EXHIBITS 9-16

Exhibits to Center for Biological Diversity Comments
EPA Call for Information on Greenhouse Gas Emissions
Associated with Bioenergy and Other Biogenic Sources
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Review of the Manomet Biomass Sustainability and Carbon Policy Study

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For the Clean Air Task Force
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EXECUTIVE SUMMARY

The purpose of this document is to evaluate the science behind the Manomet biomass report and the validity of the report's main conclusions concerning net carbon emissions from biomass energy, relative to fossil fuels. The report comes to two main conclusions:

1. **For utility-scale generation, net emissions are higher from biomass than fossil fuels.** When biomass is used to generate electricity in utility-scale plants, the net emissions after 40 years, even taking forest regrowth into consideration, are still higher than if the power had been generated with coal. When biomass is used instead of natural gas, net emissions are still higher even after ninety years (exhibit 6-14, p. 112).

2. **Net emissions profiles from biomass thermal and CHP plants may be better.** The Manomet study concludes that when biomass replaces fossil fuels for small-scale thermal applications and in combined heat and power plants, net emissions by 2050 can be lower than would occur if oil had been burned, but are still significantly higher than if natural gas were used as fuel.

The study relies on a number of assumptions to achieve these conclusions that minimize the calculation of net carbon emissions from biomass power, meaning that actual emissions are likely greater than the study concludes. Thus, the first conclusion of the report – that net emissions from biomass are greater than from coal and especially natural gas even after decades of regrowth by forests – is qualitatively correct, but it likely underestimates the magnitude of biomass emissions. The second conclusion, that small-scale thermal and CHP biomass facilities may yield a carbon "dividend" relative to fossil fuels after forty years is likely not correct, since actual biomass emissions likely exceed fossil fuel emissions even under the thermal and CHP scenarios.

The study's major conclusion, that net biomass emissions are significantly higher than if natural gas were used as fuel even after ninety years of forest regrowth, is especially notable for the New England area where the majority of electricity generated comes from natural gas. Using biomass to "reduce" emissions from the power generation sector will have the opposite effect, particularly where biomass displaces power generation from natural gas.

The Manomet model estimates net carbon emissions for both biomass and fossil fuels as fuel lifecycle emissions minus forest carbon sequestration on a hypothetical acre which is cut for timber but not biomass (the fossil fuel/business-as-usual scenario), and one which is cut for both timber and biomass (the biomass scenario). As the forest regrows on the plot cut for biomass, the net carbon balance transitions from representing a "carbon debt" to providing a "carbon dividend", as carbon moves from the atmosphere into new forest growth. This "single plot" analysis of forest recovery after cutting serves as the building block for an integrated analysis, which assesses the cumulative impact of a biomass industry that cuts new forest for fuel each year and thus increases the relative amount of land that still has a "carbon debt". The study unfortunately downplays the cumulative effects analysis, instead focusing on the "single plot" analysis, which would only be relevant to the calculation of carbon impacts from a facility that operated for a single year.

Some of the many assumptions upon which the Manomet study's conclusions rely are listed here; all minimize the calculation of carbon emissions from biomass. The model is sensitive to these assumptions, therefore if any one of them is violated in reality, actual emissions will be greater than reported in the study's conclusions.

1. **Large trees are used for biomass fuel**. Because forest regrowth rates in the model are to a large extent a function of the intensity of harvest (with heavier harvests of larger, older trees opening up more space for regrowth to occur), the model achieves maximal regrowth and resequestration of carbon released by biomass burning by assuming that relatively large, old trees are logged for biomass. However, this is not representative of actual biomass harvesting, which is more likely to remove low-diameter, low-value material. Actual regrowth rates of forests where low-diameter material is removed will be much slower than modeled.

2. **Harvested forest stands must not be recut pending carbon sequestration.** The model additionally requires that once a stand has been cut, it must not be re-cut until it has achieved a large proportion of the amount of standing carbon in an unmanaged stand. The Manomet report itself acknowledges this is unlikely.

3. **A high percentage of tops and limbs are used as fuel**. Because the tops and limbs of trees harvested for timber under the BAU scenario are assumed to stay in the forest and rot, producing carbon, the model assumes almost no carbon penalty for collecting this material and burning it. The model assumes that 65% of all tops and limbs generated on acres harvested for biomass can be removed from the forest for use as fuel, supplying a relatively large "low carbon" source of fuel in the model. Removal of this amount of tops and limbs appears to be necessary to achieving the transition from biomass carbon debt to carbon dividend in the model, but is not compatible with maintaining soil fertility and other forest ecological functions.

4. **Biomass harvesting only occurs on land that is already being harvested for timber**. The study takes as its BAU assumption that when land is harvested for timber, all residues are left in the forest, whereas a portion is collected for fuel in the biomass scenario. The study draws no conclusions concerning carbon dynamics and regrowth in forests cut solely for biomass. This assumption is necessary for generating the "low carbon" fuel source of tops and limbs from commercial timber harvesting that is integral to calculating carbon dividends from biomass in a timely way. Land cut solely for biomass would take a much longer time to achieve a carbon dividend.

5. **Soil carbon emissions are negligible**. The soil carbon pool is extremely large, and a significant fraction of it is easily decomposed and evolved as CO2 when soils are disturbed by logging. However, the Manomet model completely disregards this source of emissions that are associated with biomass harvesting. This assumption is challenged by the author of a major review on soil carbon emissions cited, and dismissed, by the Manomet study.

6. **Firewood harvesting is not impacted**. Although indirect land use effects can be major sources of greenhouse gas emissions from biomass harvest, and although the RFP for the Manomet study requested that the study evaluate indirect land use effects, the study does not acknowledge that displacement of firewood harvest by biomass harvest could result in "leakage" of firewood harvesting and more forestland being cut for firewood.

7. **Wood pellet manufacture incurs no more carbon debt than green chips**. Although it is well-established that manufacture of wood pellets requires significant inputs of green wood in excess of the heating value actually embodied in the pellets produced, as well as significant fossil fuel expenditures, the study treats wood pellets as embodying the same amount of carbon and energy as green wood chips.

8. **Wood from land-clearing incurs little carbon debt.** The study concludes that woody biomass from non-forestry sources, such as from land-clearing, will not entail any greater greenhouse gas emissions than forestry wood. However, no modeling is conducted to substantiate this conclusion. The study also does not discuss how wood from land-clearing can be considered eligible under requirements that biomass fuels be available on a renewable and recurring basis, as required under the Regional Greenhouse Gas Initiative.

To the extent that these assumptions are not warranted, the Manomet study has underestimated the net carbon emissions of biomass power, and policy-makers should be extremely cautious about accepting the study's optimistic conclusions concerning the point in time when biomass can start providing a carbon dividend.

INTRODUCTION

The purpose of this document is to evaluate the science behind the Manomet biomass report and the validity of the report's main conclusions. The Manomet study is large, and covers much background material on biomass policies in the United States and internationally. This evaluation will focus only on the core conclusions of the study that deal with carbon accounting. Overall, the conclusion of this evaluation is that the Manomet study's basic approach to calculating net carbon emissions from biomass is valid, but it relies on a number of overly optimistic assumptions and omits categories of greenhouse gas emissions from the study's lifecycle analysis. It is highly likely that net carbon emissions from biomass are actually higher than concluded by the Manomet study.

Organization of this paper
This summary reviews the carbon modeling aspects of the Manomet report. It begins by setting out the two main conclusions of the study. This is followed by an explanation of how the Manomet carbon model was constructed.

Next is a short list of the main assumptions of the model, upon which the conclusions depend. This is followed by a critique of each assumption.

Once the assumptions behind the modeling are aired, this allows the conclusions of the Manomet study to be assessed more thoroughly.

Throughout, this summary paper relies extensively on text copied from the Manomet report itself, with page numbers included to guide the reader to relevant sections. Points of particular importance are highlighted.

MAIN CONCLUSIONS OF THE MANOMET STUDY

Regarding net carbon emissions from biomass relative to fossil fuels, the study had two main conclusions:

1. **For utility-scale generation, net emissions are higher from biomass than fossil fuels.** When biomass is used to generate electricity in utility-scale plants, the net emissions after 40 years, even taking forest regrowth into consideration, are still higher than if the power had been generated with coal. When biomass is used instead of natural gas, net emissions are still found to be higher after ninety years (exhibit 6-14, p. 112).

2. **Net emissions profiles from biomass thermal and CHP plants may be better.** The Manomet study concludes that when biomass replaces fossil fuels for small-scale thermal applications and in combined heat and power plants, net emissions by 2050 can be lower than would occur if oil had been burned, but are still significantly higher than if natural gas were used as fuel.

Prior to further discussion, it is important to note that the results presented in the executive summary of the Manomet report do not represent the full results presented in the body of the report. Most importantly, the study concluded that the net carbon balance of biomass energy depended on the intensity of harvesting both for commercial timber and biomass removal itself, and thus examined six different harvesting scenarios, reporting the carbon balance results under each. Unfortunately, the results of only one of the scenarios is presented in the executive summary.

These are the results for cumulative carbon impacts presented in the executive summary. Negative numbers indicate that in the year specified, net emissions from biomass still exceed those from fossil fuels:

Figure 4: Cumulative Carbon Dividends from Biomass Replacement of Fossil Fuel

Biomass Cumulative % Reduction in Carbon Emissions (Net of Forest Carbon Sequestration)				
Year	Oil (#6) Thermal/ CHP	Coal, Electric	Gas, Thermal	Gas, Electric
2050	25%	-3%	-13%	-110%
2100	42%	19%	12%	-63%

Below is the full table from Chapter 6, from which the results presented in the executive summary are drawn. The table presented in the executive summary repeats the results from Harvest Scenario 1. The assumptions behind these results are discussed in more detail below, but critical to placing these results in context is understanding that all harvest scenarios assume that biomass harvesting occurs only on land already harvested for timber at varying intensities, and that a large proportion of tops and limbs from commercial timber harvesting are available as "low-carbon" biomass fuel.

Exhibit 6-14: Cumulative Carbon Dividends: 2010 to 2050

Harvest Scenario	Fossil Fuel Technology			
	Oil (#6), Thermal	Coal, Electric	Gas, Thermal	Gas, Electric
1	22%	-3%	-13%	-110%
2	34%	11%	3%	-80%
3	8%	-22%	-34%	-148%
4	15%	-13%	-24%	-129%
5	16%	-11%	-22%	-126%
6	7%	-25%	-36%	-153%

Exhibit 6-15: Cumulative Carbon Dividends: 2010 to 2100

Harvest Scenario	Fossil Fuel Technology			
	Oil (#6), Thermal	Coal, Electric	Gas, Thermal	Gas, Electric
1	40%	19%	12%	-63%
2	56%	42%	36%	-18%
3	31%	8%	0%	-86%
4	43%	24%	17%	-54%
5	37%	16%	9%	-69%
6	31%	8%	-1%	-86%

HOW THE MANOMET MODEL WAS CONSTRUCTED

The Manomet model compares the emissions from biomass power for electricity only, thermal only, and combined heat and power plants against emissions from gas and coal in the case of electricity only plants, and gas and oil in the case of thermal and CHP plants. Lifecycle emissions consist of emissions at the stack from fuel combustion, as well as emissions associated with collection and transportation of the fuel.

Net carbon emissions are estimated as fuel lifecycle emissions minus forest carbon sequestration on a hypothetical acre which is cut for timber but not biomass (the fossil fuel/business-as-usual scenario), and one which is cut for both timber and biomass (the biomass scenario). Net carbon emissions from fossil fuels and biomass burning are compared by calculating the amount of lifecycle carbon emissions which are sequestered into new forest growth under the two scenarios. The model employs the Forest Vegetation Simulator, a model that uses Forest Service data on tree growth and forest composition, to estimate the recovery and regrowth of the forest following harvesting.

The report describes the approach:

> In general, the carbon accounting model should be premised on some knowledge of how lands will be managed in the future absent biomass harvests, and this becomes a critical reference point for analyzing whether burning biomass for energy results in increased or decreased cumulative GHG emissions over time. **(p. 99).**

> At the most general level, the carbon accounting framework we employ is constructed around **comparisons of fossil fuel scenarios with biomass scenarios producing equivalent amounts of energy**. The fossil fuel scenarios are based on lifecycle emissions of GHGs, using "CO2 equivalents" as the metric (CO2e). Total GHG emissions for the fossil scenarios include releases occurring in the production and transport of natural gas, coal or oil to the combustion facility as well as the direct stack emissions from burning these fuels for energy. Similarly, GHG emissions from biomass combustion include the stack emissions from the combustion facility and emissions from harvesting, processing and transporting the woody material to the facility. Most importantly, both the fossil fuel and biomass scenarios **also include analyses of changes in carbon storage in forests through a comparison of net carbon accumulation over time** on the harvested acres with the carbon storage results for an equivalent stand that has not been cut for biomass but that has been harvested for timber under a business-as-usual (BAU) scenario. Our approach includes the above- and below-ground live and dead carbon pools that researchers have identified as important contributors to forest stand carbon dynamics.

> The conceptual modeling framework for this study is intended to address the question of how atmospheric GHG levels will change if biomass **displaces an equivalent amount of fossil fuel generation** in our energy portfolio. With this objective, the modeling quantifies and compares the **cumulative net annual change in atmospheric CO2e for the fossil and biomass scenarios**, considering both energy generation emissions and forest carbon sequestration. In the fossil fuel scenarios, there is an initial CO2e emissions spike associated with energy generation—assumed here to be equivalent to the

energy that would be produced by the combustion of biomass harvested from one acre—**which is then followed by a drawing down over time (resequestration) of atmospheric CO2e by an acre of forest from which no biomass is removed for energy generation**. For the biomass scenario, there is a similar initial release of the carbon from burning wood **harvested from an identical acre of natural forest**, followed by continued future growth and sequestration of carbon in the harvested stand. **(p. 96)**

In the modeled acre cut for biomass, the forest is cut for timber at the same intensity as in the BAU scenario, but then more trees are removed to provide biomass fuel. Additionally, a portion (65%) of the branches and treetops from the trees cut for timber are removed as fuel, and the same amount of tops and branches from trees cut for biomass are removed as fuel, along with all trunk wood. The model thus assumes that 35% of all tops and branches are left onsite, and that this material rots and emits CO2 over time. The Manomet study examines six alternative harvesting scenarios at various intensities of removal.

The analysis that compares the carbon sequestered over time on a single forest acre under the BAU scenario, versus that on an acre cut for biomass, serves as the basic "building block" of an integrated analysis that considers the summed emissions over time, and the summed regrowth over time. This can best be explained by inserting the figures 6-2a and 6-2b from the Manomet study. The first graph shows the regrowth on an acre of forest harvested only for timber (BAU) and one harvested for timber, with additional trees cut to provide biomass fuel. Because the heavier removal on the acre cut for biomass actually increases the growth rate in the recovering forest, the two curves eventually converge:
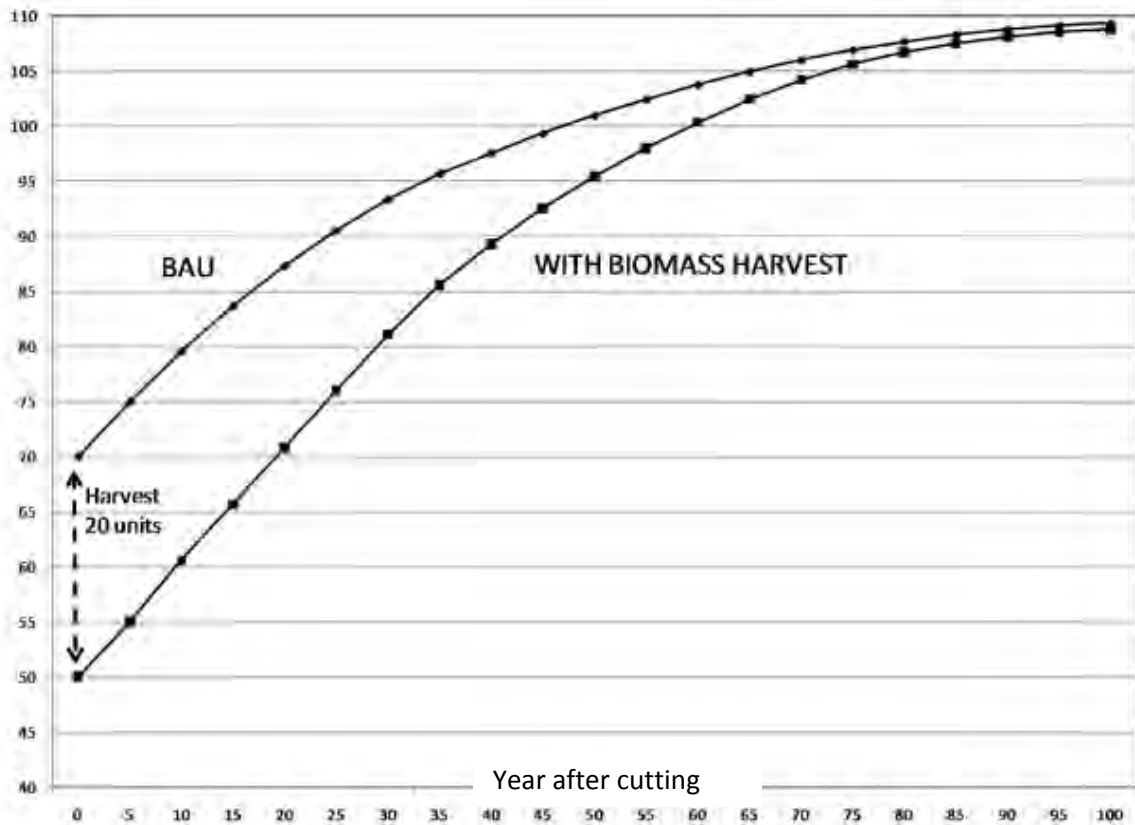

Figure 1. Forest growth following harvest in the "business-as-usual" timber harvesting scenario, and the scenario which harvests for both timber and biomass. This graph is labeled 6-2a in the Manomet report.

