INTEGRATING PRODUCTS, EMISSION OFFSETS, AND WILDFIRE INTO CARBON ASSESSMENTS OF INLAND NORTHWEST FORESTS

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Abstract. Forest inventory and harvest data from life-cycle inventory and life-cycle assessment for the forest resources of the Inland Northwest region covering Idaho, Montana, and eastern Washington were used to estimate the impacts of management action on the full suite of carbon accounts that can accrue from forest management. The carbon accounts include the forest, wood products, the benefit gained from using wood products as substitutes for alternative products that are fossil fuel-intensive to produce, and the displacement value of using woody biomass to replace fossil fuel. A landscape-level assessment of projected carbon storage by owner group shows that in 100 yr, management on State and Private Forests can sequester or avoid emissions equal to 294 t/ha of carbon, which equals over 1.9 Gt of carbon across 6.5 Mha. Seventy-nine percent of the carbon accumulates beyond current forest carbon inventories. On National Forests, carbon sequestration and avoided emissions are 152 t/ha over 11 Mha of unreserved forests equaling 1.4 Gt of carbon under predictions for a doubling of the 20th century fire rate. The carbon storage in buildings and the substitution benefits override the potential gains of attempting to leave high carbon stocks stored in the forest in this region where disturbance from fire and insect outbreaks dominates the forest ability to sequester carbon.

Keywords: Inland Northwest, LCI, LCA, wildfire risk reduction, forest, wood product substitution, fossil fuel displacement.

INTRODUCTION

This article provides an analysis of the carbon consequences of forest management integrating product impacts back to a unit hectare for forests of the US Inland Northwest (INW) region. It is part of a larger set of studies produced by the Consortium for Research on Renewable Industrial Materials (CORRIM) that used life-cycle inventory (LCI) and life-cycle analysis (LCA) methods to quantify the carbon consequences of timber growth, harvest, and product manufacturing for the four main timber supply regions of the US: the Pacific Northwest (PNW) and the Southeast (Johnson et al 2005; Milota et al 2005; Wilson and Sakimoto 2005), the Northeast/North Central, and the INW (Oneil et al 2010; Puettmann et al 2010a, 2010b). INW covers timber-producing land in Idaho, Montana, and east of the Cascade Mountains in Washington where manufacturing infrastructure still remains. It accounts for 91% of the roundwood production for the Rocky Mountain region and 10% of the roundwood production for the PNW region based on 2006 Resource Planning Act data (FIA 2009) with most of that volume coming from private and state sources. Together the LCI/LCA for the forest resources and manufacturing processes provide a comprehensive cradle-to-gate assessment of the inputs and emissions and product outputs in these four major US timber supply regions. To extend the cradle-to-gate analysis to end-of-product life for harvested wood products, it is necessary to

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expand the boundary conditions to include changes through time for all carbon pools and examine key factors specific to each region that could impact the inputs and outputs at each gate in the chain of events.

Forestry operations store carbon in the forest as one carbon pool. Harvesting mature trees produces a stream of manufactured products that transfers sequestered forest carbon into products carbon, a second pool. If forest biomass is used as energy, it can permanently offset the carbon emissions from displaced fossil fuels as a third pool called the displacement pool. Each pool reduces carbon emissions, although displacement offsets carbon emission increases rather than storing carbon directly. The displacement carbon pool is equal to the reduction in fossil-fuel use and emissions by burning woody biomass as a biofuel in place of natural gas for heating and processing energy needs (Puettmann and Wilson 2005). Using wood products in buildings stores carbon for extended time periods, but it may not be permanent after the end of building life. However, using wood products that substitute for fossil fuel-intensive products such as steel and concrete displaces emissions from these products as another permanent emission offset, providing an important fourth forest-related carbon pool called the substitution pool (Perez-Garcia et al 2005a; Lippke and Edmonds 2006). Together these four carbon pools, forests, products, displacement, and substitution, and the requisite fossil energy inputs needed to harvest and manufacture products constitute the components of a full carbon accounting of harvested wood products from the INW region.

Perez-Garcia et al (2005a) used LCI/LCA data for a representative PNW coastal Douglas-fir stand and the products from the harvested log to integrate all carbon impacts back to a unit hectare. They also extended the analysis for the useful life of construction products, including the substitution and displacement value where wood is used in place of alternative materials that generally produce higher fossil-fuel emissions. The Perez-Garcia study provided a framework for identifying how LCI/LCA could be integrated back to the landscape for capturing life-cycle carbon impacts in other regions. The goal of this article was to integrate the forest resources analysis for the INW across all stages of processing using protocols developed by Perez-Garcia et al (2005a). That integration includes an assessment of the impact of management activities on potential carbon storage and displaced emissions for the forest, product, displacement, and substitution pools and an assessment of how natural disturbance and management actions designed to address natural disturbance can influence carbon sequestration and storage outcomes. Fire and insect outbreaks are more prevalent in this region complicating the analysis of management alternatives.

Background information on INW forest conditions, characterization of fire rates, and assumptions used are described below. Next, details are provided on forest type stratification, treatment simulations using growth and yield models, projected harvest rates, and resulting stand conditions from which to estimate carbon pools for all owners and forest types. We then categorize the results by owner with National Forests being treated separately from State and Private Forests. First we report on the impact of management on State and Private carbon pools followed by an analysis of fire rates and their impacts on National Forests. Finally, we summarize the differences across owners and treatments and highlight performance improvement opportunities.

BACKGROUND

In contrast to the relative uniformity of timber producing forests in the PNW, the forests of INW are a complex mosaic of species mixes, habitat types, productivity classes, and ownerships with 63% of the timber-producing forests under the National Forest jurisdiction. As a result, management regimes are highly variable. In addition, natural disturbances, in particular, wildfire and more recently massive outbreaks of mountain pine beetle (*Dendroctonus ponderosae*) (MPB), play a dominant role in the carbon consequences of forest management across the region. Both disturbances substantially alter forest structure, affecting the rate of carbon sequestration and emissions, and the potential to store carbon in product, displacement, and substitution pools. Wildfire impacts can be reduced by thinning treatments designed to reduce ladder fuels and canopy bulk density (Agee 2002). These treatments may also serve to restore the natural forest structure that has been lost by a century of fire suppression.

Given the complex nature of the forests and ownership pattern for the INW timber supply region, integrating LCI/LCA data back to the forested hectare required us to: 1) develop a weighted average hectare to represent the diversity of species, habitat types, and forest management regimes as a landscape average; 2) separate the analysis into National Forest and State and Private Forest owner groups to account for substantially different management goals and approaches; and 3) examine how wildfire will influence the carbon consequences of forest management.

In generating LCI/LCA data for the forest resources in the four US supply regions, the CORRIM research protocol treats forests under sustainable management as carbon-neutral, ie the forest carbon neither increases nor decreases. This means that the carbon in the harvest that is removed from the forest plus any decomposition of dead wood is balanced by new growth. The carbon-neutral assumption suggests that harvested areas are regenerated to current stocking and there are no long-term soil carbon losses. The soil carbon-neutral assumption is supported by a meta-analysis of harvesting impacts on soil carbon that shows no substantial differences before and after harvest under normal management treatments (Johnson and Curtis 2001). The 2007 IPCC summary to policymakers indicates that emissions from above-ground decay of biomass remaining after timber harvest is not counted as an emission source and that core mitigation policies include harvested wood product management and the use of forest biomass to offset fossil-fuel use (IPCC 2007). In 1996 and 2001, the IPCC emphasized the critical role that substitution of sustainably produced products for fossil fuel-intensive products would have over and above the finite carbon storage potential available through reforestation and afforestation alone (IPCC 1996, 2001). The IPCC (1996) report goes so far as to identify substitution as having up to four times more carbon benefit than conserving forest stocks or afforestation alone, presumably because of the impact on reducing fossil-fuel sources of emissions. A meta-analysis of the fossil-fuel savings that can be obtained by using wood products in place of building materials such as steel and concrete (Sathre and O'Connor 2008) found a substitution benefit of 2.0 units of carbon displaced per unit of carbon in wood used. The meta-analysis estimate is close to the carbon impact of house framing substitution studies produced by CORRIM but representative of a much broader set of substitution studies.

