 0 and period 1, the demand elasticities for total demand (ε_T) and NR demand (ε_{NR}), an exogenous shift (s) to increase total demand (compared with the baseline scenario), endogenous shifts in total demand from GDP (A), and shifts caused by both GDP and housing (B):

$$q_1NR = \left[q_0NR \left(\frac{q_1T^{**}}{q_0T} \right) \frac{A}{B} + \left(q_1T \left(\frac{1}{\left(\frac{p_1}{p_0}\right)^{\varepsilon_T}} \right) \left(\frac{s}{1+s} \right) \frac{1}{\left(\frac{p_1}{p_0}\right)^{\varepsilon_{NR}}} \right) \right] \tag{1}$$

where

$q_tT =$ total demand for SW lumber or structural panels at time t,

$q_tNR =$ SW lumber or structural panel demand for NR construction at time t,

$\varepsilon =$ elasticity with respect to price for total or NR demand,

$p_t =$ price of SW lumber or structural panel at time t,

$A =$ fractional shift in total demand for SW lumber or structural panel because of GDP only between $t = 0$ and $t = 1$,

$B =$ fractional shift in total demand for SW lumber or structural panel because of GDP plus housing between $t = 0$ and $t = 1$, and

$s =$ exogenous shift (in %, > 0) to cause target net increase in SW lumber and structural panel consumption in NR construction.

Equation 1 uses two values obtained from Equations 2 and 3 below, based on total consumption in period 1.

Equation 2 calculates period 1 total product consumption removing the effect of change in product price between periods 0 and 1.

$$q_1T^* = q_1T \left(\frac{1}{\left(\frac{p_1}{p_0}\right)^{\varepsilon_T}} \right) \tag{2}$$

Equation 3 calculates period 1 total product consumption, removing further the effect of the exogenous shift, *s,* applied to increase total demand.

$$q_1 T^{**} = q_1 T^{*} \left(\frac{1}{1+s} \right) \tag{3}$$

With these calculations, we break total SW lumber or structural panel demand (qT) into two categories: (1) demand for NR construction (qNR) and (2) demand for other construction ($qT - qNR$). When we exogenously shift total demand (qT) in the high wood use scenario, demand for NR wood (qNR) increases in the high wood use scenario but demand for other wood ($qT - qNR$) decreases when compared with the baseline scenario. It is logical for the consumed wood for other uses to decrease because the price for the products increases. Thus, the increase in NR wood use is greater than the increase in total wood use between the baseline scenario and the high wood use scenario (Fig. 1).

Estimates of net CO_2 savings

Table 1 shows the processes and emissions we included within the system boundary for this CLCA study. These include process and emissions to produce wood and replacement nonwood products, including wood harvest, forest growth processes, and attendant net C sequestration including generation and decay of logging slash after harvesting, processes to use and store C in wood products in long- and short-lived uses and in landfills, and market processes that stimulate investment in forests in the form of planting new forests (more plantation area). The C estimates for logging slash left to decay on forest sites represents the amounts that are left after recovery for wood energy use when it is economical. In both scenarios, the model uses logging slash to meet a portion of total US wood energy demand if it is economical to do so. The total wood energy demand is based on the econometric relationship between historical GDP growth and the fuelwood consumption (Simangunsong and Buongiorno 2001). Whether logging slash is recovered for energy depends on endogenously projected wood energy demand, fuel feedstock price, and assumed logging recovery costs (Ince et al. 2011). The study used a ''production accounting approach'' in estimating C stored in HWP (Intergovernmental Panel on Climate Change 2000). Under the production C accounting approach, C stored in wood and paper products that are produced within the country are tracked. C in exports is included, while C in imports is excluded. The study also included emissions from forestry operations such as planting and fertilizing, harvest and thinning, and transportation. These are our intended boundaries but there is some uncertainty because the estimate of net change in manufacturing emissions we use from Sathre and O'Connor (2010) is the average from 21 studies that vary in their system boundaries. For example, most, but not all, of the studies assumed wood energy emissions were offset by forest regrowth. Some studies included fossil emission offset from energy from wood by-products (in addition to fossil emission offsets from burning residue for energy at lumber and panel mills). All studies excluded C stored in products in end uses, but some included C stored long term in landfills with some offsetting storage of CO_2e for methane emissions. The studies that included storage in landfills include methane emissions that offset storage to a degree that limits double counting of C storage in landfills in our analysis. Given the type and mix of system boundaries used by studies included in the Sathre and O'Connor (2010) estimate, some causing overestimates and some causing underestimates, it is the judgment of the authors that the net change in manufacturing emissions we use could result in a slight overestimate of net emission reduction in our analysis.