The next graph shows the cumulative net emissions from biomass and fossil fuel combustion, tracking the reduction in net emissions through time as the forest grows back. The single regrowth curve represents the subtraction of the BAU curve from the biomass curve in the graph above, essentially treating the carbon that would have been sequestered under the BAU scenario, which is now lost, as an "emission" that is associated with biomass harvesting. Immediately following harvest, the biomass scenario thus starts out with a "carbon debt" of an additional 9 tons of carbon that are harvested for biomass fuel after the initial 11 tons of carbon are removed for commercial timber. The point where the curve (which describes the net emissions from biomass burning) intersects the flat line (which describes the cumulative emissions from fossil fuel burning) is the point in time when the net emissions of the two scenarios are equivalent. This occurs at approximately Year 32 in this scenario.
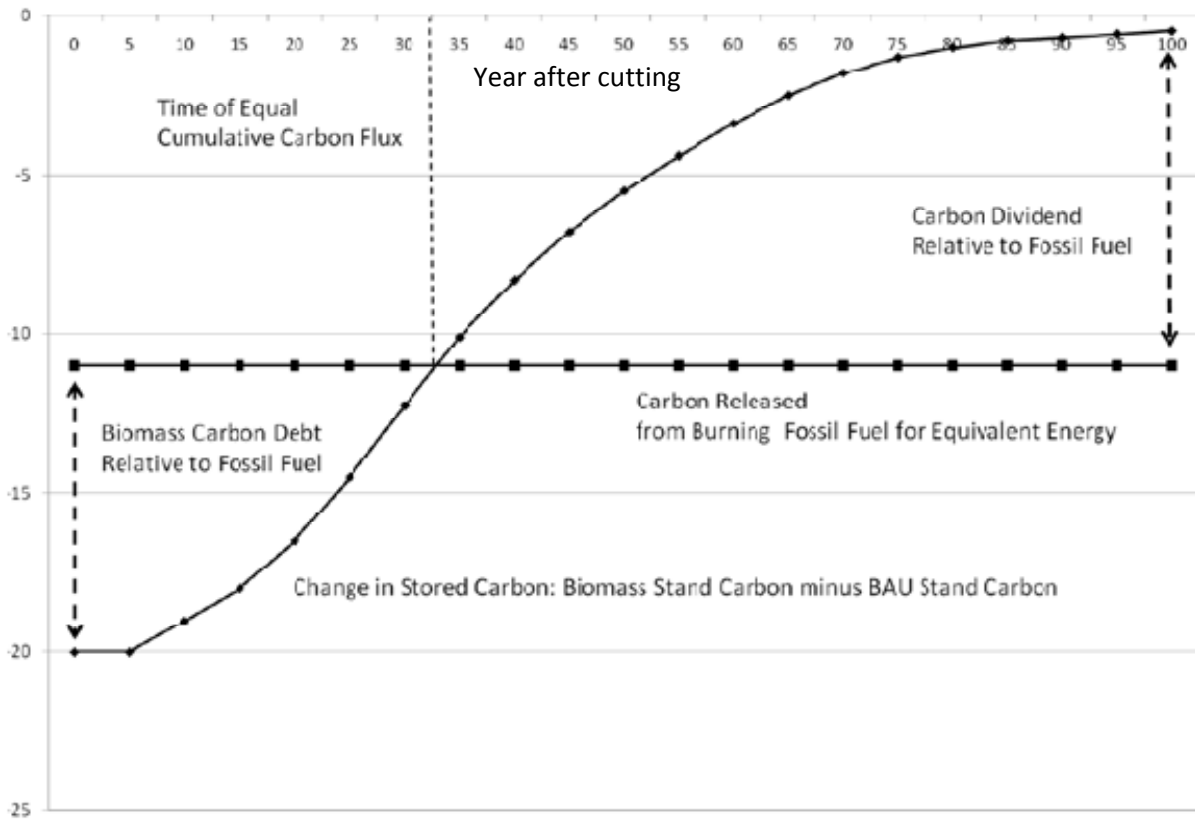

Figure 2. Biomass carbon debt under the biomass scenario, relative to carbon emissions from fossil fuel use, for forest cutting from a single year. The two lines cross at Year 32, a point where net emissions from biomass have achieved parity with net emissions from fossil fuels. Prior to this point, biomass power represents a carbon debt; after this point, it provides a carbon dividend, but only for a single year's worth of cutting on a particular harvested area. This graph is labeled 6-2b in the Manomet report.

It is important to understand that this curve only describes the recovery of carbon and the net carbon balance on a single acre of land harvested for biomass. The objective of the approach is to track regrowth following harvest through time, to determine the year in which the net carbon release from biomass is equivalent to the net release by an equivalent amount of energy produced by fossil fuels – the "time of equal cumulative carbon flux", which for this plot is approximately at Year 32. However, this does not describe the integrated picture of carbon emissions from a biomass facility, which operates continuously over many years and requires new forest to be cut every year. The integrated picture is more complex and consists of a series of curves, one for each year of cutting. Taking Manomet's graph of recovery on a single acre, above, and changing it to represent

several years of cutting produces the following graph, which for the sake of clarity and spacing treats the forest cutting episodes as if they happen every five years, instead of every year as they would in reality:
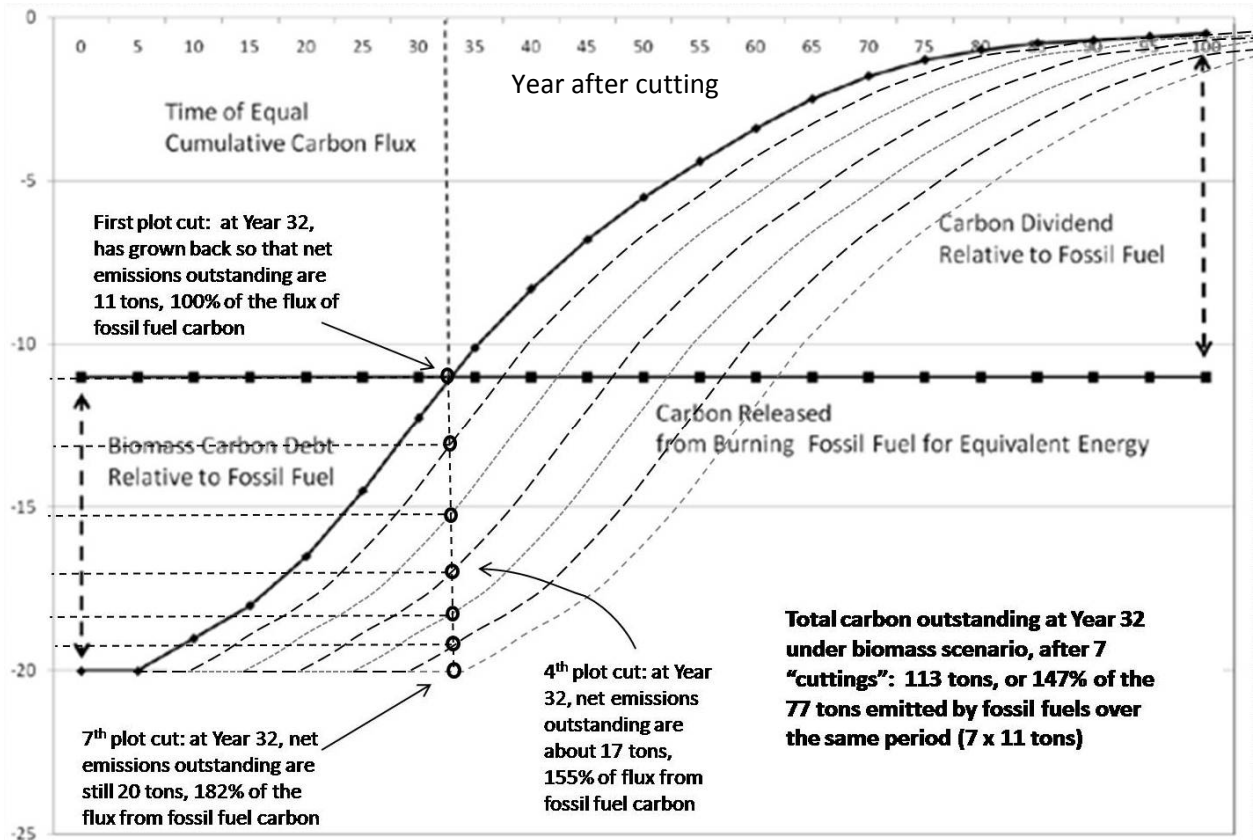

Figure 3. Integrated carbon emissions for a hypothetical biomass facility, assessed over a number of years. Whereas the first plot cut has regrown and achieved parity with fossil fuel emissions by Year 32, this is not the case for all subsequent plots cut, which still have carbon debts outstanding. Emissions from an actual facility, as assessed in this integrated picture, are considerably higher than for the single plot analysis presented above. At Year 32, net emissions from biomass in this hypothetical example are still 147% of those from fossil fuels.

A central problem with the Manomet study is the amount of space the report devotes to discussing the recovery of a single plot after cutting, and the relatively minimal of space used to discuss the integrated picture of total facility carbon emissions. This may be confusing to the typical reader of the report who is unlikely to have time to review it in detail. As can be seen from the analysis associated with the last graph, it takes much longer to achieve parity between biomass and fossil fuel carbon emissions when more than one year of cutting is considered. Whereas the single year analysis finds parity for biomass and fossil fuel emissions at Year 32, cumulative analysis of several years finds that biomass emissions are still 147% of fossil fuel emissions at Year 32.

The Manomet report does make this point. However, the analysis of cumulative effects, which is central to understanding the impact of biomass power, is only found at the very end of the report, where it is described as simply "another way" of looking at the data (this description also occurs in the executive summary, after the explanation of single-year results).

Another way of comparing the relative contributions of carbon debts and carbon dividends is to **estimate the difference in cumulative net atmospheric carbon emissions between using biomass and fossil fuel for energy at some future point in time**. Due to the importance of demonstrating progress in reducing greenhouse gas emissions by 2050 as part of the Massachusetts Global Warming Solutions Act, we have provided such a comparison for our six harvest scenarios in Exhibit 6-14. **(p. 111)**

Chapter 6, where the modeling results are described, devotes 15 pages to developing the results for the single-year analysis, even presenting charts such as exhibit 6-13 that create the impression that the time to parity under different forms of energy generation (thermal, CHP, and electric-only) is as low as 7 years when oil thermal heat is replaced, *a conclusion that would only be true if a biomass facility operated for a single year, then shut down*. A single page is devoted to discussion of the integrated multi-year analysis.

In the executive summary, these same time-to-parity results for the single-year analysis are presented *before* the discussion of cumulative effects, giving the impression that these are the more significant results. This is a major deficit in the report, particularly since the actual time-to-parity results for the cumulative effects analysis are never calculated.

Assumptions upon which the study conclusions depend

All of the assumptions listed below minimize the calculation of net carbon emissions from biomass. To the extent that these assumptions are not warranted, the Manomet study has underestimated the actual carbon emission impacts of biomass power, calling into question its conclusion that biomass will emit less net carbon over time than other forms of generation.

1. **Large trees are used for biomass fuel**. Because forest regrowth rates in the model are to a large extent a function of the intensity of harvest (with heavier harvests of larger, older trees opening up more space for regrowth to occur), the model achieves maximal regrowth and resequestration of carbon released by biomass burning by assuming that relatively large, old trees are logged for biomass.

2. **Harvested forest stands must not be recut pending carbon sequestration.** The model additionally requires that once a stand has been cut, it must not be re-cut until it has achieved a large proportion of the amount of standing carbon in an unmanaged stand.

3. **A high percentage of tops and limbs are used as fuel**. Because the tops and limbs of trees harvested for timber under the BAU scenario are assumed to stay in the forest and rot, producing carbon, the model assumes almost no carbon penalty for collecting this material and burning it. The model assumes that 65% of all tops and limbs generated on acres harvested for biomass can be removed from the forest for use as fuel, supplying a relatively large "low carbon" source of fuel in the model.

4. **Biomass harvesting only occurs on land that is already being harvested for timber**. The study takes as its BAU assumption that land is harvested for timber, and that all residues are left in the forest in this case, whereas a portion is collected for fuel in the

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biomass scenario. The study draws no conclusions concerning carbon dynamics and regrowth in forests cut solely for biomass.

5. **Soil carbon emissions are negligible**. The soil carbon pool is extremely large, and a significant fraction of it is easily decomposed and evolved as CO2 when soils are disturbed by logging. However, the Manomet model completely disregards this source of emissions that are associated with biomass harvesting.

6. **Firewood harvesting is not impacted**. Although indirect land use effects can be major sources of greenhouse gas emissions from biomass harvest, and although the RFP for the Manomet study requested that the study evaluate indirect land use effects, the study does not acknowledge that displacement of firewood harvest by biomass harvest could result in "leakage" of firewood harvesting and more forestland being cut for firewood.

7. **Wood pellet manufacture incurs no more carbon debt than green chips**. Although it is well-established that manufacture of wood pellets requires significant inputs of green wood in excess of the heating value actually embodied in the pellets produced, as well as significant fossil fuel expenditures, the study treats wood pellets as embodying the same amount of carbon and energy as green wood chips.

8. **Wood from land-clearing incurs little carbon debt.** The study concludes that woody biomass from non-forestry sources, such as from land-clearing, will not entail any greater greenhouse gas emissions than forestry wood. However, no modeling is conducted to substantiate this conclusion.

Review of assumptions

Large trees are used for biomass fuel

Because forest regrowth rates in the model are to a large extent a function of the intensity of harvest (with heavier harvests of larger, older trees opening up more space for regrowth to occur), the model achieves maximal regrowth and resequestration of carbon released by biomass burning by assuming that relatively large, old trees are logged for biomass.