We use the Sathre and O'Connor (2008) substitution parameter in conjunction with the LCA of forest resources for the INW (Oneil et al 2010) and LCA of manufacturing (Puettmann et al 2010a) to demonstrate the integrated impact of forest management at any given time for the four carbon pools noted previously. We then extend the LCA through time to include end-of-useful-product-life considerations.

Because the LCI/LCA for INW forests examined two scenarios called the base case and alternate case, we continue that nomenclature here. The base case describes current management intensities and harvest rates for all owner groups. The alternate case increases management intensity on State and Private Forests and increases the area treated with thinning treatments designed to reduce fire risk but retaining a large tree overstory as an ecosystem restoration objective on National Forest lands. To integrate wildfire impacts into the analysis, we simulated a range of changing fire rates based on changes in fire history and studies on the impacts of climate change. A secondary treatment on National Forest lands was designed to capture the mortality from MPB on lodgepole pine stands.

Baseline Fire Rates

In the INW, wildfire is the dominant natural disturbance regime (Agee 1993; Calkin et al 2005), although insect and disease outbreaks are common with MPB outbreaks creating extensive swaths of mortality in the recent past (Carroll et al 2003; Gibson 2006; Oneil 2006). Collectively, these disturbance agents along with climate determine the range of forest stand structures, density, and timber volume in the forest. Fire histories for the region are complex. In the dry forests typically found at low elevations, wildfire frequencies as short as a decade had historically reduced stand density, leaving only large fire-resistant trees in the overstory, whereas at high elevations, less frequent wildfires typically caused complete stand replacement every 100 – 300 yr (Agee 1993) often in concert with an insect outbreak. Low-elevation forests with frequent fires are classed as having low-severity (frequent) fire regimes with the opposite end of the spectrum being high-severity (infrequent) fire regimes at high elevations. At mid elevations, a mosaic of fire intensities and impacts produces a complex mix of forest structures and species and is classified as a mixed-severity fire regime (Agee 1993). Departure from these historic fire regimes because of fire suppression has been used to categorize forests for fire risk assessments with those forests more distant from their historic norm having a higher fire risk condition class (Schmidt et al 2002). For this analysis, we classified ponderosa pine, dry Douglas-fir, and dry grand fir habitat types as dry forests; moist Douglas-fir, grand fir, cedar-hemlock habitat types, and the mixed conifer forest types as moist forests in mixed-severity fire regimes; and subalpine fir, spruce, larch, and lodgepole pine forests at high elevations as cold forests in high-severity fire regimes.

In recent years, the extent and severity of wildfire has been increasing with a multiplicity of reasons identified as key drivers. Most prominent among the reasons are extreme weather conditions associated with climate change and the long history of fire suppression and its effect on increasing fuel loads outside the historic range of variability for the region. Climate impact research on western forests indicates that we can expect at least a doubling of the area burned for this region (McKenzie et al 2004; Gedalof et al 2005; Westerling et al 2006; Littell et al 2009b). The predictors for large fire years are all related to climate, suggesting that it is playing a critical role in the increasing extent and severity of wildfire. With this in mind, using expectations for fire rates for the 21st century was viewed as a more useful baseline for assessing forest conditions than using fire rates from the 20th century history of successful fire suppression. To understand how a doubling of area burned would change the fire rate, it was first necessary to identify a historical fire rate that is characteristic of the INW region.

Estimating a historical fire rate (fractional area burned) is difficult because of the confounding influences of 20th century fire suppression, the complexity of natural fire regimes in different forest types, forest management, climatic variability, and the stochastic nature of wildfire occurrence. Even before fire suppression was common, stand-replacing wildfires did not burn all areas, leaving patches of fire refugia, and burning some land more than once in a short period to produce a mosaic of stand conditions. Camp et al (1997) estimated that fire refugia would have historically (presettlement) included approximately 12% of the landscape in the dry and mesic forests of the Washington East Cascades region. This suggests that 88% of that landscape burned at least once in a 100-yr period as the historical fire rate for stand-replacing fires. Historic fire rates for low- and mixed-severity fire regimes calculated using fire return interval methods vary widely across the analysis region. The detail provided by Agee (1993) for the various forest types in the INW suggests nonlethal fire return intervals from 6 yr on dry forests to 100 yr on some mixed-severity forest types.

Littell et al (2009b) used a top-down analysis to model fire rates for National Forests as a function of climate variables. They used 1980 – 2000 weather and fire data that was extrapolated from 1916 – 2003. A weighted average of the Littell et al (2009b) results covering the forested area in this study suggests an average of 24% of the National Forest lands would burn over a 100-yr period under 20th century climate, fire suppression, and forest management regimes. Littell's result is reasonably consistent with our analysis of Forest Inventory and Analysis (FIA) data (FIA 2009) that identifies approximately 24 – 36% of National Forest timberlands as originating from fire during nearly the same period with a weighted average fire rate of 30% for the entire region covered by the study. Differences in fire rate are likely a result of the inclusion of higher elevation reserved lands in the Littell et al (2009b) work because they burn less frequently than mid- and low-elevation forests. The analysis of FIA data and the Littell et al (2009b) estimates provide a reasonably reliable indication of fire rates during a period of effective fire suppression.

Examination of land burned during 2002 – 2009 in INW using NIFC (2009) fire data indicates that the rate of wildfire since 2002 has increased substantially. Although the NIFC fire data do not allocate burns to ownerships, wildfires occur more often on National Forests. Analysis of Washington Department of Natural Resources data on land burned by ownership category for 1991 – 2006 indicates that 81% of the land burned was on National Forests that comprise only 44% of the forested land base in eastern Washington where most of the fires occur (Oneil 2010). This produces a 5.5 times higher National Forest burn rate than that in State and Private Forests. Allocating burn rates accordingly results in 7.5 times more land burned on National Forest lands in the post-2002 period than the 0.24%/yr estimated by Littell et al (2009b) for the 20th century. If the average area burned during 2002 – 2009 fires became the 21st century fire rate, it would take approximately 14 yr to burn the same amount of forest as was burned during the 20th century. This is more than a doubling of the presettlement fire rate of Camp et al (1997). However, in all likelihood, we could expect substantial negative feedback on area burned over time as ongoing wildfires eliminated excess fuels that would eventually reduce the fire rate.

Top-down climate parameters control drying trends, fire ignition, and rate of spread. Bottom-up controls on fire behavior and outcomes are mostly determined by the amount of fuel in the forest. In a managed landscape, fuel loading is driven by the method and timing of harvest and fuel treatments. Whereas pre-European settlement stands had frequent understory burns allowing a stable overstory of a smaller number of large trees, in particular for the dryer stands (Everett et al 2008), with human population growth and a century of fire suppression, unmanaged stands today have become overly dense with excessive fuel loads. Now, when fires do occur, they are generally stand-replacing fires (Sampson et al 2001) instead of low- or mixed-severity fires that were a natural part of INW disturbance history and forest ecology (Agee 1993). Here we use top-down controls on wildfire to determine a 21st century fire rate and bottom-up controls on fire behavior to infer potential fire impacts on forest carbon stores.

METHODS

Inventory Data, Stratification, and Growth and Yield Modeling

The analysis used forest inventory data from FIA plots for Idaho, Montana, and eastern Washington covering all unreserved coniferous productive forests that generate more than 1.4 m^3/ha·yr of volume. Plot data were segregated by owner group and major habitat type. The ownership categories used were state, private, and National Forest. Table 1 provides a summary of expected yield by site class and owner group averaged across the INW region based on these FIA plots. For the INW, FIA site quality is given in growth potential (m^3/ha·yr) rather than site index as the preponderance of mixed-species forests and of the abundance of forests with multiple size classes and age cohorts makes the site index less meaningful. For the same reasons, the Forest Vegetation Simulator (FVS) (Wykoff et al 1982) variant used to estimate inventory growth and yield in the region uses habitat type rather than site index to identify differences in growth and yield across the landscape.