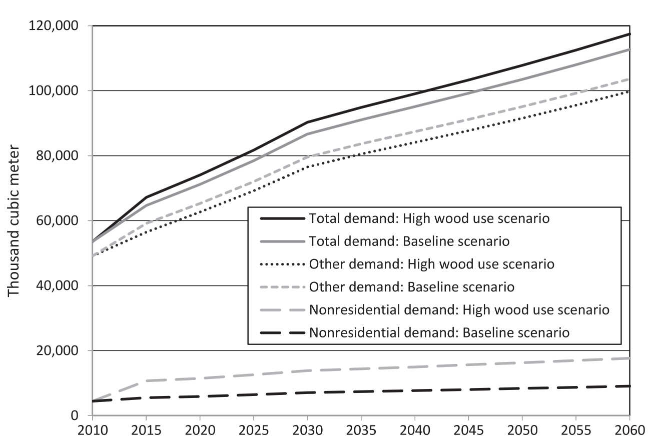

Figure 1.—Total use, nonresidential (NR) use, and other use (total minus nonresidential) of softwood lumber in baseline and high wood use scenarios, 2010 to 2060.

Net change in CO_2 emissions per unit of increased SW lumber and structural panel use was calculated based on the difference between the baseline and the high wood use scenarios in C sequestered in forests, in C stored in HWP, in C stored in logging slash left to decay in forests, and in manufacturing emissions resulting from displacement of nonwood materials by wood materials using the following equation:

$$\text{Net change in } CO_2\text{emissions} = \frac{\begin{array}{c}FC_{Base} - FC_{High} + HWPC_{Base} - HWPC_{High} \\ + LSC_{Base} - LSC_{High} + ME_{High} - ME_{Base}\end{array}}{WC_{High} - WC_{Base}} \tag{4}$$

where

FC = C stored in forests (million tCO_2e),

HWPC = C stored in harvested wood products (million tCO_2e),

LSC = C stored in logging slash left in forests (million tCO_2e),

ME = manufacturing emissions because of production of wood and nonwood materials (million tCO_2e),

WC = C equivalent of SW lumber and structural panel production (million tCO_2e),

Base = the baseline scenario with no increased wood use for NR construction,

High = the high wood use scenario with increased wood use for NR construction.

For each scenario, C stored in forests was calculated by SRTS for the South and by USFPM/GFPM for the North and the West. The projected timber growing-stock inventory was converted to total tree biomass weight using regional average ratios of total dry weight biomass to timber growing-stock volume (Nepal et al. 2012a) using data from the FIA database on dry weight of tree biomass and growing-stock inventory volume on US timberland (USDA Forest Service 2014). The estimated forest C pool includes C stored in tree boles, tops and limbs, saplings, stump, bark, and coarse-root biomass of all live and standing dead trees above 2.5 cm dbh, but it does not include foliage biomass, as defined by FIA (USDA Forest Service 2014). The estimated dry weight of biomass weight was multiplied by 0.5 to estimate C weight and then multiplied by 44/12 (ratio of molecular weights of CO_2 and C) to convert to the CO_2 equivalent. In the South, timber-price–induced changes in forest land area and forest types were accounted for when calculating C stored in forests. Consistent with historical changes, we did not expect forest land area or types to change over time in the West or North. C stored in HWP was estimated using the Woodcarb II model, where the inputs were projected wood product production and trade from USFPM/GFPM. C stored in logging slash left to decay at harvest sites, after any removals for wood energy use, was estimated using an exponential decay function with a half-life of 16.5 years (equal to an annual decay rate of about 4%; Nepal et al. 2014).

A manufacturing emissions displacement factor (DF) for substitution of wood for nonwood building material is based on estimates from Sathre and O'Connor (2010). A DF of wood product substitution is an index that quantifies the amount of avoided GHG emissions resulting from the use of wood instead of nonwood materials in given construction applications (Sathre and O'Connor 2010). It is derived as the difference in emissions resulting from the use of nonwood material and use of wood material for the same application, divided by the amount of wood use. Emissions are compared for processes from raw material ''cradle'' to mill ''gate'' and include emissions from acquisition of raw materials (mining or forest operations and management), transportation, and processing into useable products. Processes included vary somewhat among studies. They can include energy needed to make additives such as adhesives used in composite wood products. It is expressed as a difference in emissions per unit of wood used. A negative DF means that GHG emissions are avoided per unit of wood use, and a positive DF means that emissions are greater per unit of wood use.