> Alternatively, for some stands, and **especially for slow-growing older stands**, harvesting would be expected to **increase the carbon accumulation rate** (at least after the site recovers from the initial effects of the harvest) and lead to relatively more rapid increases in carbon dividends. Determining the time path for paying off the carbon debts and accumulating carbon dividends is a principle focus of our modeling approach. (**p. 99**)

Although biomass harvesting is often presented as a way of clearing out small trees in overgrown forests, the model does not treat the smallest trees as an available source of biomass fuel, instead setting a minimum diameter of 7 inches for trees to be cut:

> Approximately 65% of the standing trees on Massachusetts timberland are 1"–5" DBH; however, in spite of their large numbers, these sapling-size trees represent only 5% of the timber volume on a tonnage basis (FIA Statistics for 2008). **It would**

> **be cost prohibitive to harvest trees in this size class based on our analysis**. In order to be competitive in current markets, biomass producers would need to harvest trees with low stumpage value that are greater than 5" DBH. (**p. 42**)

> The model suggests that the **minimum size threshold for whole-tree harvesting in Massachusetts is in the range of 7.0–9.0 inches DBH** if the economic objective is to deliver chips to a bioenergy plant at a cost of about $30 (or less) per green ton.(**p. 41**)

> It seems that **pre-commercial thinnings and small trees should be excluded as part of the biomass resource in Massachusetts**—as one logger in Maine told us anecdotally, "the fastest way to go broke in the biomass business is to harvest 2-to-6 inch trees... These model results clearly demonstrate the critical importance of tree size and handling costs in the economics of whole-tree harvesting: **whole-tree harvesting appears to be cost prohibitive for sapling-size trees**. (**p. 41**)

The study concedes that some of the harvesting assumptions in the model could decrease the future economic value of the forest:

> However, new biomass markets may cause the harvest of trees that would **eventually develop into valuable crop trees if left to grow**. A straight, healthy 10" oak tree that would someday grow to be an 18" high-value veneer log might be removed too early in order to capture its much lower biomass value today. The misuse of low thinnings to remove biomass **could also remove the future sawtimber crop** as well as the forest structure referred to earlier.(**p. 73**)

Biomass harvesting is often portrayed as a way to create a market for small, low-value understory trees that are removed in thinning operations on commercial timber stands. However, removal of such trees does not cause the same growth and recovery in forest carbon as removing large trees does. Therefore, actual carbon recovery times are likely longer than represented by the harvesting scenarios that Manomet modeled, meaning that carbon debts persist longer.

Harvested forest stands must not be recut pending carbon sequestration.

The model additionally requires that once a stand has been cut, *it must not be re-cut* until it has achieved a large proportion of the amount of standing carbon in an unmanaged stand. However, the study itself acknowledges this assumption is likely unwarranted:

> The scenarios we defined as "biomass" harvests (Biomass 40%, Biomass BA40, Biomass BA60) maintain high growth rates for several decades. Because of this increased growth rate, even the heavier harvested stands can reach almost 90% of the volume that could have been achieved in an unmanaged scenario. So, over a long period of time, biomass harvests have an opportunity to recover a large portion of the carbon volume removed during the harvest. However, this **assumes no future harvests in the stand as well as an absence of any significant disturbance event. Both are unlikely**. (**p. 86**)

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Despite acknowledging that it is unlikely that having been cut for biomass, forests would be left uncut until a required level of carbon sequestration had been achieved, some of the central findings of the model depend on this assumption. For instance, the table of cumulative carbon dividends presented in exhibit 6-15, which describes the amount by which carbon sequestration under the biomass scenario would exceed that under the fossil fuel scenario for the 2010 to 2100 period, is based on the assumption that these acres would not be re-cut over this period. Even assuming that every one of the approximately 22,000 acres of private land cut for timber each year[1] were also available to provide biomass, the assumption that no acre could be recut over the 2010 to 2100 period would take a cumulative 1.98 million acres of forest out of production pending carbon resequestration. On a practical level, it seems unrealistic to assume that forests would be left uncut for even much shorter periods, if only because of the difficulties of enforcement. Presuming that biomass fuel would be licensed in some way by the state, the permissibility of any source would depend on future actions – i.e., the ongoing management of land into the future to ensure carbon sequestration – which seems much to ask for an already overburdened state government.

The study does attempt to grapple with the kinds of protections and enforcement that would be necessary to put in place at the state level, noting that many existing protections in forestry are voluntary and are probably not sufficient:

> Although in many cases BMPs are voluntary, water pollution control requirements are not, and therefore landowners are compelled by law to adopt water quality BMPs to avoid legal penalties. This may explain the relatively high rates reported for national compliance (86%) and in the Northeast (82%) (Edwards 2002). Biomass harvesting standards must address several management criteria such as protection and maintenance of forest structure for wildlife habitat, soil nutrient protection, and forest-stand productivity. **These criteria, unlike those for water quality, typically have no legal foundation to compel compliance**. (**p. 69**)

The study concedes that the harvest scenarios upon which their results depend are probably not realistic for other reasons, as well. For instance, the Forest Vegetation Simulator does not have the flexibility to simulate the kinds of harvests that are actually conducted by landowners:

> The impact of different silvicultural prescriptions has been more difficult to evaluate using the FVS model. The present set of scenarios uses a thin-from-above strategy linked to residual stand carbon targets for all harvests. **These types of harvests tend to open the canopy and promote more rapid regeneration and growth of residual trees**. While this silvicultural approach may provide a reasonable representation of how a landowner who harvests stands heavily in a BAU is likely to conduct a biomass harvest, **it is less likely that someone who cuts their land less heavily would continue to remove canopy trees for biomass** (unless they had an unusual number of canopy cull trees remaining after the timber quality trees are removed). More likely in this case is that the landowners would harvest the BAU timber trees and then selectively remove poor quality and suppressed trees across all diameter classes down to about 8 inches. We hypothesized that **this type of harvest would result in a slower recovery** compared to thinning from above. Unfortunately, the complexity of this type of harvest was difficult to mimic with FVS.

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[1] The study states on p. 31 that an average of 22,000 acres of private land are harvested each year.

Although project resources were not adequate to manually simulate this type of harvest for all FIA stands, we did conduct a sensitivity analysis for two stands with average volumes. For each of these stands we simulated a BAU harvest removing 20% of the stand carbon, followed by removal of residual trees across all diameter classes above 8 inches down to basal areas similar to the target in Scenario 4. For these two stands, the results, shown in Exhibit 6-11, **do indicate a slowing of carbon recovery profiles** relative to Scenario 4, although two stands are not enough to draw any conclusions about average impacts of this silvicultural prescription. **What can be said is that stands harvested in this manner will probably recover carbon more slowly** than would be suggested by Scenario 4; how much more slowly on average we did not determine; it is clear however that on a stand-by-stand basis the **magnitude of the slowdown can vary considerably**. (**p. 109**)

It is unfortunate that despite acknowledging a number of uncertainties in the text, the Manomet study still presents results for the time required for biomass scenarios to switch from incurring carbon debts to providing carbon dividends as if there is a high degree of confidence in the modeling.

A high percentage of tops and limbs are used as fuel

Because the tops and limbs of trees harvested for timber under the BAU scenario are assumed to stay in the forest and rot, producing carbon, the model assumes almost no carbon penalty for collecting this material and burning it. The model assumes that 65% of all tops and limbs generated from timber harvesting can be used for fuel, supplying a relatively large "low carbon" source of biomass in the model. The study states the rationale as follows:

> In order to project biomass supplies that can be used to meet potential demand from new bioenergy plants, we have assumed that 65% of the tops and limbs from harvested trees can be recovered on acres where silvicultural prescriptions include whole-tree biomass harvests. This percentage was selected for two reasons: 1) **it leaves behind more than enough material to conform to the ecological guidelines** that have been spelled out in Chapter 4; 2) it recognizes that a significant share of tops and limbs remain uneconomic due to timber breakage, small pieces, and small branches. (**p. 39**)

However, the ecological guidelines set out in Chapter 4 are quite general, an issue treated in more detail below, and the reader is left with little confidence that firm ecological guidelines have been set, much less conformed to. It seems likely that selection of 65% as an allowable level of harvest for tops and limbs, which are essentially treated as a low-carbon source of fuel by the model, is actually necessary to achieve the switch from biomass carbon debt to carbon dividend in a timely manner:

> The harvest and use of tops and limbs for biomass can have an **important influence on carbon recovery times and profiles**: tops and limbs decay quickly if left in the forest and so their use comes with little carbon "cost" which tends to shorten carbon recovery times. **Conversely, if tops and limbs from a biomass harvest of cull**

trees were left in the woods to decay, this "unharvested" carbon would delay recovery times, effectively penalizing wood biomass relative to fossil fuels.(**p. 109**)

When tops and limbs are left on-site, all three scenarios show net carbon losses between the initial period and the 10-year mark; in addition, **carbon losses in year 10 are substantial** relative to the recovery levels in the scenarios in which tops and limbs are taken and used for bioenergy. (**p. 110**)

In other words, it seems likely that the Manomet study would not have been able to portray biomass with even as favorable a carbon profile as it did, had a smaller percentage of tops and limbs been considered available as fuel. Given the several permutations on modeling described in the study, it is regrettable the study did not provide more detail about how leaving more tops and limbs in the forest would affect net carbon emissions.

Is it feasible to collect tops and limbs? The study in fact concludes that the practice is economical only in conjunction with whole-tree harvesting:

As discussed in the wood supply analysis in Chapter 3, the **harvest of tops and limbs would likely be economical only when harvested with whole-tree systems**. Biomass harvested in this manner can be used for any type of bioenergy technology. However, biomass can also be harvested with traditional methods or cut-to-length methods when these systems are preferred due to operating restrictions and/ or landowner preferences. These roundwood operations tend to be more costly, **but yield higher-quality bole chips that are preferred by thermal, CHP and pellet facilities**. Importantly, leaving tops and limbs behind as forest residues would **increase carbon recovery times** for bioenergy technologies that utilize the bole chips that are produced. (**p. 109**)

The distinction between facilities that use just chips from boles/ trunks and those that use whole-tree chips is an important one. Many small thermal biomass facilities depend on "higher quality" wood produced from boles, wood that is cleaner-burning and more consistent in quality. Pellet manufacture also preferentially uses bole wood. Questions of how preferentially harvesting for bole wood will affect the total amount of trees cut for clean chips and pellet feedstock are just starting to be explored, but given the Manomet study's endorsement of the thermal and CHP facilities that prefer these higher quality wood sources, it is unfortunate that the study does not explore these questions in more detail.

Soil nutrient implications of taking tops and limbs for fuel

The tops and limbs of a tree are the repository of a large share of its total nutrients, and this low-diameter material may actually represent a significant proportion of the biologically available pool of soil nutrients. How much such material should be left after logging not only to maintain these nutrient stocks, but also to protect soils against erosion and provide wildlife habitat, is the focus of many questions concerning the responsible use of woody biomass. Regarding the importance of leaving tops and limbs for forest ecological function, the Manomet study relies heavily on studies from the Forest Guild, specifically the Evans and Kelty "Ecology of Deadwood" report, which is included in an appendix to the study. The Manomet study repeats the conclusions of Forest Guild studies that there is little consensus regarding how much material should be left:

A review of scientific data suggests that when both sensitive sites (including low-nutrient) and clearcutting with whole-tree removal are avoided, then nutrient capital can be protected (see also Hacker 2005). **However, there is no scientific consensus on this point because of the range of treatments and experimental sites** (Grigal 2000). It is important to emphasize that the **impact on soil nutrients is site dependent**. Low-nutrient sites are **much more likely to be damaged** by intensive biomass removal than sites with great nutrient capital or more rapid nutrient inputs. A report on impacts of biomass harvesting from Massachusetts suggested that with partial removals (i.e., a combination of crown thinning and low thinning that removes all small trees for biomass and generates from **9 – 25 dry t/ac** or 20 – 56 Mg/ha) **stocks of Ca, the nutrient of greatest concern, could be replenished in 71 years** (Kelty et al. 2008). The Massachusetts study was based on previous research with similar results from Connecticut (Tritton et al. 1987, Hornbeck et al. 1990). Leaching, particularly of Ca due to acidic precipitation, can reduce the nutrients available to forests even without harvests (Pierce et al. 1993). However, the Ca-P mineral apatite may provide more sustainable supplies of Ca to forests growing in young soils formed in granitoid parent materials (Yanai et al. 2005). (**p. 141** of Manomet report).

The Kelty study cited in the report concluded that removal of just 9 – 25 dry tons of biomass per acre, an amount similar to that contemplated in the Manomet harvesting scenarios, could lead to soil nutrient depletion that lasted seven decades. The Manomet study downplays this finding, instead calling for more study of the issue and formulation of site-specific guidance for how much top and limb material can be removed:

> In Massachusetts it will be **important to identify the soils where there are concerns regarding current nutrient status** as well as those soils that could be degraded with repeated biomass harvests. (**p. 75**)

Despite acknowledging considerable uncertainty regarding the ecological sustainability of removing a large proportion of tops and limbs, the Manomet study does not present any substantive data or nutrient budgets to support the conclusion that 65% of tops and limbs can be removed at all sites. However, the carbon accounting component of the study relies on at least this much material being available, implicitly assuming that the maximum amount of tops and limbs can be removed in every case.

Although the study does call for the creation of guidelines on how much material should be retained in the forest, there is little discussion of how such guidelines could be practically implemented or the unusual amount of knowledge about a site's nutrient status and both <u>past and future</u> harvest plans that would be required of foresters when deciding how much material to leave:

> In areas that do not qualify as low-nutrient sites, where 1/3 of the basal area is being removed on a 15- to 20-year cutting cycle, it is our professional judgment that **retaining 1/4 to 1/3 of tops and limbs will limit the risk of nutrient depletion** and other negative impacts in most forest and soil types. Additional retention of tops and limbs may be necessary **when harvests remove more trees or harvests are more frequent**. Similarly where the **nutrient capital is deficient** or the **nutrient status is unknown**, increased retention of tops, branches, needles, and leaves is recommended. **Conversely, if harvests remove a lower percentage of**

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basal area, **entries are less frequent**, **or the site is nutrient-rich**, then fewer tops and limbs need to be retained on-site. (**p. 48**)

Implementing such protections and ensuring sufficient material is left onsite to maintain soil productivity would also involve foresters willingly forgoing a revenue stream from which they would otherwise profit.

With regard to use of tops and limbs from timber harvests as a "low carbon" biomass fuel source, the picture that emerges is that removal of at least 65% of this material is necessary for the Manomet model to reduce the apparent carbon emissions from biomass, since this material is assumed to decompose anyway and thus to represent a negligible addition of carbon if it is combusted. However, the study is not able to say with confidence or produce a body of evidence to demonstrate that removal of this amount of tops and limbs will not deplete soils or damage other forest functions, instead stating that much more detailed study is needed. In sum, it appears that the goals of achieving low carbon dioxide emissions from biomass fuel and maintaining soil nutrient status may be incompatible in many cases.

Biomass harvesting only occurs on land already being harvested for timber

The study takes as its BAU assumption that land is harvested for timber, and that all residues are left in the forest in this case, whereas a portion is collected for fuel in the biomass scenario. The study does no modeling and draws no conclusions concerning carbon dynamics and regrowth in forests cut solely for biomass. Because the BAU scenario assumes that all sawlog residues are left in the forest, this generates a large amount of relatively "low carbon" material to be harvested as fuel under the biomass scenario, because the FVS model treats this material as if it decomposes relatively quickly. The fact that the study does not examine carbon dynamics in stands cut solely for biomass is a considerable omission from the model; in fact, under such scenarios, carbon debts would be considerably longer than the Manomet study concludes.

Soil carbon emissions are negligible

The soil carbon pool is extremely large, and a significant fraction of it is easily decomposed and evolved as CO2 when soils are disturbed by logging. However, the Manomet model completely disregards this source of emissions that are associated with biomass harvesting.

The study states

> Our FVS model simulations captured the carbon dynamics associated with the forest floor and belowground live and belowground dead root systems. **Mineral soils were not included in our analyses, but appear generally not to be a long-term issue**. A meta-analysis published in 2001 by Johnson and Curtis found that forest harvesting, on average, had little or no effect on soil carbon and nitrogen. However, a more recent review (**Nave et al., 2010**) found consistent losses of forest floor carbon in temperate forest, **but mineral soils showed no significant, overall change in carbon storage due to harvest, and variation among mineral soils was best explained by soil taxonomy**.(**p. 83**)

The preceding paragraph was sent to the lead author on the Nave study, to ask whether he agreed with this assessment of his paper's conclusions. From his answer, it seems that the significance of the Nave paper bypassed the Manomet team. Here is Lucas Nave's answer in its entirety, as he requested (emphases added):

> "Thanks for asking about the meta-analysis paper we had in Forest Ecology and Management. My coauthors and I went over every sentence of that manuscript to be sure that we had the whole thing right, and now you've provided a great example of what happens when one statement is considered without the context of the rest of the document.
>
> We did indeed use those exact words: 'variation among mineral soils was best explained by soil taxonomy.' However, we were not referring to the background level of variation in the amounts of carbon (C) stored in different forest soils, which is what is implied by the quote you sent (orig message below). **What we were referring to with that statement was that**, when you assess the degree to which forest mineral soils vary in their C storage responses to harvest, **meta-analysis of the entire database shows that the most important factor controlling that variation is soil type (or taxonomic order**). Hence, a more complete characterization of our study results would have included discussing the two soil taxonomic orders that consistently lost soil C after forest harvesting, and the fact that following certain post-harvest management prescriptions can be used to prevent those losses. In a biome-level sense (ours included all temperate forests), it is true to say that mineral soil C storage doesn't generally change following a forest harvest. But that ignores underlying complexity that matters when you're not just talking about general concepts, but rather a specific location with an actual biomass harvest/C accounting plan on the table. **If our study is used to suggest that it's not necessary to include the mineral soil (typically the largest temperate forest C pool) in a management plan that includes C accounting, then it is being misused**. **The authors of that section of the Manomet report would benefit from closely re-reading our entire paper, which has more detailed, relevant information concerning the effects of forest harvesting on mineral soil C storage.**"

It thus appears that omitting soil carbon losses from the Manomet model means that actual biomass carbon debts are probably larger than the Manomet model concludes, and that time to parity with fossil fuel emissions is longer.