Table 1. *Forest yield allocation by owner group in the Inland Northwest.*

Site class		1	2	3	4	5	6	Total	Average site class
M3/ha/yr		>15.8	11.6 – 15.7	8.4 – 11.5	6.0 – 8.3	3.5 – 5.9	1.4 – 3.4		
Owner group		Thousands of hectares and percent							
Private	Ha	3	78	421	986	1790	1716	4994	4.93
	%	0.1	1.6	8.4	19.8	35.8	34.4	100.0	
State	ha	1	41	172	446	488	376	1526	4.64
	%	0.1	2.7	11.3	29.2	32.0	24.7	100.0	
NF	ha	10	71	622	2053	4170	4090	11,015	5.05
	%	0.1	0.6	5.6	18.6	37.9	37.1	100.0	
Total by site class	ha	14	190	1214	3485	6448	6183	17,534	4.98
	%	0.1	1.1	6.9	19.9	36.8	35.3	100.0	

NF, National Forest.

To be consistent with site quality estimates from the data and their projection in the growth model, we segregated all plots by owner and habitat type for all habitat types comprising more than 4% of the total area for each owner group. For each owner and habitat type category, we used the median inventory with respect to basal area and quadratic mean diameter as a representative sample. The habitat type/owner categories were then weighted according to the percentage of the land base they occupy to generate "stands" that were used for simulating growth, harvest, and product outputs for the region. Stand growth and yield were simulated using three FVS variants, Inland Empire, East Cascades, and Blue Mountains, inside the Landscape Management System software (McCarter et al 1998). Fire impacts were simulated using the FVS Fire and Fuel Effects (FFE) model (Reinhardt and Crookston 2003; Rebain 2009).

Modeling Assumptions

Simulated treatment regimes. Simulated treatments included: precommercial thinning; even-aged silviculture systems including clearcut, seedtree, and shelterwood; and uneven aged silviculture systems, which removed either the merchantable or nonmerchantable materials depending on owner objectives. Decisions on the simulated treatment were based on management objectives and existing stand conditions with multiple treatment paths within a given ownership/prescription class.

To avoid harvests containing substantial pulp volumes, entries were timed throughout the simulation period so that stands would yield merchantable volume. Decision rules indicating merchantable volumes were based on minimum values in top height, volume, and the equivalent of top diameter. In addition, treatments were designed to ensure that the residual stand contained sufficient overstory to meet statutory green tree retention targets. When a merchantable entry was simulated, both the removed and remaining volumes vary as a function of starting inventory that influenced how pole/sapling and understory layers were treated at the time of logging. Testing determined if understory and pole/sapling layers were cultured as residual understory or removed because they lacked the capacity to respond to cultural treatments. The tests themselves assess the residual stand for species mix relative to site potential and habitat type, stocking levels, height-to-diameter ratios, and release potential that is essentially a function of site quality and live crown ratio. For high elevation and wet forests on private lands, even-aged harvest systems were simulated, leaving a minimum of 10 trees/ha (TPH) greater than 250 mm dia at breast height (DBH) to meet statutory requirements for green retention trees.

On private dry forests, seed tree and/or shelterwood regimes were simulated to re-establish the next crop, but with no retention of the dominant cohort except for statutory requirements for green retention trees. On private moist forests, regular entries removed merchantable volume with only minimal stand improvements. Alternative strategies in moist forests also had regular stand entries but with additional fill planting and stand improvements to maximize forest growth and yield. Harvests on state forests assumed similar treatment regimes as those for private forests except that more and larger leave trees were retained as part of a statutory requirement, a seed tree system, or as shelterwood, including retention of the largest trees in the stand. On State and Private Forests, simulations included regeneration with seral species in the range of 620 – 865 TPH with additional natural regeneration added to reflect the species diversity in the overstory. Seral species included ponderosa pine, larch, and white pine depending on the habitat type.

For National Forests, we assumed that thinning from below was standard with trees removed up to 300 mm DBH. After that limit was reached, on pine stands at risk for MPB outbreak, a further removal to a basal area of 14 m^2/ha was also applied. No planting was assumed, but natural regeneration was included in the simulations with species compositions based on forest type, overstory species composition, and habitat type.

Harvesting. As a base case, simulated historical harvest rates by ownership were applied across the habitat types according to their percentage on the land base. For habitat types that represent a large percentage of the land base, the inventory data were replicated with the total area of all stands representing the total area of the habitat type. Even flow harvest and allocation between the management intensities were simulated by treating the replicated stands at different time intervals and intensities but still maintaining the forest cover diversity and expected landscape level outputs across the region. Base case harvest treatments were targeted to meet the average volumes harvested in the past 30 yr by State and Private owners. Cross-validating the 30-yr average harvest rates on private lands to yield from the median stand inventory weighted by habitat type suggests that current management on private forests leaves very little merchantable volume remaining in any cutting cycle, especially over the first 30 yr. This situation is more apparent in some subregions than others and has been explored in detail on the future of Washington's forests (Eastin et al 2007; Lippke et al 2007) for the eastern Washington portion of the study area. For the alternate case on State and Private Forests, we simulated an increase in the number of entries and the volume removed at each entry with additional control over regeneration stocking and species selection.

Beginning in 1994, National Forests show a marked decline in volume harvested across all three states in the study area relative to the 30-yr average; therefore, the base case for National Forests reflects harvest trends from 1999 – 2002. Specifically identified reserve land on public lands was not included in the analysis. Forest types on dry and mesic forests where management would likely ensue under programs such as the Healthy Forest Initiative were included up to the area treated currently and all such areas are included in the alternate case analysis. For the alternate strategy, all National Forest land in dry, moist Douglas-fir, and grand fir forests were treated using restoration thinning strategies to reduce fire risk by restoring a savanna-like overstory of larger trees similar to pre-European conditions. In addition, we included treatment of lodgepole pine forests to address MPB risks and infestations, recognizing that these treatments would not be successful in ameliorating fire risk. Treating all National Forest land within these four habitat types roughly corresponds to thinning within low- and mixed-severity fire regime locations as well as managing the escalating impact of MPB in lodgepole pine forests. Total treatable area and volume were determined for the alternate case and then compared against area treated on National Forests during the current management era to arrive at a scale-down factor for the base case on National Forests. Under the National Forest alternate case, there is a 5-fold increase in area treated in Idaho and Montana and a 4-fold increase in eastern Washington. The alternate regime for National Forests assumed that continued management to reduce fire and insect risks would occur despite lack of financial incentive after the first entry. For the National Forests, each initial entry generated enough merchantable volume to haul material to the processing facility. To maintain the benefit of thinning treatments with respect to fire safety, additional treatments were required on a 30 – 40-yr return interval. Depending on the forest growth rates, the second and any subsequent thinning treatments typically did not yield much merchantable volume; therefore, the carbon accounting data include estimates of harvest inputs but no hauling costs because there is insufficient wood to warrant removal.

Harvest volume from all owners was categorized as sawlog, hewsaw log (a small sawlog less than 180 mm dia), or pulp log using the Eastside Scribner log scaling rule. Sawlog and hewsaw are merchantable, but direct delivery of pulp logs is contingent on cyclical pulp markets. For that reason, direct delivery of pulp logs from the forest was not assumed to represent a significant part of the volume, although some pulp quality material would be brought in as part of the larger merchantable log. Whereas the range of management practices used across all owners is only an approximation for the diversity of approaches used in the region, they define a logical range of management intensities and are sufficient to characterize the impact of current and potential future management practices. This analysis shows the carbon accumulation yearly over 110 yr reflecting trends in forest carbon sequestration and wood use assuming continued and ongoing active management and silvicultural investment.

Carbon estimates. Forest pools include the live trees and all their components, the dead trees that decompose at species specific rates as defined by Aber and Melillo (1991), and inputs and decay of forest litter. Biomass allocation of tree components to stem, crown, and root used the species-specific coefficients in Jenkins et al (2003) biomass equations with conversion of biomass to carbon equivalent based on a ratio of 0.512. Soil carbon is not explicitly tracked in the carbon charts because research suggests that on average, there is little change in soil carbon under active forest management (Johnson and Curtis 2001; Ter-Mikaelian et al 2008), even when transitioning from old- to second-growth conditions (Fredeen et al 2005) unless there is application of urea fertilizer (Canary et al 2000 Adams et al 2005) that increased soil carbon accumulations along with tree biomass. Use of woody debris as a bioenergy source material may alter the assumption of no net soil carbon change (Johnson and Curtis 2001), but removal of waste materials was not explicitly included in this analysis because there are currently no markets for this material. Fire may also reduce soil carbon either from volatilization during the burn (Bormann et al 2008) or by erosion in the immediate period after vegetation loss (Helvey et al 1985; Baird et al 1999).