Expressed as CO_2 per unit of CO_2 equivalent (CO_2e) in wood products, Sathre and O'Connor (2010) estimated an average DF of –2.1 tCO_2e/tCO_2e based on the 21 studies they examined. Most factors for the 21 studies are in the range of –1.0 to –3.0 tCO_2e/tCO_2e. Most of the studies used to develop this average, however, excluded wood energy emissions for wood products that are emitted during manufacturing. We made an adjustment and reduced the average DF of –2.1 tCO_2e/tCO_2e to –1.68 tCO_2e/tCO_2e by including wood energy emissions used to make wood products. To adjust the DF value, we estimated the average wood energy emissions (0.42 tCO_2e/tCO_2e) generated during the production of a range of timber products in the United States as reported in Bergman et al. (2014) and added this average value from the original DF value of –2.1 reported in Sathre and O'Connor (2010). We needed to include the wood energy emissions because those emissions are only offset over time in our framework with the regrowth of forests that provided the wood used for energy. C content of SW lumber and structural panel production was estimated based on a rough conversion factor of 481 kg dry weight biomass per m^3 (30 lb/ft^3).

Integrating the USPFM/GFPM and the SRTS model

USFPM/GFPM and SRTS each provided important projection outputs for the baseline and high wood use scenarios. We used SRTS to model Southern forest inventory as driven by wood harvest projected by USFPM/GFPM. Specifically, SRTS provided projections of Southern forest inventory and forest land area changes, while USFPM/GFPM provided projections of (1) national and regional forest product production, consumption, and net trade, (2) regional timber harvest and logging slash generation and use, and (3) timber inventory for the North and West. These projections were used to estimate C stored in forests, logging slash, and HWP. The change in projected wood use from USFPM/GFPM was used to estimate change in wood use for NR construction between scenarios and the change in manufacturing emissions as wood was substituted for nonwood materials.

Iterative procedures were used to integrate projections from USPFM/GFPM and SRTS. The goal of the first iterative procedure was to have SRTS projections of timber prices for the South match timber price projections from USFPM/GFPM. The first step was to use the baseline USFPM/GFPM 50-year timber harvest for the South (for HW and SW, sawtimber and nonsawtimber) to drive an SRTS projection for 50 years. Projected timber prices from SRTS were compared with those from USFPM. Adjustments were then made to the (1) SRTS timber supply price elasticities and (2) SRTS cull factors, which indicate what proportion of HW and SW sawtimber qualifies as pulpwood. Then, SRTS is rerun using the same harvest driver as before. Additional SRTS adjustments and runs are done until SRTS timber price projections match projected prices from USFPM. We assume USFPM/GFPM is providing correct timber price projections given national and global market drivers. We need SRTS prices to match in order to get correct SRTS land use investment (between forest and agriculture and between forest management types). To get matching timber price projections we used timber supply price elasticities of 0.5, 1.1, 0.7, and 0.7 and cull factors of 1.0, 0.48, 1.0, and 0.22 in SRTS for softwood nonsawtimber, softwood sawtimber, hardwood nonsawtimber, and hardwood sawtimber, respectively. These SRTS elasticities and cull factors were used for both the baseline and the high wood use scenario projections.

The second iterative procedure matches USFPM/GFPM harvest and inventory for the South to SRTS harvest and inventory. To develop a match, 50-year projections of timber harvest from USFPM/GFPM were used as exogenous drivers in SRTS runs, and the resulting timber inventory (for the four timber categories) from SRTS (added together to get one inventory per species group) is used in the subsequent run of USFPM/GFPM as a shifter in the timber supply curves. Each timber supply curve has an elasticity with respect to inventory of 1.0. This iterative procedure was continued until the projected timber harvest quantities from the USPFM/GFPM and the southern timber inventory quantities from SRTS did not change. At this point, the two models were considered to have converged. Such iterative procedures have been used to obtain convergent solutions for other forest sector models where submodels exchange projections and converge to stable solutions (e.g., Haynes et al. 2006).