Firewood harvesting is not impacted

Although indirect land use effects can be major sources of greenhouse gas emissions from biomass harvest, and although the RFP for the Manomet study requested that the study evaluate indirect land use effects,[2] the study does not acknowledge that displacement of firewood harvest by biomass harvest could result in "leakage" of firewood harvesting and more forestland being cut for firewood.

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[2] The RFP for the sustainability study published by the Department of Energy Resources states: "The analysis will consider the carbon stack emissions of combusting biomass, the carbon absorbed by the forest growth, and emissions associated with biomass harvesting, processing, handling, transportation, and address whether there are any indirect land use impacts and the appropriate account for the displaced carbon emissions from fossil fuel otherwise used for energy."

To the extent that tops and branches and other low-value wood cut during timber harvesting are currently being removed as firewood, taking this material for biomass fuel could displace this firewood harvesting and lead to an overall increase in forest cutting. The study also does not consider the potential effects that use of low-value wood for biomass fuel could have on firewood costs.

The study states that firewood harvesting is a significant proportion of the wood removed from Massachusetts forests. The sources of this wood, which include cull trees, dead trees, tops and stumps of growing stock trees, overlap with the types of wood that are harvested for biomass fuel.

> The Timber Product Output reports provide one estimate of fuelwood production in Massachusetts; however, these data are derived from U.S. Census data rather than collected directly from U.S. Forest Service surveys (the source of other TPO data). TPO data indicate that **fuelwood production in Massachusetts in 2006 was 41.3 million cubic feet** (517,000 cords or 1.3 million green tons), **which would suggest that it would have accounted for about 83% of the timber harvest in Massachusetts** (see Exhibit 3C-1.) **According to this report, virtually all of the fuelwood comes from non-growing stock sources, which includes cull trees** (rough and rotten), dead trees, tops and stumps of growing stock trees, and non-forestland sources of trees such as yard trees. (**p. 136**)

However, the study is mixed in its acknowledgement that biomass harvesting could displace firewood harvesting, stating in some places that there are no leakage effects of increased biomass harvesting:

> More importantly for our analyses however, Chapter 6 assumes that the increase harvest intensity for biomass energy wood **doesn't change the disposition of materials that would be harvested absent biomass extraction**. (**p. 82**)

Elsewhere, the study does seem to acknowledge that biomass harvesting could displace other uses of wood, if not firewood specifically, at least under a scenario where biomass is worth more:

> This outlook assumes that biomass stumpage prices rise to $20 per green ton as a result of higher demand from bioenergy plants. A substantial increase in landowner income brings more land into production. Forest biomass fuel becomes a primary timber product, much as pulpwood is today, **and we assume that bioenergy plants can outbid their competitors for pulpwood and low-grade sawlogs and that this material is harvested more intensively as well**. (**p. 49**)

The study seems to acknowledge the impacts this could have on firewood harvesting on public lands, but does not discuss this issue for private lands:

> The main vehicle for achieving the increased biomass production on public lands will be the **diversion of wood from other end uses**: at the projected price levels for biomass stumpage, **bioenergy plants will be able to outbid their competitors for low-grade sawtimber, pulpwood, and residential fuelwood**. (**p. 53**)

However, public land is treated in the report as only a minimal potential source of biomass. Nowhere does the study examine the question of whether increased use of low-value wood as

biomass fuel could increase firewood harvesting elsewhere, or whether there might be increases in price for the firewood resource upon which many households depend.

Wood pellet manufacture incurs no more carbon debt than green chips

Although it is well-established that manufacture of wood pellets requires significant inputs of green wood in excess of the heating value actually embodied in the pellets produced, as well as significant fossil fuel expenditures, the Manomet study treats wood pellets as embodying the same amount of carbon and energy as green wood chips.

> Our analyses also considered the carbon debt characteristics of wood pellet technology and CHP systems. **In general, we find that carbon debts associated with burning pellets in thermal applications do not differ significantly from debts resulting from use of green wood chips**. The differences relate primarily to location of GHG emissions associated with water evaporation from green wood rather than the overall magnitude of the lifecycle GHG emissions. (**p. 106**)

However, the conclusion that carbon debts will not differ between green chip- and pellet-fueled facilities will only be true if the two kinds of fuel require the same amount of tree harvesting, and the same amount of production inputs in terms of fossil fuel power, to produce the same amount of thermal energy. Without delving into the complexities of where the energy to drive off moisture is expended (at the pellet plant, where wood heat or fossil fuels are used to dry the pellet material; or in the case of green chips, in the actual combustion process), it is easy to see that this is not the case. The pellet industry prefers the use of bole or trunk wood for pellet production, and thus requires harvesting far more trees to acquire the same amount of wood than if whole tree chipping were used. Thus, even assuming that the only difference between green chips and pellets was the moisture difference in the product, the pellet industry would still require more trees to produce product.

The report cites a pellet industry-funded study[3] to support their conclusion that lifecycle emissions from pellets are approximately equivalent to those from green wood chips:

> Emissions for thermal pellet applications require the addition of emissions from plant operations and for transport and distribution of pellets from the plant to the final consumer. The limited analysis that we have seen for these operations (for example, Katers and Kaurich, 2006) suggest that the increased efficiencies in boiler combustion achieved with pellets approximately offsets most of the increased emissions from plant operations and additional transport of pellets from the plant to their final destination. (**p. 104**)

In fact, the energy and fossil fuels expended during pellet manufacture and drying do appear to be considerable; where fossil fuels are used for drying, the study cited by the Manomet report shows that drying and plant operation require about 13% of the energy inherent in the pellet product itself. To the extent that wood is used to provide process heat at pellet plants, this is an additional wood input in the pellet manufacturing process that has not been accounted for by the Manomet study.

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[3] Katers, J. and Kaurich, J. 2007. Heating fuel life-cycle assessment. Study prepared for the Pellet Fuels Institute, February, 2007. University of Wisconsin, Green Bay. 54 pp.

The Manomet study also underestimates the amount of trees cut for pellet production because it underestimates typical wood moisture content. Their estimate that 1.575 tons of green wood is required to produce one ton of pellets at 6% moisture (p. 28), depends in part on the assumption that the green wood chips used to make pellet fuels have a moisture content of 40%, an assumption that does not match the standard industry estimate of 45% moisture content for green chips. Even the Katers and Kaurich study cited by the Manomet study itself assumes that green wood has a moisture content considerably higher than 40%:

> Dry wood feedstock can generally be obtained from saw mill waste or other similar industries that utilize kiln dried wood. This study assumed that a dry wood feedstock was available and drying the wood was not necessary, **which would not be the case for wood fuel pellets manufactured from green wood waste**. Green raw materials can often have **a moisture content in excess of 60%.** Moisture content will depend on time of harvest, relative humidity, as well as type of wood harvested. For this study it was assumed that the wood had a harvested moisture content of 55%. (**p. 8, Katers and Kaurich**).

The industry standard is that at least two tons of green wood are required to generate one ton of pellets, a calculation that is used in the commercially available wood products database from RISI, the global wood products information provider. The Manomet study appears to have significantly underestimated the actual amount of trees that would be required to provide pellet fuels.

Wood from land-clearing incurs little carbon debt

The Manomet study concludes that wood y biomass from non-forestry sources, such as from land-clearing, will not entail any greater greenhouse gas emissions than forestry wood. However, no modeling is conducted to substantiate this conclusion.

The report makes about 25 references to wood from land-clearing being a potential source of biomass fuel, but at no point are the carbon implications of this source of fuel critically examined. For instance, the study states

> Our carbon analysis considers only biomass from natural forests. Tree care and landscaping sources, biomass from land clearing, and C&D materials have very different GHG profiles. **Carbon from these sources may potentially enter the atmosphere more quickly and consequently carbon debts associated with burning these types of biomass could be paid off more rapidly, yielding more immediate dividends**. Our results for biomass from natural forests **likely understate the benefits of biomass energy development relative to facilities that would rely primarily on these other wood feedstocks. (p. 113**)

This conclusion, which is not substantiated with any analysis, appears to rest on the assumption that all wood from land-clearing must decompose very quickly, as is assumed for tops and limbs cut during BAU harvesting. This assumption is not warranted if the current fate of wood from land-clearing is not known; it is also not warranted if indirect land-use effects are not taken into account with regard to firewood harvesting. To the extent that wood from land-clearing is currently used for firewood, its use as biomass fuel could push timber harvest for firewood into new areas and result in an increase in forest cutting overall.

There is also no consideration of the impossibility for wood on permanently cleared land to regrow, which is the chief way that net emissions are considered to be reduced through time in the conventional biomass harvesting model. In Appendix 1-A, the study cites the Regional Greenhouse Gas Initiative (RGGI) Model Rule for the types of "eligible biomass", which, if used at a facility, generate emissions that can be deducted from the facility's total:

> *Eligible biomass includes sustainably harvested woody and herbaceous fuel sources that are available on a renewable or recurring basis (excluding old-growth timber), including dedicated energy crops and trees, agricultural food and feed crop residues, aquatic plants, unadulterated wood and wood residues, animal wastes, other clean organic wastes not mixed with other solid wastes, biogas, and other neat liquid biofuels derived from such fuel sources (quoted from the RGGI Model Rule, **p. 122** of Manomet report).*

There is no discussion within the Manomet report of how wood from permanent land-clearing can be considered "available on a renewable or recurring basis" as required under RGGI. Given that biomass facilities currently proposed in Massachusetts are claiming they will use wood from land-clearing as fuel, this is a serious omission in the report.

CONCLUSIONS

As disruptive as the results of the Manomet study could ultimately prove to the biomass industry, the study's conclusions actually likely significantly under-represent the actual carbon impacts of biomass energy. The conclusions that small-scale thermal and CHP biomass applications can repay carbon debts and yield carbon dividends relative to fossil fuels by 2050, and that net emissions from utility-scale biomass power exceed even those from coal after forty years of regrowth, rely on a number of assumptions that minimize the apparent emissions from biomass. These include assuming that large trees, rather than understory cull trees, are used as biomass fuel; that stands cut for biomass are not re-harvested until carbon resequestration has been achieved (a process that requires these stands be locked up from harvesting for decades); that only those lands already cut for timber are harvested for biomass; that a large proportion of "low-carbon" tops and limbs from timber harvesting are available for biomass fuel and that removal of this amount of material will not harm forest ecological function; that soil carbon emissions do not increase with harvesting; that indirect land use effects, particularly leakage of firewood harvesting, do not occur; and that pellet manufacturing does not incur a greater carbon debt than using green wood chips for fuel. In some cases, the report itself acknowledges that these assumptions are not likely justified; in other cases, the report is unfortunately silent on acknowledging the complexity of the carbon equation.

Even making these assumptions, the Manomet study concludes that net biomass emissions at utility-scale facilities still exceed those from coal after forty years, and are dramatically higher than emissions from natural gas. The lesson for New England, which generates much of its power from natural gas, is clear – relying on utility-scale biomass power to provide electricity to the grid causes a net increase in carbon emissions which undermines the emissions reductions goals of the Regional Greenhouse Gas Initiative. The best result that the Manomet model can produce for biomass performance relative to fossil fuels is that biomass carbon dividends in 2050 are on average 17% greater than from oil for small-scale thermal and CHP applications (averaging over the six modeled harvest scenarios) – a result that probably also underestimates actual greenhouse gas emissions from biomass power. In other words, this result depends on waiting 40 years to achieve a reduction in net greenhouse gas emissions that is at best is an extremely optimistic

scenario, and likely within the range of model error, given the many assumptions upon which the modeling relies. Over this 40 year period, much may happen to forests. Permanent forest loss due to development is continuing apace at about 5,000 acres per year in Massachusetts, and climate change, including potential effects of warming stress and invasive insects, may increasingly threaten forest carbon sequestration. The results in the Manomet study should thus be viewed by policy-makers as an extreme best-case scenario unlikely to be achievable in reality, and any policy designed to promote small-scale thermal and CHP biomass should be further evaluated with modeling that makes more critical and realistic assumptions. Further promotion of utility-scale biomass should be discontinued immediately as a threat to climate, and forests.

Carbon dynamics of Oregon and Northern California forests and potential land-based carbon storage

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Abstract. Net uptake of carbon from the atmosphere (net ecosystem production, NEP) is dependent on climate, disturbance history, management practices, forest age, and forest type. To improve understanding of the influence of these factors on forest carbon stocks and flux in the western United States, federal inventory data and supplemental field measurements at additional plots were used to estimate several important components of the carbon balance in forests in Oregon and Northern California during the 1990s. Species- and ecoregion-specific allometric equations were used to estimate live and dead biomass stores, net primary productivity (NPP), and mortality. In the semiarid East Cascades and mesic Coast Range, mean total biomass was 8 and 24 kg C/m^2, and mean NPP was 0.30 and 0.78 kg C·m^{-2}·yr^{-1}, respectively. Maximum NPP and dead biomass stores were most influenced by climate, whereas maximum live biomass stores and mortality were most influenced by forest type. Within ecoregions, mean live and dead biomass were usually higher on public lands, primarily because of the younger age class distribution on private lands. Decrease in NPP with age was not general across ecoregions, with no marked decline in old stands (>200 years old) in some ecoregions. In the absence of stand-replacing disturbance, total landscape carbon stocks could theoretically increase from 3.2 ± 0.34 Pg C to 5.9 ± 1.34 Pg C (a 46% increase) if forests were managed for maximum carbon storage. Although the theoretical limit is probably unattainable, given the timber-based economy and fire regimes in some ecoregions, there is still potential to significantly increase the land-based carbon storage by increasing rotation age and reducing harvest rates.

Key words: carbon cycle; climate regime; coarse woody debris, CWD; inventory data; management; mortality rates; net primary production; west-coast forests, USA.

Introduction

The amount of carbon sequestered by forest ecosystems plays an important role in regulating atmospheric levels of carbon dioxide (Canadell et al. 2007, Denman et al. 2007). Factors affecting the amount and rate at which forests sequester carbon include climate, disturbance, management, land use history, and species composition (Peet 1981, Harcombe et al. 1990, Law et al. 2004, Krankina et al. 2005, Gough et al. 2007). Pending and future forest management policies are attempting to offset anthropogenic carbon dioxide emissions by increasing and maintaining land-based sinks of carbon (IPCC 2007). Thus, it is important to quantify current and potential forest carbon pools and fluxes, and to understand factors that account for geographic variation.

Of particular interest is examination of age-specific trends. Commonly accepted patterns of live biomass accumulation and NPP in relation to stand age show a stabilization or decline as stands age (Bormann and Likens 1969, Odum 1969, Peet 1981). These patterns

have been identified using a variety of small-scale ecological studies that tend to select homogeneous stands (McCune and Menges 1986). However, because federal inventory plots are located in all types of stands (including transitional forests, uneven-aged stands, and areas that have experienced partial disturbances), the patterns of growth that emerge from inventory data may not follow idealized trends. The distinction between patterns elucidated from a sample of forest plots selected based on defined structural criteria, and a sample of stands selected regularly from the entire population is important when validating process models applied across landscapes (e.g., Jenkins et al. 2001). For this reason the patterns of carbon pools and fluxes over time, identified in inventory data, are uniquely powerful in that they can reveal collective trends (i.e., average values) in addition to idealized trends (i.e., upper bounds or maximum values).