Regional milling surveys were used to allocate logs into long-, medium-, and short-lived product uses and to calculate displacement carbon based on energy equivalents and the amount of hog-fuel and mill residuals used to displace fossil fuels that would be otherwise used in milling processes for INW mills (Puettmann et al 2010a). End of life for long-lived product uses was estimated to be 80 yr (Perez-Garcia et al 2005b; Winistorfer et al 2005). No recovery or landfill sequestration of embodied carbon was considered to be consistent with Perez-Garcia et al (2005b). Skog (2008) using different techniques does establish that landfills do provide positive carbon stores net of emissions, hence this assumption will understate product carbon stores. Substitution carbon in the initial CORRIM research was derived from the differential between the carbon footprint of a wood-framed house and their dominant concrete frame substitutes (Perez-Garcia et al 2005a). The substitution parameter varies by material end use and will likely increase with the price of fossil fuel or value of carbon because this would motivate greater use of relatively lower-cost wood in applications in which fossil-fuel intensity and cost are highest. Sathre and O'Connor (2008) developed a meta-analysis of all substitution studies in the literature and concluded the average substitution rate was 2.0

displaced units of carbon for each unit of carbon in the wood used. We use this parameter to identify the carbon benefit in substitution for that portion of the wood basket that is milled into long- and medium-lived solid wood products. This substitution rate is 20% lower than the substitution derived in prior CORRIM studies for wood substitution for concrete walls as the dominant form of substitution (Perez-Garcia et al 2005a). Short-lived products such as chips for pulp and paper are decomposed rapidly, although the biofuel portion generally displaces fossil fuels producing another conservative underestimation of carbon stores and offsets.

Simulations generated estimates of carbon production in the forest, product, displacement, and substitution pools in decadal time steps. Because harvest operations and forest growth have yearly incremental impacts, we allocated the 10-yr increment to yearly time steps to show the time series of carbon change over a 110-yr period and summarized results based on a 100-yr period.

Fire risk and fire rate. Simulations of wildfire impacts used two approaches, each with simplifying assumptions. For both approaches, area burned is distributed across the landscape so that the carbon in the average hectare is reduced proportionately to the area affected. This assumption may underestimate greenhouse gas emissions and forest carbon impacts because stands with higher fuel loads and correspondingly higher carbon equivalents tend to have a higher fire risk and therefore may be disproportionately represented in the area burned.

For bounding the range of fire rates, we assume that all fires are stand-replacing events that overestimate the increase in the dead pools, offsetting the first assumption to some degree. We use the base case forest management and netted out wildfire impacts on forest carbon using variables from Wiedinmyer et al (2006). Wiedinmyer data estimate an average fuel consumption of 30% for the woody materials and 90% for the litter when weighted across the five Global Land Cover classes in the region.

For calculating fire impacts under the two management scenarios, we simulated wildfires in each decade using FVS-FFE model default parameters for severe fire weather conditions. Simulations of wildfires and no fire were allocated as a percentage of the expected fire rate during that period. For example, if 50% of the area was expected to burn over a 100-yr period, 5% of the area would be allocated to burn in each decade of the simulation. This approach ensured that FFE outputs for stand mortality and volume losses were weighted by decade with all land having an equal likelihood of burning (or not burning) in each period. Each simulation reports results on a 10-yr interval with fires initiated after treatments during that decade of the simulation.

With the substantial changes in recent fire rates and the large uncertainties around future rates, it is not practical to describe a confidence range for all fire rates. Instead, we established a range of impacts selecting a predicted rate as a benchmark for comparison. We applied a weighted average of the McKenzie et al (2004) future climate predictions by state to the historic FIA fire rate estimate to arrive at a benchmark 21st century fire rate. With this estimate, 67% of the forest will burn at least once during the 100-yr period. This rate is likely conservative because 59% of the area is in dry and moist forests that naturally would be expected to burn at least once every 100 yr and as often as every 6 – 25 yr in the case of dry forests. In addition, the weighted average age for high-elevation forests with a fire return interval of 100 – 300 yr is currently 145 yr. That would suggest at least 50% of those forests are due to burn within the next 100-yr period making the more likely fire rate 80% or higher if we account for multiple burns in low-severity fire regimes. Although no attempt is made to model the stochastic nature of fire, applying the average expected fire rate with climate change provides a useful sensitivity range on impacts. This method of analysis identifies the limits to carbon accumulation associated with climatic controls on fire behavior without explicitly evaluating how increasing inventory leads to increasing fuel loads, fire risk, and stand-replacing events.

RESULTS AND DISCUSSION

State and Private Management Impacts on Landscape-Level Carbon Pools Over Time

Results are provided by tracking the carbon in forest and product pools over time, showing the average per hectare carbon in each pool for all stands in each owner group. Because downed wood inventories were not consistently available across the entire data set, estimates of downed wood for the initial period were not calculated but are included when harvest or wildfire occurred. Total carbon for the composite stand represented by the aggregated FIA data for that owner group includes carbon in the forest, and in nonforest pools when treatments occur, with accumulations and losses accruing through time as management occurs.

The base case for the simulated management of State and Private commercial forests is calibrated to a harvest rate consistent with the rate over the past 30 yr. Figure 1 shows the base case carbon storage for both the forest and other pools that would ensue under current marketing and infrastructure conditions from the sustainable harvest of INW State and Private Forests. The total forest carbon is the weighted average of forest components, including the live stem, roots, crown, litter, and slash but not including soil carbon or estimates of large woody debris at the beginning of the simulation. The simulation covers all projected harvest volumes for the State and Private Forests in the base case. Forest carbon begins at 44 t/ha and increases to about 74 t/ha after 100 yr of active management. The average forest carbon for all decades is 57 t/ha.

During all decades, the forest is composed of a range of stands at every age and although stands are continually harvested, the overall average inventory volume and thus carbon stored remains relatively stable. Decreases in forest carbon in any particular time period reflect shortfalls in available inventory relative to his-

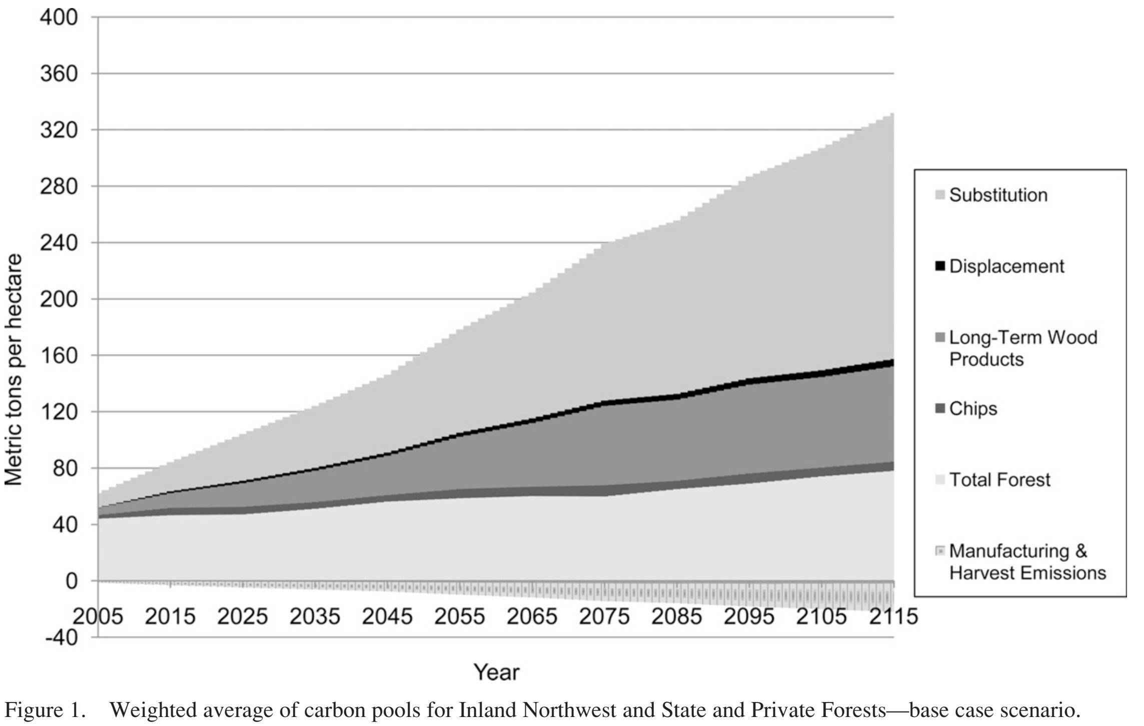

Figure 1. Weighted average of carbon pools for Inland Northwest and State and Private Forests—base case scenario.