Results

Table 2 presents regionwide and nationwide net change in CO_2 emissions per ton of CO_2e in additional SW lumber and structural panels that were used in NR construction over a period of 50 years between the baseline and high wood use scenarios. The analysis indicates a nationwide net change in CO_2 emissions of –2.03 by 2060, meaning that for each ton of CO_2e in extra wood products used in NR buildings over 50 years there is a net change of –2.03 tCO_2e. The estimated net CO_2 savings accrue from the differences between the baseline and the high wood use case in C sequestered in forests, HWP, logging slash left to decay in forests, and C equivalent of manufacturing emissions. For the 428 million tCO_2e difference in wood use for NR construction (cumulative over 50 yr), the largest change in CO_2 emissions (–870 million tCO_2e) comes from the reduction in manufacturing emissions from the displacement of nonwood materials by additional wood products, followed by extra C stored in HWP (–124 million tCO_2e), and extra C stored in logging slash left to decay in forests (–81 million tCO_2e; Table 2). The increased harvests needed to meet the increased SW lumber and structural panel demand drive down the nationwide timber inventory between 2010 and 2060, reducing C sequestered in forests between scenarios by 55 million tCO_2e (Table 2). However, this reduction in forest C is more than offset by the combined increase in C accumulation in HWP and logging slash and in the reduction in manufacturing emissions. The net C emission change (reduction) is –870 million tCO_2e (Table 2). This amount, divided by the cumulative C equivalent of additional wood products used for NR production by 2060 (428 million tCO_2e), results in nationwide net change in CO_2 emissions of –2.03 per ton of CO_2e in wood products over 50 years (Table 2).

Net CO_2e emission reductions vary by US region as a result of regional differences in timber harvests, SW lumber and structural panel production, and associated differences in C stored in forests, HWP, logging slash left to decay in forests, and manufacturing emissions. The regional variation in net CO_2e emission reduction also is due to differences in biological timber regrowth and market-induced changes in land use and forest management that influences total forest C. Note that the manufacturing emissions per unit of output do not vary by region or product, and the HWP C storage per unit of output does not vary by region.

The largest net CO_2e emissions reduction per unit of extra wood use occurs in the South (–2.83 tCO_2e/tCO_2e in wood), followed by the North (–1.89 tCO_2e/tCO_2e in wood), and the West (–0.60 tCO_2e/tCO_2e in wood; Table 2). The South provides the largest net CO_2 savings, and this region provides the largest increase in SW lumber and OSB between the baseline and the high wood use case. Nevertheless, the southern timber inventory increases in the high wood use case because of increased investment in pine plantations because of increased timber prices (Figs. 2a and 2b). The South experiences an increase in pine plantation area between the baseline and high wood use scenarios of 0.36 million hectares (0.90 million acres) and in upland hardwood area of about 0.16 million hectares (0.38 million acres), with an overall increase in forestland area of 0.60 million hectares (1.48 million acres) by 2060 (Figs. 3a and 3b). The South also saves the largest amount of additional C from C stored in HWP (–62 million tCO_2e), C stored in logging slash accumulation and decay (–39 million tCO_2e), and emissions savings from displacement of nonwood materials by wood (–414 tCO_2e; Table 2).

The North, with lower harvest, lower wood product production, and lower reduction in forest C (39 million tCO_2e; but no market-induced investment in forests) than the South and West, has a lower net change in CO_2e emissions (–1.89 tCO_2e/tCO_2e in wood; Table 2). The West harvests less than the South but more than the North and produces the largest amount of SW plywood/veneer. The resulting difference in forest C between the two scenarios is larger in the West (198 million tCO_2e) compared with the North and South, but still resulting in net change in CO_2 emissions of –0.60 tCO_2e/tCO_2e in wood (Table 2).

The West shows a greater depletion of inventory per unit of product than the North because modeled growth stimulation from a unit of timber harvest in the West is notably less than for the North. Inventory growth in the North and West is a function of many factors including timber density per acre (Nepal et al. 2012b). When timber is harvested, timber density declines, which results in an increase in forest growth per acre. The current average growth per acre in the West is less than for the North. For a 1 percent decrease in timber density there is an increase in softwood and hardwood growth of 0.5 percent for the North and an increase in softwood growth of 0.7 percent in the West. But the greater growth response in the West is from a

Table 2.—Cumulative differences in carbon in various carbon pools (million tCO_2e) between baseline and high wood use scenarios, and the resulting net change in CO_2e emissions per ton of CO_2e of increased wood use for nonresidential construction by 2060.[a]

Region	Forest	HWP	Logging slash	Manufacturing emissions	Net CO_2e[b]	CO_2e of increased NR wood production	Net CO_2e/tCO_2e of wood[c]
North	39 (0.81)	−25 (−0.52)	−24 (−0.50)	−81 (−1.68)	−91	48	−1.90
South	−182 (−0.74)	−62 (−0.25)	−39 (−0.16)	−414 (−1.68)	−698	246	−2.83
West	198 (1.49)	−37 (−0.28)	−17 (−0.13)	−224 (−1.68)	−80	133	−0.60
United States	55 (0.13)	−124 (−0.29)	−81 (−0.19)	−720 (−1.68)	−870	428	−2.03

[a] Positive values indicate a contribution to net emissions increase. Negative values indicate a contribution to net emissions decrease. Numbers in parentheses indicate values per ton of carbon dioxide equivalent (tCO_2e) of extra wood use. HWP = harvested wood products.