In this study, data from supplemental field plots (i.e., foliage and fine-root metrics) and federal inventory data (FIA) were used to examine patterns of NPP, mortality, and live and dead carbon stores in different ecoregions of Oregon and Northern California, USA. The objectives were to: (1) identify age-related patterns of mean and maximum live biomass, dead biomass, NPP, and mortality; (2) determine the influence of forest type,
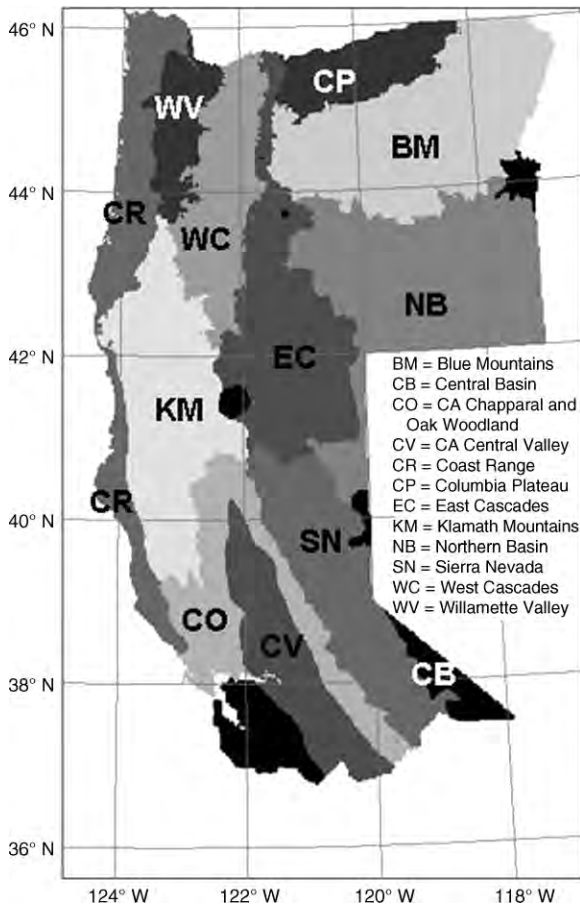
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FIG. 1. ORCA study region (Oregon and Northern California, USA) divided by Omernik Level III ecoregions (Omernik 1987); different shades of gray show the different ecoregions.

BM = Blue Mountains
CB = Central Basin
CO = CA Chapparal and Oak Woodland
CV = CA Central Valley
CR = Coast Range
CP = Columbia Plateau
EC = East Cascades
KM = Klamath Mountains
NB = Northern Basin
SN = Sierra Nevada
WC = West Cascades
WV = Willamette Valley

carbon stocks using supplemental data and species- and ecoregion-specific allometrics.

METHODS

Study area

The ORCA project is part of the North American Carbon Program (NACP), where the goal is to quantify and understand the carbon balance of North America. The study area is the entire state of Oregon and the northern half of California (Fig. 1). The disturbance history of the region includes frequent windthrow near the coast, relatively short harvest cycles, and centuries-long fire cycles west of the Cascade crest, moderate-length harvest cycles with more frequent natural fire cycles to the drier east, and livestock grazing in the Great Basin.

The area was divided into 12 ecoregions using the U.S. EPA Level III Omernik classification scheme (Omernik 1987). The ecoregions are classified according to similar biotic and abiotic characteristics, including dominant land cover type, climate, soils, and topography. They encompass several cover types such as chaparral, juniper woodlands, coastal Douglas-fir and hemlock, and true fir alpine forests. Age-related patterns of biomass accumulation, NPP, and mortality were only examined for the dominant six ecoregions (Table 1). Totals and means were calculated for all ecoregions (Appendix: Tables A1 and A2). Approximately 50% of this area is forested land, with 56% under public ownership and 44% under private ownership (Table 1). There is a steep west to east climatic gradient, with annual precipitation ranging from 2510 mm in the Coast Range to 120 mm in the Central Basin.

Data were used from several different inventories collected by federal and state agencies and our field crews. Within plots, allometric calculations were made for each individual tree, shrub, and woody detritus record and summations were made to obtain plot total live and dead biomass carbon estimates per unit ground area and an NPP and mortality estimate per unit ground area per year.

FIA database

The federal inventory program (FIA, Forest Inventory and Analysis) has undergone recent changes in sampling protocols starting in 2001. Historically, states were measured in subsections with a complete inventory of the states completed within 10–12 years (referred to as periodic inventories). The last complete inventory (1991–1999) in Oregon and California is summarized in the Integrated Database version 2.0 or IDB (Waddell and Hiserote 2005). Under a more recent protocol (annual inventory), portions of each subsection are completed each year, with a complete inventory expected by 2010. We chose to use the earlier periodic database because it is the most recent complete cycle and thus more representative of the study region. Our results thus approximate conditions in the mid-1990s.

ecoregion (climate), and ownership (management) on these patterns; (3) quantify total and theoretical forest carbon stocks and NPP over the study region; and (4) investigate the potential for increased land-based carbon sequestration in Oregon and Northern California forests. We examined differences in carbon pools and fluxes due to species composition, climate, and management indicated by forest type, ecoregion, and ownership, respectively. We used the patterns to determine the current land-based total stocks (excluding forest floor and soil carbon) and NPP, as well as the theoretical stocks and NPP in the absence of stand-replacing disturbance. The theoretical estimates are representative of the potential for carbon sequestration in this region through management practices. This study complements studies by Hicke et al. (2007) and Van Tuyl et al. (2005) by including additional carbon pools and fluxes (dead wood and shrub biomass, NPP, and mortality), examining possible causes of variation due to management and climate, and by further refining the estimates of

TABLE 1. Ecoregion mean annual precipitation (MAP), mean winter and summer temperature (MAT), total and forested area, and stand age.

Ecoregion†	No. plots	MAT (°C) (winter/summer)	MAP (mm)	Total area (ha)	Forested area (ha)	Mean stand age (yr)‡ All	Mean stand age (yr)‡ Private	Mean stand age (yr)‡ Public	All plots, and plots >200 yr old (%) Private	All plots, and plots >200 yr old (%) Public
BM	1266	−2/17	630	6 206 770	2 852 987	195 (3)	121 (9)	204 (3)	10, <1	90, 42
CR	737	7/18	2500	3 633 280	3 280 871	128 (4)	83 (3)	156 (6)	35, <1	65, 14
EC	1834	−3/17	500	4 795 480	2 971 042	185 (2)	133 (5)	190 (3)	8, 1	92, 36
KM	1526	3/21	1500	4 850 310	4 076 569	219 (4)	132 (6)	231 (4)	11, 1	89, 47
SN	1268	5/15	1500	3 931 700	3 059 246	196 (4)	146 (7)	201 (4)	9, 1	91, 38
WC	1896	4/16	2200	3 038 260	2 889 914	248 (4)	105 (36)	254 (4)	4, <1	96, 57
CB	26	4/11	150	834 689	165 932	210 (27)	84	219 (28)	4, 0	96, 50
CO	116	4/30	630	4 222 085	1 773 527	120 (6)	117 (7)	113 (14)	70, 7	30, 3
CP	4	0/21	250	1 756 090	40 046	145 (45)	145 (45)	...	100, 25	...
CV	0	10/17	450	2 656 190	49 832	...	...	...	...	...
NB	36	−1/18	250	6 556 370	174 125	150 (21)	184 (54)	177 (26)	17, 3	83, 22
WV	46	4/20	1270	1 373 040	504 923	109 (13)	86 (7)	162 (36)	70, 0	30, 9
Total	8755	2/18	1075	43 854 264	21 839 014	201 (2)	115 (2)	213 (1.5)	31, 1	41

Note: Ellipses in cells indicate that no data are available.

† BM, Blue Mountains; CR, Coast Range; EC, East Cascades; KM, Klamath Mountains; SN, Sierra Nevada; WC, West Cascades; CO, California Chaparral and Oak Woodlands; WV, Willamette Valley; NB, North Basin and Range; CB, Central Basin and Range; CP, Columbia Plateau; CV, Central California Valley.

‡ Standard errors in parentheses.

§ The first value is the percentage of total plots in private vs. public ownership. The second value after the comma is the percentage of total plots that are >200 years old in private vs. public ownership.

There were 14 188 plots with live tree data and 12 380 plots with woody detritus and understory data within the study area boundary (Fig. 1). The inventory design consists of 0.404-ha (1-acre) plots systematically placed across the landscape, encompassing a representative range of stand ages, disturbance histories, ownerships, and land cover types. The inventory data include tree diameter (dbh), actual height, wood increment, age, and species. Understory woody shrub data include percent cover, height, and species. Coarse woody debris and snag data include diameter, decay class, and species. We excluded plots that did not have enough increment data (1290 plots) to suitably calculate a stand age or a radial growth. One or more condition classes were assigned to plots that had more than one ownership, forest type, or disturbance history. Because we were interested in differences due to ownership and forest type, we chose to use plots with only one condition class. There were 4143 multi-condition plots distributed throughout the study area. A separate analysis of live biomass that included the multi-condition plots was done to determine if exclusion of these plots affected the overall results. The ecoregion means for total live biomass were slightly higher in the majority of areas, equivalent in some, and lower in the Coast Range by 0.90 kg C/m^2 (Appendix: Table A3). The regression of the 5-year age bin means for all plots vs. the corresponding single-condition bin means had values ranging from $r^2 = 0.75$ in the Blue Mountains to $r^2 = 0.91$ in the West Cascades.

After exclusions, 8755 plots remained with live tree and understory data, of which 8135 plots had measured woody detritus data. These plots were used to analyze NPP, mortality, and biomass for age-related trends as influenced by ecoregion, management, and forest type.

To evaluate and augment the federal inventory plots, we also used data from 170 supplemental field plots systematically dispersed among the ecoregions in the study area. Although these 1-ha plots cover a larger spatial area, the subplot and transect layouts, measurement protocols, and data collected met or exceeded the minimum standards of the federal inventory. Our plot locations were selected using a hierarchical random sampling design based on climate, forest type, and age (Law et al. 2006). The additional sampling was designed to allow a more comprehensive assessment of the carbon stocks and fluxes (Law et al. 2004, Sun et al. 2004, Van Tuyl et al. 2005, Law et al. 2006) by measuring foliage and soil carbon and nitrogen, leaf retention time, foliage and fine-root biomass and production (on selected plots), leaf area index, and litter stocks.

Stand age

Plot stand age was computed as the mean of the oldest 10% of trees (Spies and Franklin 1991, Van Tuyl et al. 2005). In cases where there were fewer than three trees in the oldest 10%, a mean of all aged trees on the plot was used. Stands older than 600 years were grouped into a single age class to account for an increasing trend in within plot variation. Although this method is the appropriate metric to best detect trends in growth and mortality, it is different than age based on time since disturbance and does not include effects of delayed establishment that vary widely (and probably with ecoregion, forest type, and ownership).

Ownership and forest types

Ownership was used as a surrogate for management practices because anthropogenic disturbances (i.e.,

clearcut harvest and thinning) have been more common on privately owned lands than on publicly owned lands in this region (Spies et al. 1994, Cohen et al. 2002). Public lands are defined as all non-private lands (federal, tribal, state, county, and so on). Private lands include small ownerships to large industrial properties.

Inventory plots were assigned a forest type code based on the dominant species on the plot. We grouped forest types into seven classes: (1) fir/Douglas-fir/hemlock, (2) larch and cedar/sequoia/redwood, (3) juniper, (4) spruce, (5) pine, (6) hardwoods, and (7) non-stocked. Non-stocked forest types are assigned to plots with a large percentage of ground area that is unsuitable for growth (i.e., rocky substrates).

Biomass

A database of volume and biomass allometric equations was compiled from prior studies (e.g., Means et al. 1994, Law et al. 2001, Van Tuyl et al. 2005) and new literature. We aimed to apply as many species-specific and ecoregion-specific equations as possible. Biomass estimates for trees included bole, bark, branch, foliage, and coarse roots (Appendix: Table A4). Species-specific wood densities (Maeglin and Wahlgren 1972, Forest Products Laboratory 1974) from wood cores obtained on our 170 supplemental plots were used to convert bole and coarse-root volume to biomass. For standing dead trees, wood densities were reduced according to decay class (Waddell 2002). Tree component biomass estimates were converted to kilograms per unit of ground area by multiplying by the tree-specific trees per hectare (TPH) and by 0.51 to obtain units of carbon (Law et al. 2001). TPH is a scaling factor for each tree record that is based on the sampled area and is supplied by the IDB specifically for generating plot-scale biomass estimates (Waddell and Hiserote 2005).

Fine-root biomass was estimated using an equation from earlier studies (Van Tuyl et al. 2005; Appendix: Table A4) relating leaf area index (LAI) and fine-root biomass. LAI is not measured on inventory plots, but was calculated by dividing foliage biomass (from the allometric equations) by the leaf mass per unit leaf area (LMA). LMA was obtained from a look-up table of species-specific values derived from measurements on the supplemental plots in each of the ecoregions. In some cases, a species-specific value was not available so a closely related (i.e., congeneric) species was used.

The conversion of shrub volume to biomass was from a database of allometric equations based on shrubs harvested at our supplemental plots. In total, 12 species were harvested, covering a wide range of morphology, leaf type, and leaf longevity, allowing for substitution of equations for all species where an equation could not be found. Shrub volume was calculated as the product of the recorded fraction of plot cover, plot area, and height. Coarse woody debris biomass was estimated following Waddell (2002). Biomass was calculated using

piece volume, species-specific wood density, and a decay-class density reduction factor.

NPP and mortality

To calculate NPP for a plot, a radial increment is necessary for every tree on the plot. Federal inventory includes stem increment cores for a subsample of the trees on each plot. For our estimates, trees on a plot were divided into dbh quartiles and the mean radial increment of cored trees in each quartile was assigned to all other trees in the same quartile (Sun et al. 2004, Van Tuyl et al. 2005).

Net primary production of all tree woody components was estimated as the difference in biomass at two points in time and was divided by the remeasurement interval (usually about 10 years). A previous dbh and height for each tree were necessary to calculate a previous biomass. Previous dbh was derived by back-calculation from current dbh and the radial increment, and previous height was recorded for remeasured trees or was modeled for unmeasured trees in the previous inventory using height–diameter regression equations from our supplemental plot data and BioPak (Means et al. 1994).

Woody shrub, foliage, and fine-root NPP were calculated using look-up tables constructed from supplemental plot data. Foliage NPP was calculated by dividing foliage biomass per tree by the average foliage retention time (average number of years of foliage that a stand carries). An ecoregion species-specific look-up table of foliage retention values was constructed from data gathered on the supplemental plots. Woody shrub NPP was calculated as a percentage increase in biomass per year. Increment disks from several shrub species were collected on the supplemental plots to produce a look-up table of average percentage increase in biomass for the species in each ecoregion. Fine-root NPP was calculated as the product of fine-root biomass and average fine-root turnover (1.2 $year^{-1}$) obtained from the literature and supplemental plot data (Keyes and Grier 1981, Campbell et al. 2004).

Mortality in kilograms of carbon per meter squared per year was only computed for trees. The IDB has assigned a mortality rate, the probability (0–1) that a given tree may die in one year due to natural causes, to each tree record. It is derived from a ratio of dead-to-live trees that were tallied on plots throughout the inventory area and developed for different groups by species and/or location (Waddell and Hiserote 2005). The amount of mortality expressed as the biomass loss per year can be estimated by multiplying the total live tree biomass by the location and/or species-specific mortality rate.

Statistical analysis

S-PLUS version 7.02 (S-PLUS 2005) was used for all statistical analysis. The questions of interest were addressed by comparing the coefficients of the response functions fit to the age-based distributions. Plots with

stand ages greater than 600 years were grouped into a single age group. Because less than 1% of plots on private land had age groups greater than 200 years, statistical analysis for comparisons between ownerships were restricted to plots aged 200 years or less for both public and private land.

Historically, private land ownership has tended to be located in lower elevation forested areas characterized by higher productivity. We confirmed this difference by comparing mean site index (a measure of site potential productivity) across ownerships for the stand area. We found a significantly higher mean site index value for private lands ($P < 0.01$, by permutation test). To isolate effects of differences in stand age distribution between ownerships from differences in site potential, the public land data set in each ecoregion was randomly subsampled using a constrained range and distribution of site indexes that was defined by private land distributions in the same ecoregion. This distribution was then used for the comparisons across ownerships.

To compare the coefficients of the fitted functions, the data for each ecoregion, ownership, or forest type were first binned into 25-year age groups and a stratified random sample of observations was chosen to ensure that the sample included data points from the entire age range. The appropriate functions were then fit to the sample data set. The sampling process and curve fitting were repeated to obtain 10 different estimates of each coefficient (used to produce a stand error) for each ecoregion and for ownership and forest type within ecoregion. A weighted one-way ANOVA using the coefficient standard errors as the weight was run for each coefficient to test for significant differences. To determine the relative influence of ecoregion, ownership, and forest type on biomass stores, NPP, and mortality across the entire study area and within ecoregion, the data were $\log_e$-transformed and linear regression models were compared using Akaike's information criterion (AIC). For the entire study area, model weights for age only, age + ownership, age + forest type, and age + ecoregion were calculated and ranked to determine the most influential explanatory variable (Burnham and Anderson 2002). Within ecoregion, model weights were calculated for age only, age + ownership and age + forest type.