toric harvest rates. Increases in forest carbon indicate that projected growth exceeds historic harvest rates during that time period. The aggregate per hectare values shown in Figs 1 and 2 can be multiplied by total area under management to obtain a landscape-level carbon impact of forest management across State and Private land holdings in the INW. These figures illustrate that under current management strategies, there is the potential to increase carbon stored in the forest if harvest continues at current rates. Some increase in forest carbon should be expected from increasing harvest constraints to meet regulatory requirements. Every carbon pool is simulated dynamically to grow or die and decompose or reflect harvest removals. The overall effect is a slight increase in forest carbon from these management simulations. The simulations are consistent with the assertion that the overall impact of sustainable forest management activities on forest carbon is at least neutral (or in this case positive) when assessed at the landscape level.

Although carbon in the forest remains relatively stable, the carbon in products continues to grow with each year's harvest when even a portion of the harvested log is used in long-lived applications. Based on mill survey data for the region (Puettmann et al 2010a), approximately 50% of the harvested logs are used in long-lived applications. Energy used to grow, harvest, transport, and process the logs into wood products is identified as manufacturing and harvest emissions in Fig 1 and subsequent figures that show all carbon pools. These emissions are small relative to the carbon stored in products and partially offset by using wood energy rather than fossil energy in manufacturing. The substitution for fossil fuel-intensive products has the highest leverage given the high fossil intensity of the nonwood substitute products. Figure 1 shows that even with increasing forest carbon, the carbon in products, displacement, and substitution pools grows sustainably and substantially exceeds the carbon storage potential of the forest alone by the end of the simulation period. This is true

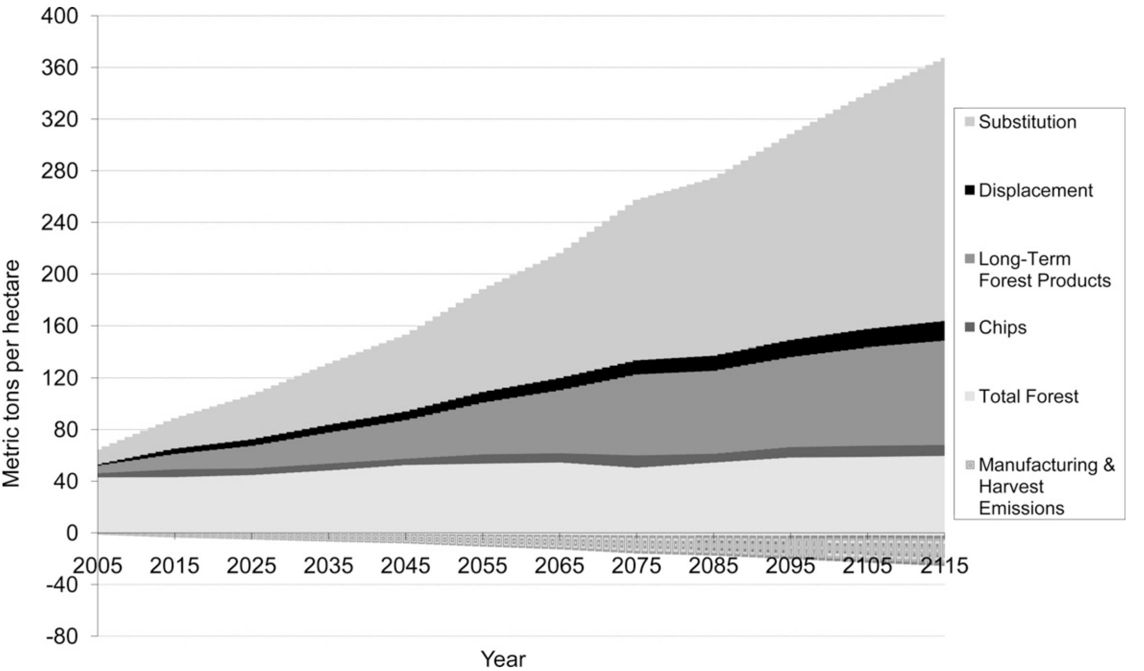

Figure 2. Weighted average of carbon pools for all State and Private Forests—alternate case.

even when accounting for the energy costs of producing products that is shown as a carbon cost below the x-axis of Fig 1. It also ignores the potential to recycle products, capture an end-of-life energy offset by using waste as a biofuel, or account for sequestration of the product carbon in a landfill. Whereas average forest carbon for the simulation period was 57 t/ha, accounting for the remaining pools (net of their emissions) increases the average carbon stored over the 100-yr period to 175 t/ha for the base case for State and Private owners. This 3-fold increase in carbon reflects the potential of well-managed forests to contribute positively and sustainably toward offsetting global warming trends. Most of the increase derives from the accumulation of long-lived products, displacement of fossil fuels directly, and by substituting solid wood products for fossil fuel-intensive products like steel and concrete. Although the carbon stored in the forest remains fairly stable for this treatment regime, the carbon storage and offsets in pools outside the forests continues to grow as long as the harvests continue.

The State and Private Forest base case harvest level used for the simulation in Fig 1 was based on the average harvest rate over the past 30 yr, suggesting that although there is no retroactive reporting of the nonforest carbon pools from earlier harvests, we could expect that at least 75 t/ha (not shown) would have already been sequestered in offsite uses from just the harvest outputs from the 30 yr before the beginning of the simulation in 2005. The harvest translates into substantial and sustainable increases in products carbon and substitution. Substitution averages 78 t/ha over the 100-yr period or 37% higher than the average forest carbon storage. The product pool surpasses the forest as a carbon storage pool after 40 yr and averages 45 t/ha over the 100-yr period or 79% of the total forest carbon. By 100 yr, the 294 t of carbon stored in all pools is 4 times greater than the carbon stored in the forest and the average growth in all carbon pools is 2.3 t/ha·yr. The long-lived products have lives modestly less than 100 yr and the short-lived products were assumed not to result in significant accumulation of carbon from one rotation to the next. From the perspective of managing a unit of land, the carbon pools supported by the forest are much better than permanent, increasing sustainably at 2.3 t C/ha·yr.

Given the modest increase in forest carbon for the base case, which infers that harvest is less than sustainable growth, an increase in harvest and management intensity was simulated as an alternate case. Figure 2 demonstrates the weighted average of carbon sinks and sources on all State and Private Forests under the alternate case where harvests are increased with concomitant increases in regeneration investment. The potential gain in carbon storage under more intensive management is approximately 9%.

Although carbon gains under the alternate scenario on State and Private Forests are possible, they require substantial increases in intensity in areas that have not historically been high timber-producing regions. The potential for increased intensity on State lands is more pronounced given the historic low intensity of management and the substantial mature inventories that they currently carry. Maximizing volume through increased management intensities may not necessarily produce a greater net present value using historic prices, because the higher volume and value tends to be generated later in the simulation period, whereas investment costs in planting and stand-tending occur early on in the simulation. A sensitivity analysis of the potential for increasing forest productivity using fertilization (not shown) suggested that substantial gains would only accrue if large areas were fertilized to increase productivity and shorten rotations. This option was not considered economically likely and was not included in the landscape level modeling.