[b] Net CO_2e = (Col. 2 + Col. 3 + Col. 4 + Col. 5).

[c] Also called displacement factor for wood substitution. Estimated by dividing net CO_2e (Col. 6) by CO_2e of increased NR wood production (Col. 7) or by summing numbers in parentheses in each row.

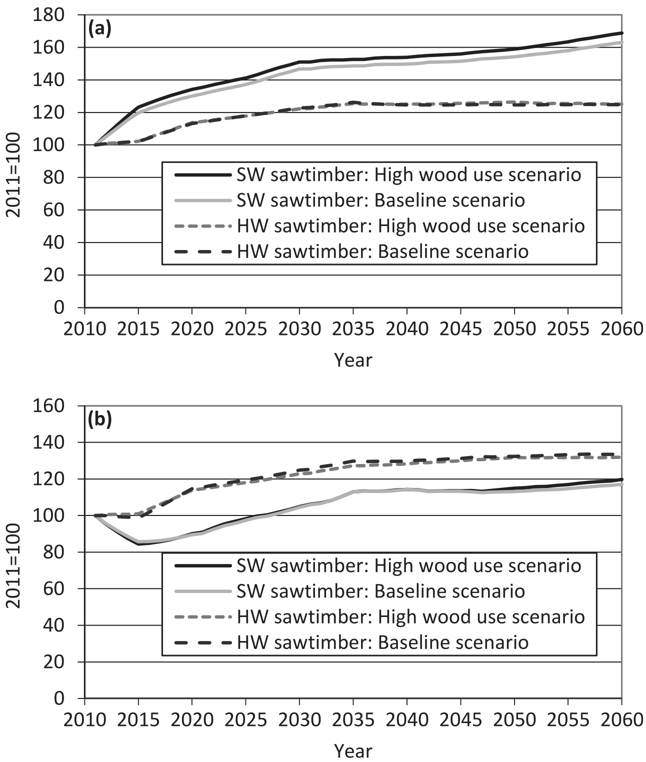

Figure 2.—Projected Southern softwood (SW) and hardwood (HW) sawtimber (a) and nonsawtimber (b) price (2011 = 100) for baseline and high wood use scenarios, 2010 to 2060.

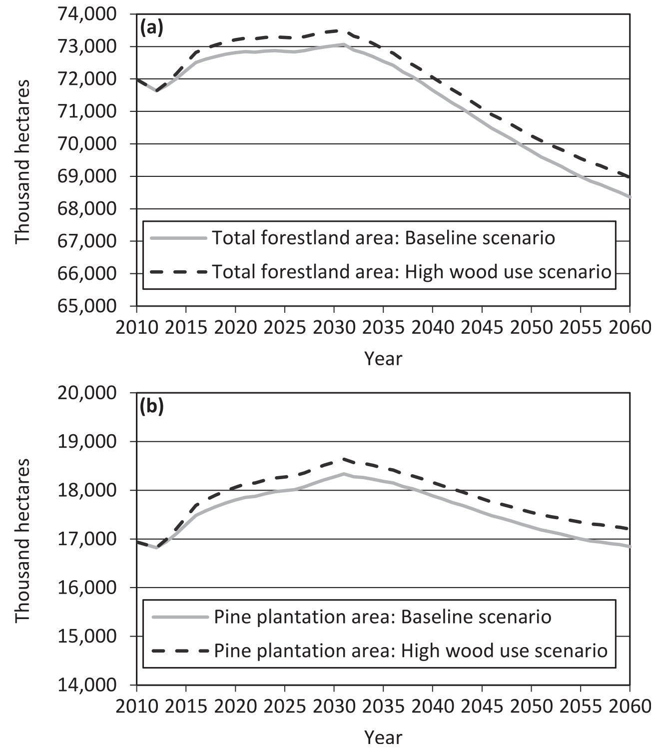

Figure 3.—Projected forestland (a) and pine plantation (b) area in the South (thousand hectares) in baseline and high wood use scenarios, 2010 to 2060.

lower average current growth, so the absolute growth response is greater in the North. The result is that there is a greater depletion of inventory per unit of product for the West than for the North. To the extent that harvest in the West would be focused in areas with greater than average growth response, the C depletion per unit of product would be less. Note that the forest investment effect was not modeled in the North and the West. Wear (2011) found that changes in timber prices were not significant in predicting changes in timberland area in these regions. The higher loss of forest inventory in the West could be an overestimate, to the extent that timber harvest would selectively come from faster growing forest areas and to the extent that the production increase is more evenly distributed across plywood, lumber, and OSB.