Chapman-Richards functions (Pienaar and Turnbull 1973) were fit to live biomass and mortality data to compare the amount (mean and maximum) of carbon stored in biomass as a function of age:

$$\text{biomass} = a[1 - \exp(-b \times \text{stand age})]^c \qquad (1)$$

where parameter a is the asymptote, or the maximum amount of biomass carbon; parameter b determines the rate in years that it takes to reach the maximum amount; and c is a shaping parameter that gives a Chapman-Richards relationship the characteristic sigmoid shape. Because we were interested in the age at which NPP peaked and started to decline, a peak function (three-parameter, log-normal) was fit to the NPP data:

$$\text{NPP} = a \times \exp\left\{-0.5[\ln(\text{stand age}/c)/b]^2\right\} \qquad (2)$$

where parameter a is the asymptote or the maximum NPP; b is the rate to reach maximum; and c represents the age of initial NPP decline.

Woody detritus data can be fit with a standard decay function plus a Chapman-Richards function (Janisch and Harmon 2002). Stands typically start with large stores of legacy dead wood from prior stand development or downed wood caused by disturbance and then start to accumulate dead biomass as they age. The decay function quantifies the decay of the legacy dead wood in clearcut/burned and very young stands and the Chapman-Richards function quantifies the accumulation of dead wood as the stand ages:

$$\begin{aligned}\text{biomass} = &\, d \times \exp(-e \times \text{stand age}) \\ &+ a[1 - \exp(-b \times \text{stand age})]^c\end{aligned} \qquad (3)$$

where parameter d is the initial carbon stores and e is the decay rate in years. Although we were able to fit this function to all of the data in each ecoregion, we were unable to detect a U-shaped pattern with stand age in the smaller random samples used to compare the coefficients of the fitted function. Therefore, the data were divided into three age classes and a permutation test for a difference of means in each age class (by ecoregion or ownership) was used.

All of the above-mentioned curves were fit to both the mean values in each 5-year age bin and to the 99th percentile (hereafter "upper bound") of each age bin. Curves fit to the mean values represent the average realized trends of biomass, mortality, and NPP, whereas the curve fit to the upper bounds should represent the maximum values of stands in the ecoregion given minimal disturbance and ideal site and growing conditions. Curves fit to the upper bounds could also represent what many ecological field studies have documented and therefore what many modelers have used to parameterize and validate model results.

Regional-scale analyses

Total biomass, NPP, and mortality estimates for each ecoregion, state, and the total ORCA study area were obtained as the sum of the product of the relevant forested area using land cover data from Advanced Very High Resolution Radiometer (AVHRR) composite images recorded during the 1991 growing season (USDA Forest Service and U.S. Geological Survey 2002) and the corresponding mean values across all plots. The same procedure was used with GIS ownership coverages (from USGS National Land Cover Data 1992) to isolate patterns by public and private ownerships. Theoretical values for live biomass, dead biomass, and NPP were determined for each forest type × ecoregion combination using the parameter estimates for the maximum (parameter a) from the curve fit to the mean values of the data (note that this was from the

mean trend, not the upper bounds). Although these values should represent the maximum amounts, given a range of site productivities, and also should allow for partial disturbance, the amounts calculated assume the absence of stand-replacing disturbance through full harvest or catastrophic wildfire. Uncertainty estimates for regional stocks and NPP were calculated with a Monte Carlo procedure using the ecoregion mean and standard errors for the current values and the ecoregion parameter *a* estimates and standard errors for theoretical values. Simulations were run using a random normal distribution. We tested the independence assumption by using correlation coefficients for no correlation (0.0) and perfect correlation (1.0). Because the results did not vary significantly, uncertainty estimates for the simulations assuming no correlation are reported. Maps (ArcGIS version 9.1; ESRI 2005) of current mean and potential maximum carbon stocks and NPP were produced using the above results and a forest type land cover layer (USDA Forest Service and U.S. Geological Survey 2002).

RESULTS

Ecoregion patterns

There is strong evidence ($P < 0.001$) that maximum amounts of live biomass and rate of accumulation differ by ecoregion (Fig. 2). The Chapman-Richards function appeared to be a good general equation to describe biomass accumulation (Appendix: Table A5), especially for the upper bounds data in the West Cascades. When fit to the mean values, the maximum amount of live biomass (*a* in Eq. 1) is highest in the Coast Range and Klamath Mountains (33–44 kg C/m^2) and lowest in the East Cascades and Blue Mountains (7–10 kg C/m^2). The rate (*b* in Eq. 1) at which biomass reaches the maximum is lowest in the Klamath Mountains, with maximum stores still increasing at 600 years. Rates are higher in the other ecoregions, yet biomass is still increasing in stands over 300 years in the Coast Range, the Sierra Nevada, and the West Cascades.

We were unable to measure the inputs to or outputs from dead biomass, but we were able to compare the dynamic balance between these processes across ecoregions as measured by the standing mass of dead wood in three age groups (Table 2). There is strong evidence ($P < 0.001$, from a permutation test) that mean dead biomass differs among ecoregions for young, mature, and old stands. The Coast Range and West Cascades had the highest mass of dead wood in all age groups (ranging from 3.1 kg C/m^2 in the young to 4.7 kg C/m^2 in the old). The East Cascades had the lowest mass of dead wood in all age groups (ranging from 0.8 to 1.7 kg C/m^2) and the Klamath Mountains and Sierra Nevada had intermediate levels (ranging from 0.8 to 2.7 kg C/m^2).

The theoretical U-shaped pattern of dead biomass over time (high levels initially after disturbance, followed by low levels as this legacy wood decays, then

by high levels as new dead wood is recruited) was mostly apparent in the West Cascades when fit to the upper bounds (Appendix: Table A6), and slightly apparent in the East Cascades and Sierra Nevada (Fig. 3). The pattern was only slightly apparent in the West Cascades when fit to the mean values.

There is strong evidence ($P < 0.001$) that maximum NPP differs among ecoregions (Fig. 4). Maximum NPP peaks earliest in the Coast Range and West Cascades at about 80 years. The most obvious cases of late-successional decline in NPP for the upper-bound data were in the ecoregions with highest maximum NPP (Coast Range, West Cascades, and Klamath Mountains). Interestingly, the Coast Range was the only case of a conspicuous decline in mean NPP with age.

There was strong evidence ($P < 0.001$) that both maximum mortality and the rate at which it is reached differs among ecoregions (Appendix: Table A5). Maximum mortality was highest in the Sierra Nevada, followed by the West Cascades and Klamath Mountains for both the mean trend and upper bounds of the data (Fig. 5). Mortality appeared to increase with stand age and become less predictable in older stands. As a percentage of live biomass, mortality ranged from 0.50% in the Coast Range to 1.20% in the Sierra Nevada for stands younger than 80 years, from 0.35% in the Coast Range to 1.30% in the Sierra Nevada for mature (80–200 year old) stands, and from 0.35% in the Coast Range to 1.35% in the Blue Mountains for old stands (>200 years old).

Ownership patterns

Stand ages varied from 0 to over 1000 years, with a higher frequency of younger stands on private land than public land and more old stands on public lands (Fig. 6). Mean stand age for private ownership ranged from 83 years in the Coast Range to 146 years in the Sierra Nevada (Table 1). Public ownership mean ages ranged from 156 years in the Coast Range to 244 years in the West Cascades, where most of the land is public. There are very few stands older than 250 years on private land.

After accounting for site index, there is strong evidence that mean and maximum biomass differed between ownerships in all but the Blue Mountains and East Cascades ($P < 0.001$), with generally higher values on public lands (parameter *a* in Eq. 1, Appendix: Table A7). Dead biomass was not as consistently influenced by ownership as was live biomass (Table 2).

There is also strong evidence that mean NPP differed between ownerships ($P < 0.001$), being generally lower on public lands. Maximum NPP (parameter *a* in Eq. 2) was lower on public lands in the Coast Range, East Cascades, and the West Cascades and showed no difference in the Blue Mountains and Klamath Mountains (Appendix: Table A7). The number of years required to reach maximum rates of NPP was higher on public land ($P < 0.01$) in the Blue Mountains, East Cascades, and Sierra Nevada. Maximum mortality was
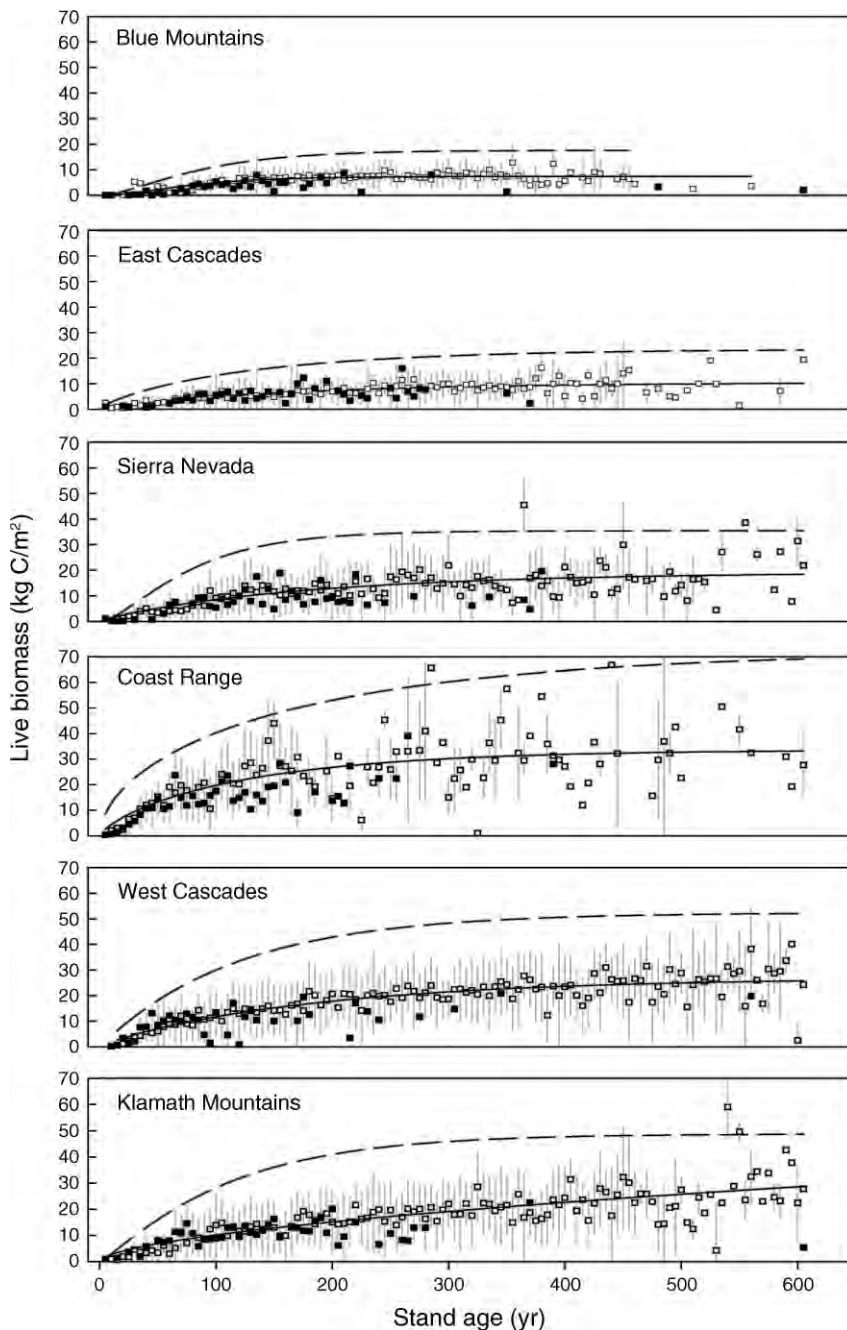
Fig. 2. Live biomass (trees and understory woody shrubs) vs. stand age. The dashed line (upper bounds) and solid line (mean trend) were fit using a Chapman-Richards function. Open (public) and solid (private) squares are the mean biomass for plots grouped into 5-year age bins. Gray vertical lines are the standard deviations in each bin. Note that, for clarity, plot mean values for the upper-bounds (99th-percentile) curve of the data are not shown on the figure.

higher on public land in the Blue Mountains and Sierra Nevada, but lower in the Coast Range.

Forest type patterns

Forest type differences were examined for live biomass and NPP. There was strong evidence ($P <$ 0.001) that forest types within an ecoregion differ in

maximum live biomass accumulation, maximum NPP, years required to reach maximum NPP, and age at initial decline of NPP (Appendix: Table A8). The overall variation in age-related patterns of biomass accumulation was reduced in some ecoregions when the Chapman-Richards functions were fit to the data separated by forest type. Douglas-fir forest types had the highest

TABLE 2. Estimates (mean with SD in parentheses) by age group, maximum of mean trend, and age at which maximum is reached for biomass, NPP, and mortality in each ecoregion and by ownership.

Ecoregion	Young	Mature	Old	Maximum	Age at max. (yr)	Young		Mature	
						Private	Public	Private	Public
Live biomass (kg C/m²)									
BM	1.9 (2.1)	5.9 (3.6)	7.4 (4.1)	7.1 (0.2)	180	3.0 (2.7)	3.1 (2.3)	4.8 (2.9)	6.5 (3.7)
CR	10.9 (6.9)	22.7 (12.4)	30.0 (14.5)	33.4 (3.0)	310	10.5 (7.9)	12.7 (6.6)	16.1 (7.8)	25.9 (12.9)
EC	3.0 (2.7)	6.2 (4.6)	8.7 (5.5)	10.1 (0.8)	310	3.1 (2.9)	3.2 (2.1)	6.4 (4.5)	5.6 (3.7)
KM	5.2 (5.0)	13.0 (8.9)	20.0 (11.9)	44.2 (33.0)	600+	8.3 (6.0)	7.3 (6.2)	12.4 (7.3)	15.0 (9.4)
SN	3.8 (3.5)	11.1 (7.3)	14.9 (9.6)	19.4 (0.7)	500	3.2 (4.3)	3.8 (3.4)	9.4 (5.0)	11.7 (7.3)
WC	6.2 (5.2)	14.6 (9.8)	22.1 (11.9)	26.9 (2.0)	430	7.9 (5.6)	7.0 (5.5)	11.3 (7.1)	16.4 (10.7)
Dead biomass (kg C/m²)									
BM	1.0 (0.9)	1.9 (1.8)	2.1 (1.9)	2.1 (1.9)	200+	0.6 (0.0)	1.0 (0.8)	1.6 (1.2)	1.8 (1.8)
CR	3.1 (2.7)	3.6 (2.9)	4.7 (4.8)	4.7 (4.8)	200+	3.2 (2.5)	3.2 (2.8)	2.9 (2.4)	3.8 (3.0)
EC	0.8 (0.8)	1.3 (1.2)	1.7 (1.6)	1.7 (1.6)	200+	1.9 (1.3)	0.7 (0.7)	1.3 (1.2)	1.3 (1.1)
KM	1.5 (1.4)	1.6 (1.5)	2.6 (2.3)	2.6 (2.3)	200+	2.0 (1.2)	1.9 (2.1)	1.9 (1.5)	1.6 (1.5)
SN	0.8 (0.8)	1.9 (1.7)	2.7 (2.2)	2.8 (0.2)	200+	2.4 (1.7)	0.8 (0.7)	1.9 (1.4)	1.7 (1.6)
WC	3.1 (3.1)	2.5 (2.2)	4.9 (4.1)	9.5 (5.2)	200+	4.0 (3.6)	3.2 (3.2)	3.2 (2.6)	2.7 (2.2)
NPP (kg C·m⁻²·yr⁻¹)									
BM	0.16 (0.12)	0.27 (0.10)	0.29 (0.09)	0.30 (0.01)	125	0.23 (0.15)	0.23 (0.12)	0.28 (0.11)	0.29 (0.07)
CR	0.75 (0.31)	0.80 (0.24)	0.77 (0.27)	0.82 (0.03)	60	0.75 (0.36)	0.79 (0.23)	0.91 (0.29)	0.78 (0.19)
EC	0.24 (0.15)	0.33 (0.19)	0.36 (0.15)	0.38 (0.01)	155	0.25 (0.18)	0.25 (0.10)	0.37 (0.18)	0.29 (0.12)
KM	0.45 (0.33)	0.63 (0.34)	0.62 (0.27)	0.65 (0.01)	110	0.66 (0.30)	0.65 (0.44)	0.70 (0.28)	0.73 (0.38)
SN	0.29 (0.19)	0.48 (0.22)	0.49 (0.20)	0.51 (0.01)	125	0.25 (0.23)	0.30 (0.19)	0.50 (0.19)	0.46 (0.21)
WC	0.46 (0.26)	0.52 (0.21)	0.49 (0.15)	0.54 (0.01)	65	0.59 (0.31)	0.50 (0.24)	0.59 (0.26)	0.54 (0.21)
Mortality (kg C·m⁻²·yr⁻¹)									
BM	0.02 (0.02)	0.07 (0.02)	0.10 (0.08)	0.10 (0.00)	140	0.03 (0.03)	0.03 (0.02)	0.05 (0.04)	0.08 (0.07)
CR	0.05 (0.05)	0.06 (0.04)	0.08 (0.04)	0.09 (0.01)	270	0.06 (0.06)	0.05 (0.03)	0.08 (0.05)	0.05 (0.02)
EC	0.03 (0.03)	0.07 (0.09)	0.10 (0.12)	0.11 (0.01)	270	0.02 (0.02)	0.03 (0.03)	0.06 (0.06)	0.05 (0.05)
KM	0.03 (0.04)	0.08 (0.09)	0.13 (0.11)	0.29 (0.27)	600+	0.06 (0.05)	0.03 (0.02)	0.07 (0.07)	0.06 (0.04)
SN	0.04 (0.02)	0.13 (0.12)	0.17 (0.14)	0.20 (0.02)	250	0.03 (0.03)	0.04 (0.05)	0.10 (0.08)	0.13 (0.13)
WC	0.04 (0.03)	0.10 (0.08)	0.14 (0.09)	0.17 (0.01)	310	0.02 (0.02)	0.04 (0.03)	0.05 (0.04)	0.10 (0.07)

Notes: Age groups are defined as young, <80 years old; mature, 80–200 years old; and old, >200 years old (information is not available for private land for the "old" group). Ecoregion codes are as in Table 1. Means for each ownership type (last four columns) were calculated with site-index-corrected data, resulting in higher means than for the entire data set (described in *Methods: Statistical analysis*).

maximum NPP, whereas pine forest types had the oldest age of initial decline of NPP.