Fire and National Forest Carbon Sequestration and Storage Potential

The impact of five different estimated fire rates on forest carbon in 21st century National Forests under current management intensities is shown in Fig 3. The Littell et al (2009b) estimate based on historic wildfire occurrence and the FIA estimate based on analysis of stands younger than 100 yr provide close agreement on the region-wide 20th century fire rate. These trend lines show how a continued, ongoing, and expensive wildfire suppression effort might permit a continuing increase in forest carbon net of carbon lost to fires and dead wood decay in the 21st century. However, wildfire suppression outcomes in the first decade of the 21st century suggest that our success at containing fire impacts to the 20th century rate is highly unlikely. The average yearly area burned over 2002 – 2009 would equal the Littell et al (2009b) 20th century area burned by 2017 and the FIA historic area burned by 2019 if that fire rate were sustained without change over the next 8 – 10 wildfire seasons.

As an attempt to understand how projected climate change scenarios would impact fire rate, we use McKenzie et al (2004) projections weighted for the three states in the INW region that predict approximately 2.25 times more wildfires than 20th century conditions. We applied this increase to the FIA estimate as a future fire rate that assumes continued fire suppression activities and label it as a doubling of the FIA historic fire rate (Fig 3). We also show the Camp et al (1997) fire rate for mixed-severity fire regimes under presettlement climate and management conditions. We would not expect a trend increase in forest carbon in presettlement times and the simulation appears to be consistent with that expectation. We also show a doubling of the Camp fire rate because it is the closest fire rate to what we have experienced in the first decade of the 21st century (2002 – 2009). Most of the increased forest carbon modeled under a 20th century fire rate assumption and fire suppression strategy is lost under a presettlement fire rate or under the doubling rate predicted using McKenzie et al (2004) climate change scenarios. The most extreme

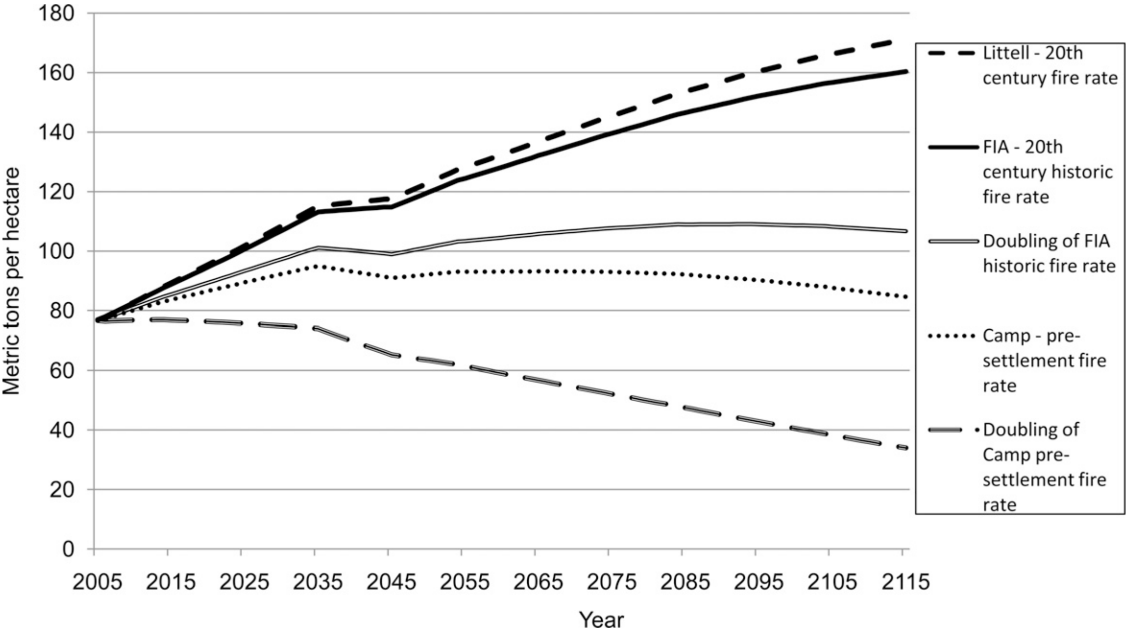

Figure 3. The impact of fire rate on future forest carbon storage potential. National Forest management simulations showing trends through time with fire rates of 24% (Littell et al 2009b showing 20th century top-down estimates), 30% (FIA 20th century from inventory data), 67% (FIA with increased rate from McKenzie et al 2004 21st century climate regimes), 88% (Camp et al 1997 presettlement estimate for low and mid elevation sites), and a doubling of the Camp presettlement fire estimate.

impact is for a doubling of the presettlement rate (akin to the hectares burned at a fire rate consistent with 2002 – 2009 actual fire occurrence), which, if sustained, would reduce forest carbon by over one-half the current storage in live trees within 100 yr.

All fire rates shown in Fig 3 assume that the dominant fire event is a stand-replacing event given the increasing fuel continuity from decades of fire suppression (Cromack et al 2000; Sampson et al 2001) and increasingly heavy fuel loads from large insect outbreaks (Oneil 2006) and other mortality events. Although individual fires may not be stand-replacing, the carbon impact of stand-replacing fires at high elevations where there is higher than average carbon storage largely offsets the differential as we found when applying the same fire rate and assessing impacts using the FVS FFE model (Reinhardt and Crookston 2003).

To assess how wildfire occurrence would affect product recovery, we use the FFE model to predict stand mortality and consequent forest carbon through time under a 21st century fire regime and concomitant management activity. Using this approach, fire impacts integrate stand characteristics, including species composition, density, stand structure, average diameter, and forest canopy bulk density, to arrive at expected mortality. However, FFE does not account for contagion between stands. We used a doubling of the FIA historic rate based on the McKenzie et al (2004) estimate of expected fire rate increase for the 21st century by state and analyzed the impact of treatment alternatives. Figures 4 and 5 use the FFE model to predict mortality and consequent product impacts under the doubled fire rate. The forest carbon in Figs 4 and 5 is slightly greater than that predicted from assuming a weighted average impact of all land with stand-replacing fires as is shown in Fig 3 for the doubling of the FIA historic rate. This is consistent with the expectation that FFE would predict less wildfire mortality in thinned stands than an assumption of stand-replacing fires across all

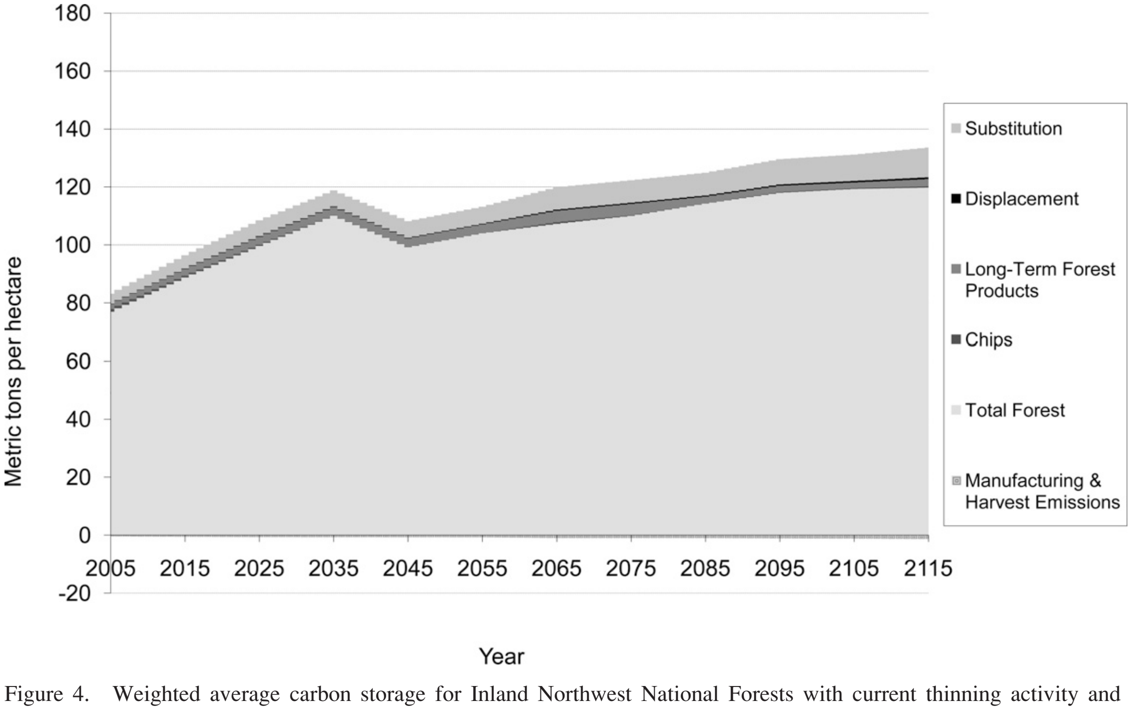

Figure 4. Weighted average carbon storage for Inland Northwest National Forests with current thinning activity and expected 21st century fire impacts.