The effect of logging slash C on regional net CO_2 savings is substantial for the North compared with the South and the West. Although the South produces the largest total logging slash C in absolute terms, the generation per ton of CO_2e of wood produced is the largest in the North (–0.50 tCO_2e/tCO_2e of wood), followed by the South (–0.16 tCO_2e/tCO_2e of wood), followed closely in turn by the West (–0.13 tCO_2e/tCO_2e of wood; Table 2). Such regional differences in logging slash C are partly determined by the proportion of logging slash generated with total timber harvests. For example, the North generates higher proportions of logging slash associated with HW harvests (28%), compared with a lower proportion of logging slash for SW harvests in the South (15%; Smith et al. 2009). Note that increasing logging slash per unit of final product could increase emissions reductions per unit of product. For this reason, the metric of emissions reductions per unit product may not be the best to judge the benefits of wood product production and use, and below we suggest the use of emissions reductions per unit of wood harvested.

To understand the contribution of each C pool category on net CO_2 savings relative to the contribution from manufacturing emissions, we estimated the ratio of change in each C pool to net change in manufacturing emissions (Table 3). We did this because the net change in manufacturing emissions is fixed and the resulting ratios and C change contributions can be compared across regions. These ratios reveal, for example, that the contribution from forests by 2060 is positive in the South (44% of

Table 3.—Percent contributions of each carbon (C) pool category to net CO_2 savings, relative to the contribution from manufacturing emissions between baseline and high wood use scenarios, 2030 and 2060.[a]

		C stocks/emissions (%)			
Region	Year	Forest	HWP	Logging slash	Manufacturing emissions
North	2030	1	29	4	100
	2060	−48	31	30	100
South	2030	18	16	11	100
	2060	44	15	9	100
West	2030	−87	14	14	100
	2060	−88	17	8	100
United States	2030	−22	16	11	100
	2060	−8	17	11	100

[a] Positive values indicate a contribution to net emissions reduction. Negative values indicate a contribution to net emissions increase. HWP = harvested wood product.

manufacturing emissions offset) and negative in the North and the West (–44% and –88% of manufacturing emissions offset, respectively). In contrast, contributions from HWP and logging slash are positive for all regions. The largest HWP contribution from a change in wood product production comes from the North (31%), followed by the West and the South (17% and 15% of manufacturing emissions offset, respectively). Similarly, the largest logging slash contribution comes from the North (26%), followed by the South and the West (9% and 8% of manufacturing emissions offset, respectively).

Discussion

Our analysis of net C emission impacts of increased SW lumber and structural panel use for NR construction reveals implications related to use and management of US forest resources in climate change mitigation. First, where forest investors respond to increased timber prices resulting from increased SW lumber and structural panel demand (e.g., in the South), the reduction in inventory (forest C) caused by harvest could be offset by increased C accumulation from added investment in plantation. Second, increased production of SW lumber and structural panels to meet the demand for NR construction can displace nonwood materials, with notably reduced net manufacturing emissions. To provide a context, the cumulative net emission reduction of 870 tCO_2e over 50 years, as a result of increased wood use in US NR construction estimated in this study is about the same as 1 year's addition of C to US forests (881 tCO_2e) in 2013, which would offset about 13 percent of the total US national GHG emissions in 2013 (US Environmental Protection Agency [EPA] 2015). We also compared our cumulative emissions reductions with the US total energy-related emissions projected in the US annual energy outlook (US Energy Information Administration [EIA] 2015) for the reference and the alternative cases. The difference in cumulative CO_2 emissions between the reference case scenario and the scenario of lower CO_2 emission (low economic growth case) projected by the US annual energy outlook (US EIA 2015) represents avoided emissions due to lower economic growth. Our avoided emissions of 424 Tg CO_2e by 2040 represents about 7 percent of the avoided emissions projected from 2013 to 2040 in US annual energy outlook, which was 6,292 Tg CO_2e by 2040. This offset resulted from relatively small increases in wood use (about 7,777 m^3 of wood) suggesting a greater offset potential with greater wood use in NR construction.