Regional-scale analyses

Across the entire ORCA study region and after accounting for stand age, variation in live biomass (model weights 0.0–1.0) and mortality (model weights 0.12–0.88) was most explained by forest type, whereas variation in dead biomass (model weights 0.40–0.60) and NPP (model weights 0.02–0.98) was most explained by ecoregion (Appendix: Table A9). Within each ecoregion or climate zone, forest type was the most important explanatory variable in all cases except for mortality in the Coast Range, where ownership was most important (higher on private lands).

Total live biomass of forests in the ORCA study region (4.4×10^7 ha of forest land) is estimated at 2.71 ± 0.28 Pg C, mean ± SD (Appendix: Table A1; Figs. 7 and 8a). Understory biomass ranged from 1% to 5% of live biomass in most ecoregions and exclusion would not have resulted in a significant underestimation of biomass. Private land accounts for 35% of live biomass (and 44% of the forested area), with nearly one-third of the regional biomass in the predominantly privately

owned Coast Range. The total live biomass, assuming that all stands are at maximum carbon storage as determined by the mean trend of the data, would nearly double to 5.19 ± 1.34 Pg C (Fig. 8b). Although it would take hundreds of years to reach these theoretical levels in all ecoregions, land-based stocks could increase by 15% in just 50 years (i.e., if all stands increased in age by 50 years). Total dead biomass for the ORCA study regions is estimated at 0.51 ± 0.19 Pg C (Fig. 8c). Total dead biomass stores would increase by 36% to 0.80 ± 0.15 Pg C at maximum levels from the mean trend of the data (Fig. 8d). Total mortality of biomass for the ORCA study regions is estimated at 0.021 Pg C/yr. Total NPP of forest in the ORCA study region is estimated at 0.109 ± 0.001 Pg C/yr, mean ± SD (Fig. 9a) and total NPP would increase to 0.118 ± 0.002 Pg C/yr, assuming that all stands were at maximum NPP (Fig. 9b).

DISCUSSION

Trends with age

The expected age related ecological patterns (i.e., Chapman-Richards logistic growth for biomass accumulation, U-shaped pattern for dead biomass accumulation, and a marked decline in NPP with stand age)
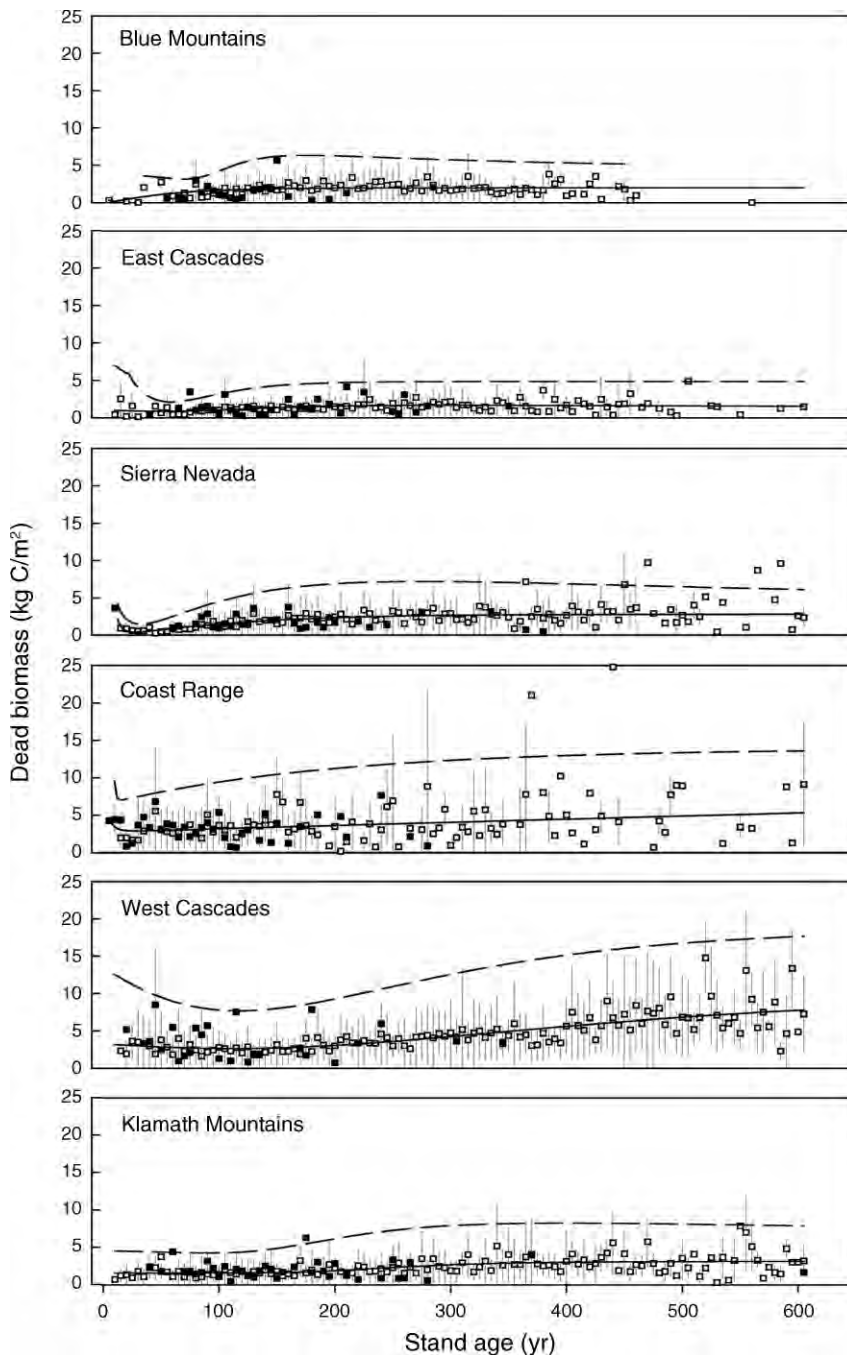
FIG. 3. Dead biomass (coarse woody debris [CWD] and standing dead trees) vs. stand age. The dashed line (upper bounds) and solid line (mean trend) were fit using a decay plus a Chapman-Richards function. Open (public) and solid (private) squares are the mean biomass for plots grouped into 5-year age bins. Gray vertical lines are the standard deviations in each bin. Note that, for clarity, plot mean values for the upper-bounds (99th-percentile) curve of the data are not shown on the figure.

were generally more distinguishable in the upper bounds of the data rather than in the age-specific mean values. In almost all cases, the West Cascades plot data were most suitably fit by these age related patterns. This is not surprising when considering that the ecological studies and data that were used to elucidate these patterns

deliberately and appropriately targeted productive, smaller scale, undisturbed, mature plots, especially in the West Cascades (Acker et al. 2002, Janisch and Harmon 2002). In contrast, federal inventory sampling, by design, includes the full suite of factors that cause a given forest to grow at the full range of rates.
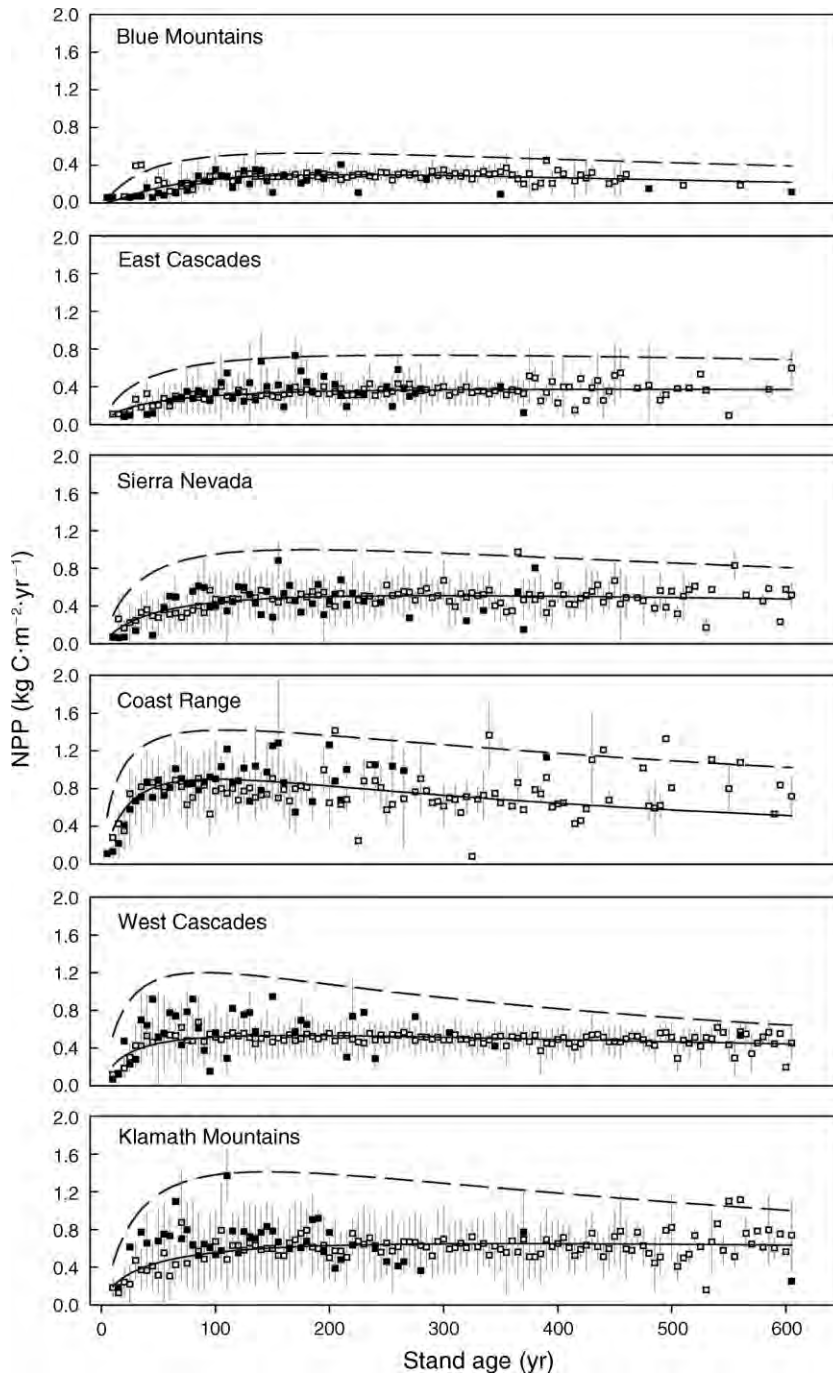
FIG. 4. Net primary productivity, NPP, of trees and understory woody shrubs vs. stand age. The dashed line (upper bounds) and solid line (mean trend) were fit using a Peak (three-parameter log-normal) function. Open (public) and solid (private) squares are the mean biomass for plots grouped into 5-year age bins. Gray vertical lines are the standard deviations in each bin. Note that, for clarity, plot mean values for the upper-bounds (99th-percentile) curve of the data are not shown on the figure.

With respect to coarse woody debris (CWD), the idealized U-shaped pattern arising from the combined and lagged effects of legacy wood decay and the recruitment of new dead wood (Harmon et al. 1986) was most apparent in the upper bounds. Coarse woody

debris biomass stores may be underestimated because the federal inventory CWD data required a minimum diameter measurement of 12.5 cm, rather than the 10 cm used by ecological studies or the 7.6 cm used by fire studies. Federal inventory data also do not include
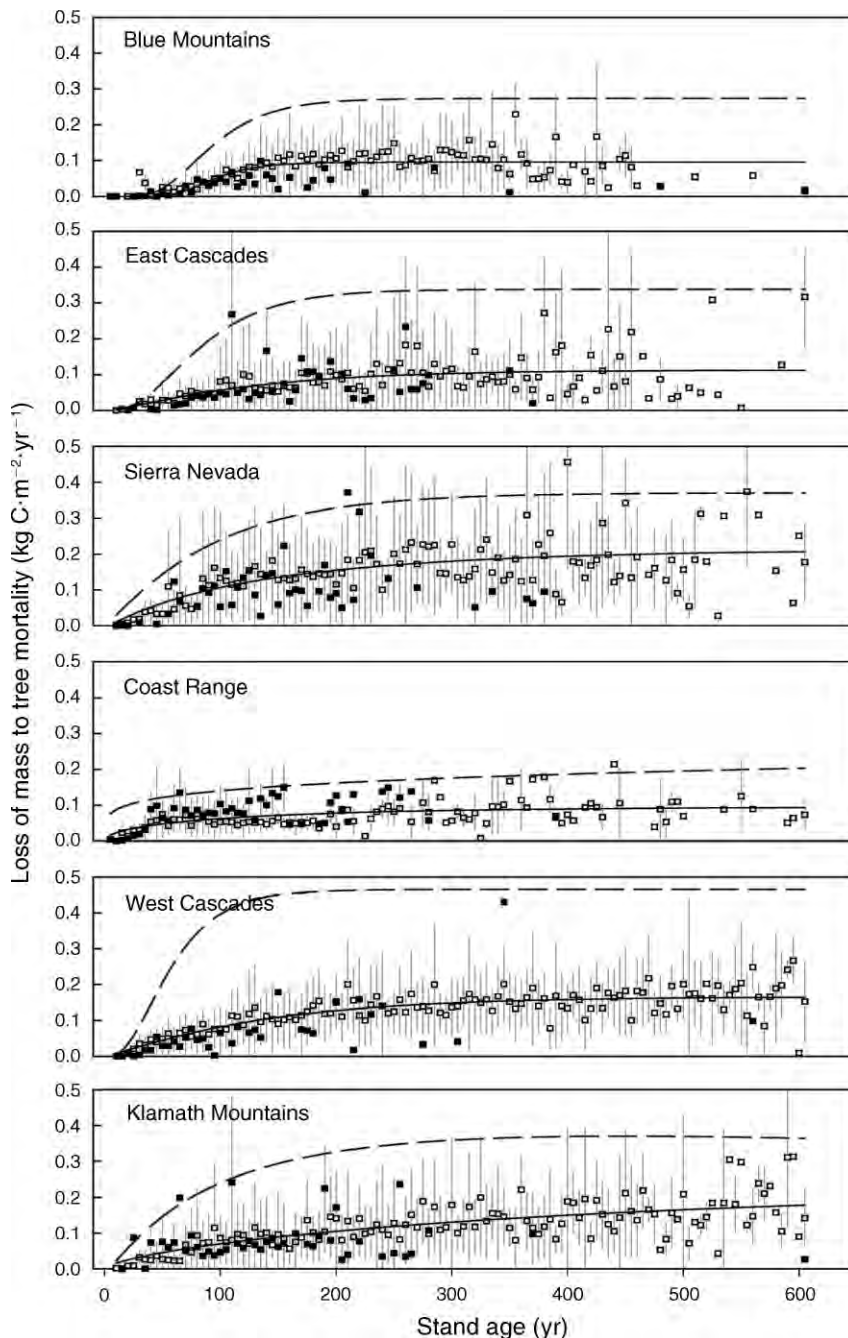
FIG. 5. Mortality of trees (loss of live carbon mass due to the death of trees) vs. stand age. The dashed line (upper bounds) and solid line (mean trend) were fit using a Chapman-Richards function. Open (public) and solid (private) squares are the mean biomass for plots grouped into 5-year age bins. Gray vertical lines are the standard deviations in each bin. Note that, for clarity, plot mean values for the upper-bounds (99th-percentile) curve of the data are not shown on the figure.

stumps, which would increase biomass in recently harvested (young) stands causing the curve to follow a more U-shaped pattern.