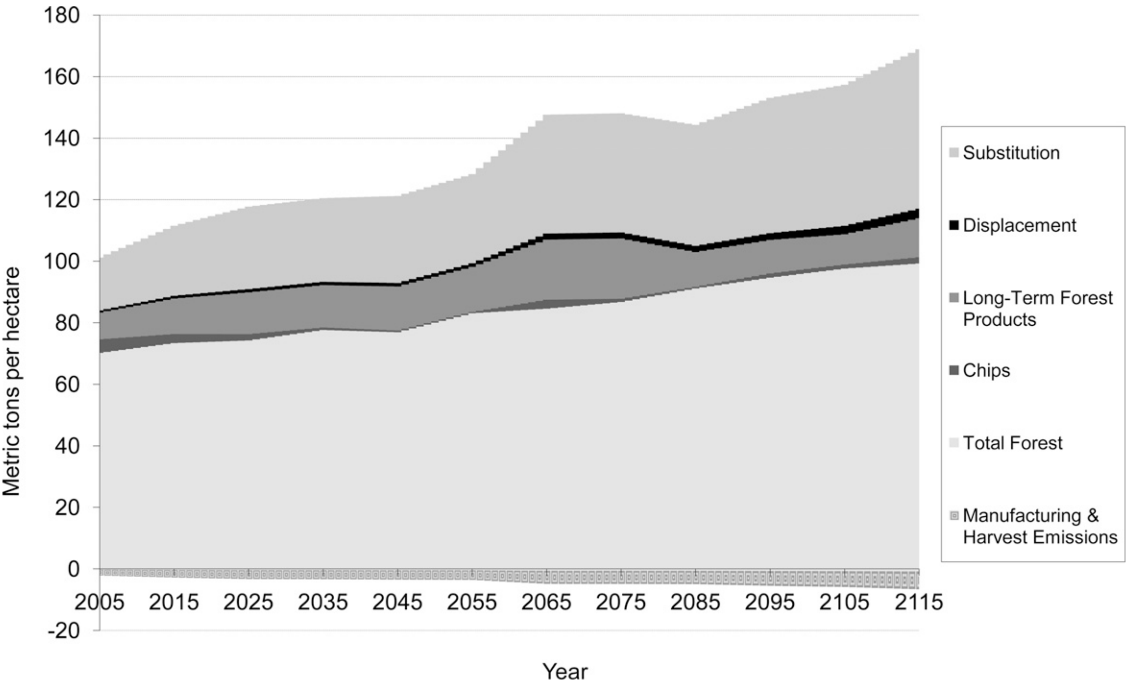

Figure 5. Weighted average carbon storage for Inland Northwest National Forests with increased thinning activity and expected 21st century fire impacts.

land. However, the differential is not large. For the base case, the FFE model predicts only 4 t/ha more forest carbon remaining on-site on average over 100 yr and only 11 t/ha more at 100 yr relative to the assumption of stand-replacing fires on all burned areas.

Although climatic controls have been implicated as the dominant predictor of area burned, thinning treatments that reduce fire risk have a high potential to reduce carbon emissions both by reducing wildfire intensity and/or area and in sequestering carbon in the products produced from the material removed (Mason et al 2006; Lippke et al 2008). However, thinning treatments need to be introduced rapidly to reduce fires significantly and the carbon accounting and methods to motivate such treatments are complex, involving the type of treatment, the rate treatments are phased in, owner objectives, and the use of wood removed from the forest (Lippke et al 2008).

Treatment alternatives for National Forests are more complex than for State and Private Forests because they are managed for forest health improvement and fire risk reduction rather than for maximum sustained yield or revenue. Figure 4 shows the carbon consequences of thinning stands at current levels of management intensity (base case) and Fig 5 shows the impacts of treating additional area in dry and moist forests as well as capturing the mortality in lodgepole pine stands at high elevations as an alternate case. The alternate case is approximately a 400% increase in the amount of area treated. In both the base and alternate case, the assumption of thin-from-below treatments to a 300 mm DBH limits results in removal of few merchantable products. Because the thin-from-below treatments generated few large sawlogs and over 45% pulpwood, harvests generated few long-lived products with corresponding small long-term product and substitution pools.

Alternate scenarios that rapidly thin National Forests contribute substantially more carbon to product pools than base case scenarios even when we remove only those trees less than

300 mm dia. Additional carbon gains could be obtained by removing larger diameter trees using proportional thinning approaches that take trees from all size classes, produce more merchantable logs, and subsequently more long-lived products and offset benefits. Bonnicksen (2008, 2009) concluded that a proportional thinning approach was necessary to reduce fire risks in old forests studied in California. Thinning treatments that are not strictly limited to trees less than 300 mm DBH or that remove greater amounts of merchantable material will increase the carbon pools outside of the forest. They are also likely to reduce the risk of fire (Mason et al 2006), increasing forest resilience and reducing climate change related stress, depending on climate sensitivities and stand-carrying capacity.

For both National Forest scenarios, there is rapid implementation of the first thinning to capture current mortality followed by a second entry with fewer residuals. We do not model salvage of mortality because of the long planning horizons required to offer a timber sale contract (USFS 2002) and the loss of timber value that ensues during the delay. Thinning all land in low- and mixed-severity fire regime habitat types in the National Forests (Fig 5) reduces the fire risk on treated stands and transfers more of the excess inventory into product carbon pools. However, total carbon, including the forest, wood product, displacement, and substitution pools, only reaches 152 t/ha in 100 yr with most of that still in the forest as compared with 317 t/ha from accelerated sustainable harvests on State and Private lands. This 152 t/ha also includes gains from harvesting lodgepole forests that are currently being lost to MPB. For these reasons, the National Forest alternate scenario is likely quite optimistic because the inventory has been growing with an increasing fire risk and MPB mortality is continuing unabated. Because the carbon across all pools with increased thinning (Fig 5) is only 14% larger than the potential carbon with less thinning (Fig 4), the benefits of increased thinning will largely be in the reduced losses from a climate related increase in fire rate.

Under both the National Forest base and alternate scenario, fire impacts are less than the volume increases predicted by the FVS growth model even with assumptions of a doubling of the FIA historic fire rate. Because the average age on these forests is over 100 yr and largely mature trees remain after treatments and wildfire, the volume and carbon increment over the next 100 yr is largely unexpected. It could be related to model parameterization based on growth rates in stands less than 100 yr or it could reflect the impact of regeneration coupled with increasing increment on surviving overstory trees. Although increases in tree density from fire suppression have been well documented, the coincident increase in large tree mortality has been shown to largely offset long-term volume and carbon accumulation (Everett et al 2008; Fellows and Goulden 2008) in some forests.

Whereas normal management treatments have shown no substantial impact on soil carbon (Johnson and Curtis 2001), severe wildfires have the potential to substantially reduce soil carbon. The simulations do not capture the impact of more severe wildfires, which can remove over 25% of the soil carbon (Bormann et al 2008) and where postfire hydrophobic soils can result in massive erosion events (Helvey et al 1985), both of which reduce soil carbon, forest productivity, and subsequent carbon gains. Because soils contain a substantial amount of the total carbon in the forest (Bradford et al 2009), it becomes increasingly apparent how important it is to reduce the extent of severe wildfires caused by excessive fuel loads.

The interaction between wildfire and permanent loss of, or change in, forest cover is also a concern in the INW. In the Southwest, forest dieback events associated with climate change have already occurred (Allen and Breshears 1998) and more are predicted (Breshears et al 2009). Models suggest that one or more tree species will be lost in parts of the INW in the coming decades with wildfire being the most likely mortality agent, although insect outbreaks may also play a role (Rehfeldt et al 2006; Littell et al 2009a).