We also estimated how much CO_2 reduction occurs (per 1,000 m^2 of space) if a typical US building classified as a store uses wood in all structural applications compared to with current average wood use. The average use would include the current mix of wood-framed as well as steel- and concrete-framed buildings. When a store uses wood in structural applications (about 153 m^3 of wood per 1,000 m^2) beyond the average amount of wood used currently for an average store (about 13 m^3/1,000 m^2), we found that it could offset an extra 255 tCO_2e. The offset estimate would be greater if we were comparing to an average non–wood-framed building.

Our analysis shows that variation in net CO_2 savings among regions is in large part owing to variation in forest harvest minus forest regrowth. One factor that could be influencing the variation could be shifts in the type of mix of products produced, e.g., toward or away from products that produce more logging residue or are more or less efficient in use of timber per unit of product. However, this was not a factor in our analysis because there is less than a 1 percentage point shift in the shares of lumber, plywood/veneer, and OSB within each region between the baseline and high wood use scenarios.

The results above indicate the effect of emissions reductions from a change made by builders to increase wood in NR buildings. That is, we give our emissions savings per unit of increased wood use. It is also possible to indicate emissions reductions impact from the point of view of forest landowners per unit of timber harvested, which for our discussion, can be defined to include sawtimber, nonsawtimber portions of trees, and the logging slash left on harvest sites (Table 4). Because we now include CO_2e of all materials cut down from forests in the denominator of Equation 4, the CO_2 savings numbers are now smaller. The net change in CO_2e emissions per ton of CO_2e of harvest over 50 years is shown in Table 4. The net emissions reduction per unit of harvest, compared with the reduction per unit of product produced, will have additional influence from (1) the efficiency of harvest (the amount of logging slash generated per unit of wood volume removed from the site) and (2) the efficiency of conversion of sawtimber or nonsawtimber to lumber and panels. A reduction in logging slash (increase in harvest efficiency) would result in more product per unit of harvest with more C stored in HWP per unit of harvest and more substitution for nonwood products per unit of harvest. The overall result would be more C emission offset overall per unit of harvest.

An increase in efficiency of conversion of timber removed from forests to lumber or panels would also result in more product per unit of harvest with more C stored in HWP per unit of harvest and more substitution for nonwood products. It would also result in no increase (from the base case) in C stored in logging slash. For the average US logging slash generation, the HWP C and substitution gains more than offset the reduction in C storage in logging slash. So the overall result would be more C emissions offset per unit of harvest. By evaluating the influences on emissions reductions per unit of harvest, we can see the importance of improvements in technology to convert timber to products, in forest operations technology and in forest management, to provide the most C emissions reductions benefit per unit of timber harvest.

We used Sathre and O'Connor's (2010) average value of –2.1 tCO_2e/tCO_2e, although we adjusted it downward to –1.68 tCO_2e/tCO_2e to exclude offset of wood energy

Table 4.—Net change in carbon dioxide equivalent (CO_2e) emissions, per ton of CO_2e of all harvest (sawtimber + nonsawtimber + logging slash) associated with increased nonresidential construction by region, by 2060.[a]

Region	Forest	HWP	Logging slash	Manufacturing emissions	Net CO_2e/tCO_2e of wood
North	0.64	−0.41	−0.39	−1.33	−1.49
South	−0.65	−0.22	−0.14	−1.47	−2.48
West	1.14	−0.21	−0.10	−1.29	−0.46
United States	0.11	−0.24	−0.16	−1.40	−1.69

[a] Positive values indicate a contribution to net emissions increase. Negative values indicate a contribution to net emissions decrease. HWP = harvested wood product.

emissions by forest regrowth. As discussed above, the studies used in their meta-analysis had some variation in system boundaries, but none considered market dynamics such as altered consumption or trade for other wood uses and/or altered investment in plantations as was done in this article. This study provides the effect of increased wood use and displacement of nonwood materials by region and over time and provides detailed 50-year estimates of net change in CO_2 emissions per ton of CO_2e of increased wood use for NR construction by including actual emissions, C recovery, and the gain in all relevant C pools, including market-driven changes in consumption and changes in planting.

This study has certain limitations and uncertainties that should be considered when interpreting the results. First, our results are valid for the assumed price elasticities of demand for products and supply of wood materials, including the supply elasticity with respect to changes in timber inventory. Whereas the elasticities are consistent with the published literature, a higher timber supply elasticity with respect to price would mean that higher timber supply could be attained with a smaller increase in price. A smaller price increase in the South would mean a smaller increase in investment in plantations and intensified management. This would result in reduced net CO_2 emission savings in the South. If instead we had used a lower timber supply elasticity with respect to price, our results would have led to a larger estimated net emission savings. Second, our results are dependent on the assumed base year input–output coefficients (timber inputs per unit of final products) and amount of logging slash produced per unit of harvest. We used input–output coefficients and logging slash factors based on the 2007 timber product output (TPO) database and assumed that they do not change over time. However, more recent timber product output data show a somewhat higher amount of logging slash and higher timber use per unit of primary products. Using the more recent input–output relationships would have resulted in lower forest C and higher logging slash C, with the net effect being uncertain. Similarly, our study did not explicitly assess the additional benefit of after-life use of construction wood materials, except to the extent that the average estimate of net manufacturing emissions reductions from Sathre and O'Connor (2010) included some studies that included those effects. Our estimated net CO_2 savings would be increased to the extent that discarded wood (1) is burned for energy that offsets fossil emissions, (2) is used in new long-lived products, or (3) goes to landfills where methane is captured and burned for energy that offsets fossil emissions.

Our model projects a similar amount of logging slash recovery for wood energy use in both scenarios (thus, a small net increase in logging slash recovery) as determined by a projected small demand for wood energy in the United States. Increased logging slash recovery with efficient use has been shown to provide climate benefits (e.g., Gustavsson et al. 2015). We did not evaluate the climate impact of increased logging slash recovery in this study because our focus here was to evaluate the net CO_2 emission impacts of increased wood use for NR construction. Including the climate benefit of increased logging slash for wood energy would have improved our estimated net CO_2 savings.

Our estimates of logging slash C are based on an assumed constant decay factor for slash. However, in reality the decay rate can vary with climate and species. We expect that effect of variations in decay rate would be low and will not significantly influence our estimated net CO_2 emission reductions. We make this conjecture based on the findings of our previous study where we estimated sensitivity of CO_2 emissions to various logging slash decay rates including two extreme assumptions of no decay and near instantaneous decay (Nepal et al. 2014). That study showed that the variation in decay rate had very little impact on the estimated C emissions primarily because the difference in logging slash C stock between the two cases of baseline and increased wood energy uses was a small fraction of the difference in total cumulative emissions, which is also true in the case of the present study.

This study did not consider the consequences of soil C loss due to increased harvests in the alternate scenarios. Studies have shown that soil C impact of harvests depends on harvesting methods (e.g., whole tree harvesting with residue removal, or stem only harvest with residue retention, e.g., Jones et al. 2008), and how forest land is converted after harvest (Mann 1986, Johnson 1992). For example, Jones et al. (2008) found that the harvesting in plantation forests that involved residue retention had significantly larger C stock in the fine and coarse fraction of forest floor and total soil than whole tree harvesting that removed residues. In general, the literature indicates that harvesting that follows reforestation has no or little effect on soil C reserve on global scale since the average effects of harvesting on soil C are minimal (Detwiler 1986; Johnson 1992, 1994). To the extent that this study considers logging slash retention and reforestation of harvested forests, the effect on soil C loss should be minimal.

Conclusions

This study estimated the net CO_2 emission reductions associated with an increase in SW lumber and structural panel use for low-rise NR building construction by comparing projections from forest sector models from the baseline and high wood use scenarios. Projected results from a national and global timber market model and a more detailed southern timber supply model incorporate biological and economic market interactions that affect forest product production, net trade, timber harvests, timber prices, regional timber regrowth, and timber inventory. A key finding is that the increased use of wood products for NR construction can result in a net CO_2 savings over the 50 years, both nationally and by region. The net savings is due to net manufacturing emissions reduction resulting from the displacement of nonwood materials by wood, increased C stored in HWP, and changes in C stored in forests and in logging slash. Results suggest that although the increased demand for wood products decreases forest inventory, much of this loss is recovered over time because of biological regrowth and price-induced investments in pine plantations in the South. The regional amounts of net CO_2 emission reductions per unit of increased production vary depending on regional differences in harvest quantities, forest product production, and biological and market-induced recovery of forest C. Estimates of net CO_2 savings over a 50-year period shown in this study imply that an increase in the use of wood in NR construction could be considered as part of a climate change mitigation strategy. Policies or practices would need to result in actual displacement of nonwood product use in NR construction.

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