Extreme weather events, wildfire, insect outbreaks, and poor seed crops interacting in a disturbance complex may permanently alter the forest structure. Although our modeling suggests the forest will continue to slowly accumulate carbon over the next century even with an altered fire regime, there are outstanding questions of whether productivity and carbon sequestration will continue at the same rates as seen today given the uncertainties in fire rate estimation.

Comparison Across Ownerships

By characterizing the impacts of alternative management and fire rate assumptions on a per hectare basis for all carbon pools as shown in Figs 1 – 5, opportunities are revealed for greater carbon storage for the INW region. Table 2 summarizes the changes in forest carbon and total carbon by owner and management scenario at two points in time. The first point, 2005, reflects the completion of the first decade harvests and treatments. The second point, 2105, reflects the impact of current management (the base case) or alternate management (increased harvest volumes from nonfederal and increased harvested land on National Forest lands) 100 yr later. All values are provided for the total landscape that takes into account that 63% of the land is on National Forests and 37% is under State and Private ownership.

After the first decade of harvest, State and Private management has resulted in 44 t/h of carbon in the forest and 61 t/ha of additional carbon, including net product stores and substitution and displacement offsets under the base case scenario. The total carbon benefit from managing State and Private Forests grows to 294 t/ha in 100 yr for a 2.3 t/ha·yr increase. After the first decade of harvest, National Forests have 77 t/ha carbon in the forest and 83 t/ha of total carbon, including net product stores and offsets, under the base case scenario. The total carbon benefit from managing National Forests grows to 130 t/ha in 100 yr for a 0.5 t/ha·yr increase. The differences in the total carbon across all pools for State and Private Forests relative to National Forests highlights the importance of the long-term product pool and its carbon substitution benefit.

Under the alternate scenario, the State and Private harvest rates and regeneration activities would increase, resulting in a decline in the rate of accumulation of forest carbon. The net effect would be an increase of 37% in forest carbon over 100 yr down from a 68% increase under the base case. Increasing the rate of federal thinnings by 400% (from 20% of eligible stands treated/yr to 100% of eligible stands treated/yr) results in a 39% increase in forest carbon in 100 yr down from 56% under the base case. Comparing time periods shows that after 100 yr under the alternate scenario, total carbon across

Table 2. *Changes in total carbon in the forest and other storage pools by owner and scenario.*

	Total carbon in (Mt) by owner group and scenario											
	Forest carbon						All carbon pools net of processing					
	S&P base	NF base	Total	S&P Alt	NF Alt	Total	S&P base	NF base	Total	S&P Alt	NF Alt	Total
2005 carbon inventory	287	847	1135	280	773	1053	399	911	1310	411	1091	1502
Difference (Alternate–Base)				−8	−74	−82				12	180	191
2105 carbon inventory	483	1314	1797	383	1075	1458	1918	1431	3350	2065	1668	3733
Difference (Alternate–Base)				−100	−239	−339				147	237	384
2005–2105 change	196	466	662	103	302	405	1519	520	2039	1654	577	2231
Increase over 100 yr (per ha)	30	42	38	16	27	23	233	47	116	254	52	127
Increase (ha/yr)	0.3	0.4	0.4	0.2	0.3	0.2	2.3	0.5	1.2	2.5	0.5	1.3

S&P, State and Private; Alt, alternate scenario; NF, National Forest.

all pools is increased by 52% on the National Forests and by 402% on the State and Private Forests. The alternate scenario for State and Private Forests has 317 t/ha of carbon across all pools after 100 yr for a 9% increase as compared with the base case. For National Forests, there are 152 t/ha under the alternate case after 100 yr for a 17% increase over the base case. Although the alternate scenario for National Forest management can reach another 5 Mha that are suitable for treatment under a fire and insect risk reduction umbrella, retention of large trees in the overstory limits the amount of product carbon and substitution offset potential.

Table 2 shows the forest carbon share of total carbon is reduced on both State and Private and National Forests under the alternate scenarios (–21% State and Private and –19% for National Forests), transferring more carbon to product and offset pools (+17% State and Private, +409% National). For State and Private Forests, the increased harvest rate under the alternate case increased the ratio of carbon outside of the forest to forest carbon from 3:1 on the base case to 4.4:1. By comparison, the National Forest alternate scenario increased the ratio of carbon outside of the forest to forest carbon from only 0.1:1 to 0.6:1. Sustained increased fire rates such as experienced since 2000 are beyond the doubling illustrated here as the base case for the 21st century would substantially reduce these gains.

When summed over the 6.5 Mha of State and Private land, the alternate scenario sequesters and stores approximately 1.65 Gt carbon over 100 yr. On National Forests, carbon sequestration, storage, and avoided emissions over the 11 Mha of unreserved forest land could remove 0.5 – 0.65 Gt of carbon depending on the management scenario. Table 2 shows that there is the potential for a cumulative 2.2 Gt increase in carbon across all land and all pools over 100 yr. That potential carbon gain is made up of a:

1. 68% State and Private sustained harvest with current management strategies and wood uses;
2. 6% increase from increased harvesting under the State and Private Forests alternate scenario;
3. 14% increase from more forest carbon stored on National Forests; and
4. 12% increase in National Forest nonforest carbon pools and offsets.

SUMMARY AND CONCLUSIONS

In INW, where the forest land base is dominated by federal ownership and the forests are managed for a multitude of benefits, LCA suggests that the optimal solution for maximizing carbon gain under both current and future climate conditions is to manage forests to maximize long-lived wood products and to minimize the risk of severe wildfires. The carbon storage in buildings and the substitution benefits therein override the potential gains of attempting to leave high carbon stocks stored in the forest. This result is augmented by the fact that leaving the carbon in the forest increases the risk of wildfires and thus carbon emissions and eliminates the opportunity to use the material lost to fires to displace fossil fuel-intensive products.

Our results suggest that if we rely solely on the forest as a carbon sink, there are high risks that the total sink will be less than it is today. If we rely on the forest to remove CO_2 from the air and sequester it in woody biomass, and then remove merchantable growth and turn it into long-lived products, even if the inherent site productivity declines with climate change, or wildfires burn more than the yearly growth, the forests and their outputs will collectively remain a growing carbon sink rather than a carbon source.

The principal driving factors affecting carbon pools in the INW region are:

1. Sustainable harvest of trees sufficiently large to produce long-lived structural materials maximizes carbon storage across all pools. In addition, the maximum carbon mitigation benefit derives from harvesting the forest before growth slows down.

2. State and Private Forests are harvesting near their maximum sustained removal rates resulting in carbon pools outside the forest growing to more than 3 times greater than the carbon stored in the forest.
3. National Forest thinnings designed to retain large overstory trees help offset the decline in forest carbon from wildfire. However, to gain even a small portion of the carbon benefits provided by State and Private Forests, some increase in larger tree removals would be necessary to produce long-lived products that store carbon and substitute for more fossil fuel-intensive products.
4. Maintaining high-density forests in fire-prone regions is not a carbon maximization strategy, although fire suppression may have contributed to increased forest carbon in recent decades. With climate change, the carbon mitigation potential of old forests degrades rapidly as forests become a source rather than a sink for carbon. The point at which forests become a carbon source rather than a carbon sink is a function of fire rate, the kinds of products that can be produced from the forest, and whether effective fire reduction treatments can be implemented. If 2002 – 2009 fire rates on INW National Forests are sustained into the future, our trend analysis suggests that they have already become carbon sources rather than sinks. If the fire rate increases to a presettlement fire rate, then carbon accumulation will cease within the next few decades as fire impacts offset growth.

There is a great need to integrate and assess how treatments can be used to mitigate the impacts of increasing areal extent of wildfire caused by climate change. Here we show the likely outcomes if we continue business as usual under an altered fire regime. Sensitivity analysis that incorporates impacts on fire severity, contagion, and associated mortality into higher fire rate scenarios is needed to determine the best treatments for different forest types and how much land would need to be treated to maintain forests as carbon sinks. Regardless of treatments, we can expect that forests of the INW will store less carbon in the future under an altered fire regime. However, they can be managed to efficiently sequester carbon in wood products and offset carbon through substitution as well as reduce disturbance risks.

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