Because mortality affects both the inputs to dead biomass and the rate of loss from live biomass, it is important to characterize the factors controlling it. Our analysis shows mortality (expressed as an amount) increasing with stand age and stabilizing in late succession (Fig. 5). Expressed as a fraction of live biomass, we found that mortality tends not to be a constant. For example, mortality increases nonlinearly with stand age in most ecoregions and then reaches equilibrium while live biomass continues to increase. Carbon cycle models that have been applied in the
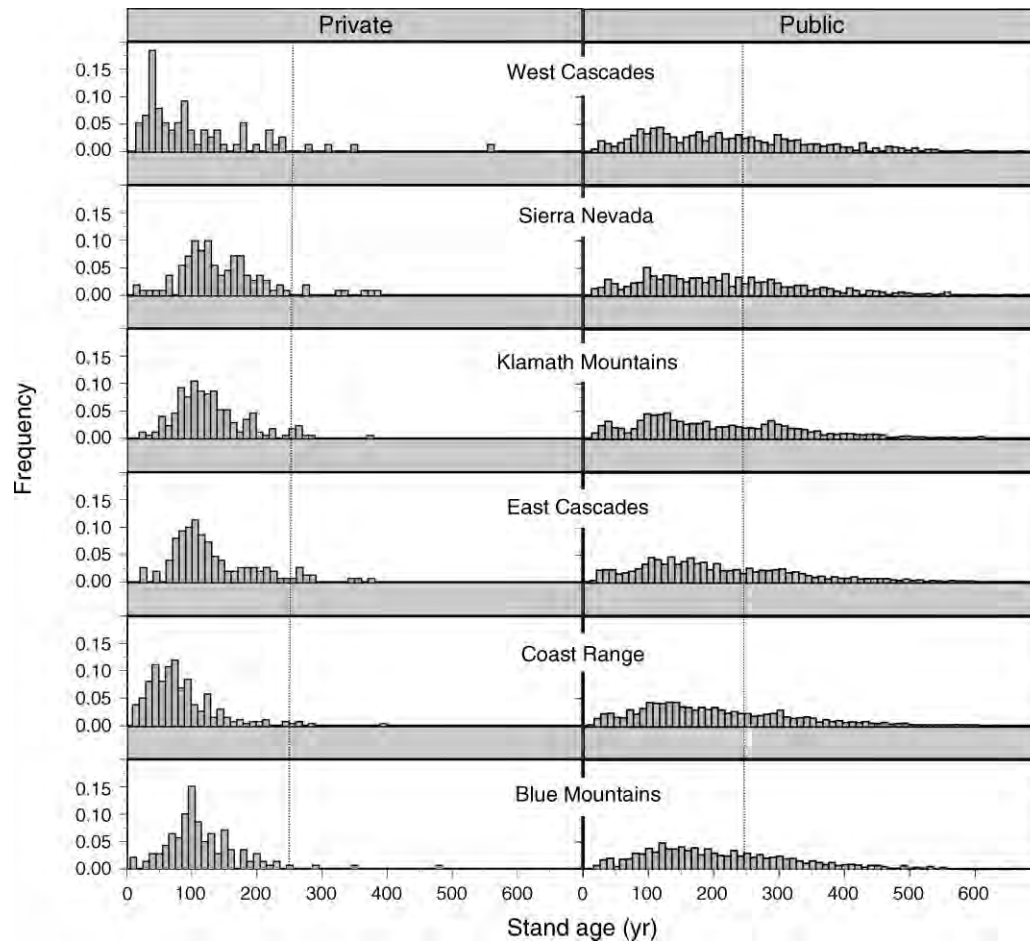
Fig. 6.   Frequency distributions of stand age by ecoregion and ownership class (private vs. public). Forests on private land tend to have more stands in lower age classes than stands on public lands. The light vertical lines delineate stands older vs. younger than 250 years.

Pacific Northwest (e.g., Turner et al. 2004) often represent mortality as a fixed percentage of live biomass and our results support the implementation of a dynamic mortality function in these models.

Increased mortality and decreased net primary production have been reported as equally responsible for late-successional stabilization of bole wood biomass in the West Cascades (Acker et al. 2002).The ecological studies upon which that conclusion is based are most relevant to our upper bound lines, and our results support these findings in some ecoregions (e.g., Coast Range, West Cascades, and Klamath Mountains). In those cases, upper bound NPP peaks at approximately 80 years and then declines (Fig. 4), and mortality (Fig. 5) increases with age to a stable rate that approximates bole-wood production. The FIA data suggest that in the most productive ecoregions, biomass continues to accumulate at low rates in very old stands.

In other ecoregions, there is less marked decline in NPP with age, or no apparent decline in NPP. For instance, pine forests in the Blue Mountains, East Cascades, and Klamath Mountains experienced NPP declines at significantly older ages than did other forest types in these ecoregions (Appendix: Table A5). The traditional explanation for NPP decline with age in forests, i.e., stable GPP and increasing autotrophic respiration, has largely been rejected in PNW forests (Ryan et al. 2004). The fact that decline is most apparent in the upper bound lines and least apparent for the relatively low productivity, more open-grown, pine forest type lends support to the hypothesis that competition-related changes in stand structure (Binkley et al. 2002) may be the critical driving factor.

Ecoregion patterns

In general, wetter ecoregions west of the Cascade Mountains crest (Klamath, Coast Range, and West Cascades) had much higher NPP and biomass stores at a given age than the drier ecoregions east of the crest (East Cascades and Blue Mountains). Despite the relatively high NPP and live biomass in the Klamath Mountains, dead biomass stores were 50–60% lower than in the

Coast Range and West Cascades. Although mortality rates are generally higher for both the Coast Range and West Cascades vs. the Klamath Mountains, the difference does not account for the large difference in dead biomass stores. Dead wood biomass is either being removed or consumed in the Klamath ecoregion at a much higher rate than in other ecoregions west of the Cascade crest. One explanation is differential decomposition rates. Decomposition is thought to be higher in the Klamath than in other west-side forests because of sufficient moisture and warm temperatures, but with fewer prolonged periods of moisture saturation, which can limit log decomposition (Harmon 1992). Others have proposed that historically frequent surface fires in the Klamath (every 5–75 years) consumed much of the coarse woody debris (Skinner 2002). It is most likely a combination of faster decomposition and shorter pre-suppression-era fire return intervals that underlie the lower amounts of dead biomass in the Klamath (Wright et al. 2002).

A consideration in interpreting biomass dynamics across ecoregions is the potential influence of 20th century fire exclusion, which has likely varied among regions. In dry forest types of some ecoregions (e.g., East Cascades, Sierra Nevada), fire suppression has resulted in long recent fire intervals relative to historical fire regimes that included frequent low-intensity surface fires (Agee 1993). These long intervals may allow greater live and dead biomass accumulations than under shorter fire intervals, as well as increases in stem densities that may affect patterns of stand productivity. By contrast, in wetter ecoregions (e.g., West Cascades, Coast Range), the fire suppression era has been brief relative to characteristic fire return intervals and probably has had little effect on biomass dynamics (Noss et al. 2006). Note that the chronosequence approach taken in our study is sensitive to the changes in disturbance regime over the last century. The effect of fire exclusion (and restoration) on carbon dynamics in different forest types remains an important direction for future research.

Ownership patterns and management implications

Forest management has always been somewhat different on private and public lands in the Coast Range, West Cascades, and Klamath Mountains (Spies et al. 1994), and reduced timber cut on federal lands in these ecoregions after implementation of the Northwest Forest Plan in 1990 has intensified the differences. Private lands in these ecoregions have, on average, less live biomass per unit area than do public lands because the typical harvest rotation (80 years) is much less than the age at which maximum biomass is reached (300 years). The frequency distribution of stand age could largely explain the differences in both live and dead biomass on public and private lands (Van Tuyl et al. 2005). Mean stand age of publicly owned forests is 50–150 years older than privately owned forests and mean carbon stores are 30–50% higher. Despite the fact that
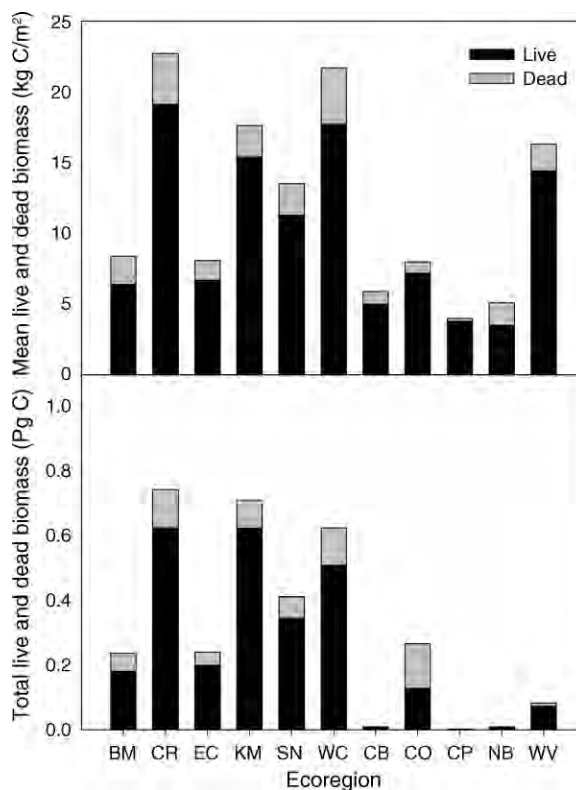

FIG. 7. Current mean and total carbon stocks in each ecoregion; mean and total live and dead biomass include trees, understory woody shrubs, and coarse woody debris.

the Coast Range has the highest percentage of private land (twice that of any other ecoregion), the lowest mean stand age, and the highest rate of removals by harvest (Law et al. 2004), it has the largest amount of biomass stored per unit area, presumably due to high NPP (climate), low natural mortality rates, and lack of recent major wildfires. Coast Range forests are among the most productive temperate forests in the world (Smithwick et al. 2002) and aboveground carbon stocks are comparable with tropical forest stocks (Houghton 2003, Sierra et al. 2007). Thus, there is high potential for increased land-based carbon storage with increased rotation age or reduction in harvest rates (Fig. 8b).

It is expected that management would affect NPP in younger stands because they are being managed for harvest and maximum wood volume production. After accounting for site index, our results supported this hypothesis, with maximum annual NPP of forests higher on private land than public land in all but the Blue Mountains and Klamath Mountains. An explanation for the lack of difference in these two ecoregions is difficult to ascertain because site index was accounted for in the analysis. There was also no difference in maximum live and dead biomass stores in the Blue Mountains. Maximum mortality is higher on public lands in the Blue Mountains, which could partially account for the decreased live biomass, but it may also
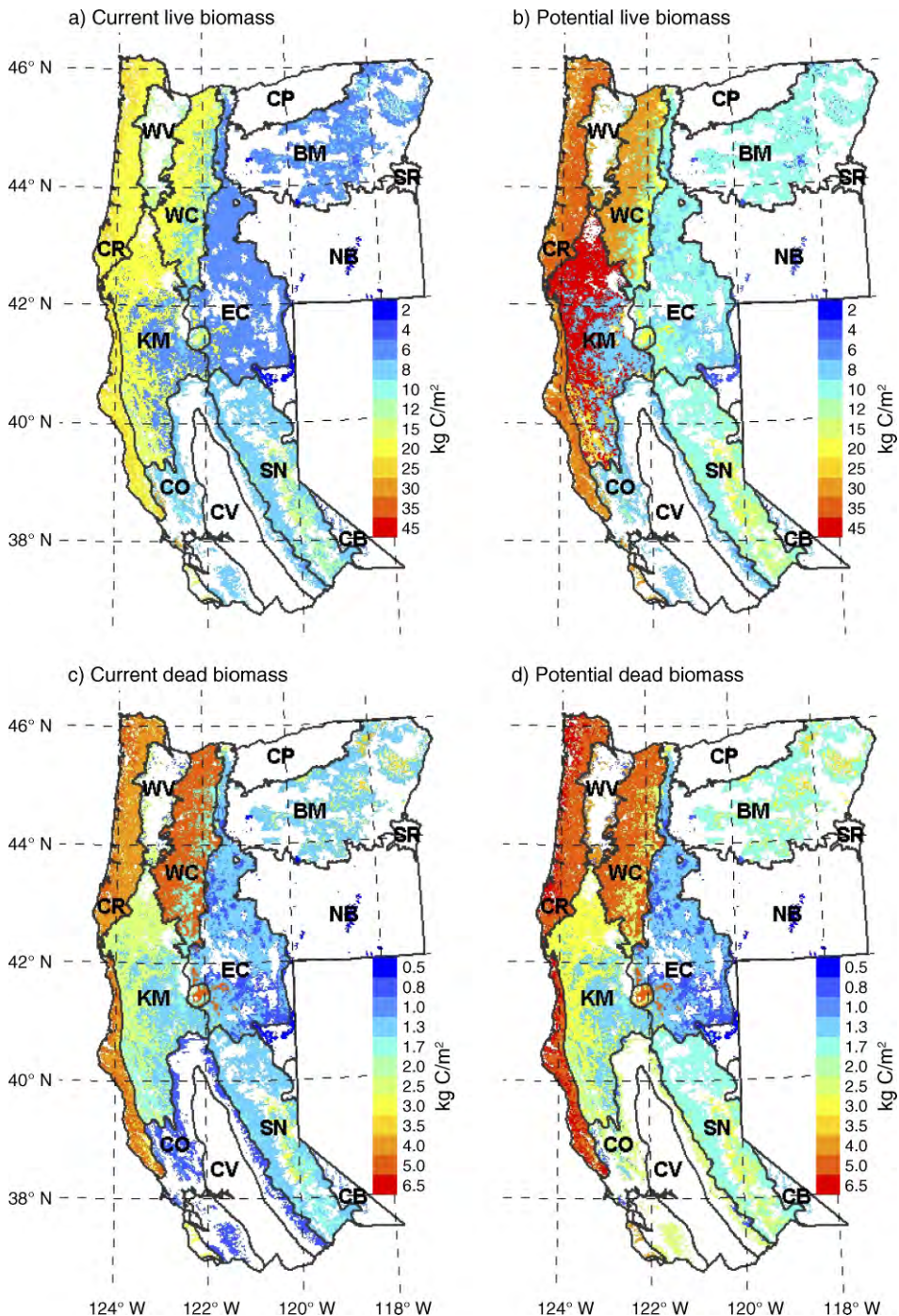
Fig. 8. Current and potential carbon stocks by forest type within ecoregion. Potential stocks were calculated using the mean trend maximums by forest type (Appendix: Table A6).

be that management practices are more similar on public and private land, as this ecoregion was also not affected by the Northwest Forest Plan.

Because ownership is associated with differences in mean biomass levels, a shift from current management on public land to a regime more like that on private land would gradually reduce mean carbon stocks on the land base. Decreasing rotation age to 50 years (i.e., mean age to 25 years) on all forested land in the Coast Range, Klamath Mountains, and West Cascades would reduce land-based live carbon stores by 53%. Some carbon can be sequestered in wood products derived from harvesting. However, due to manufacturing losses, only about 60% of the carbon harvested enters the products pool
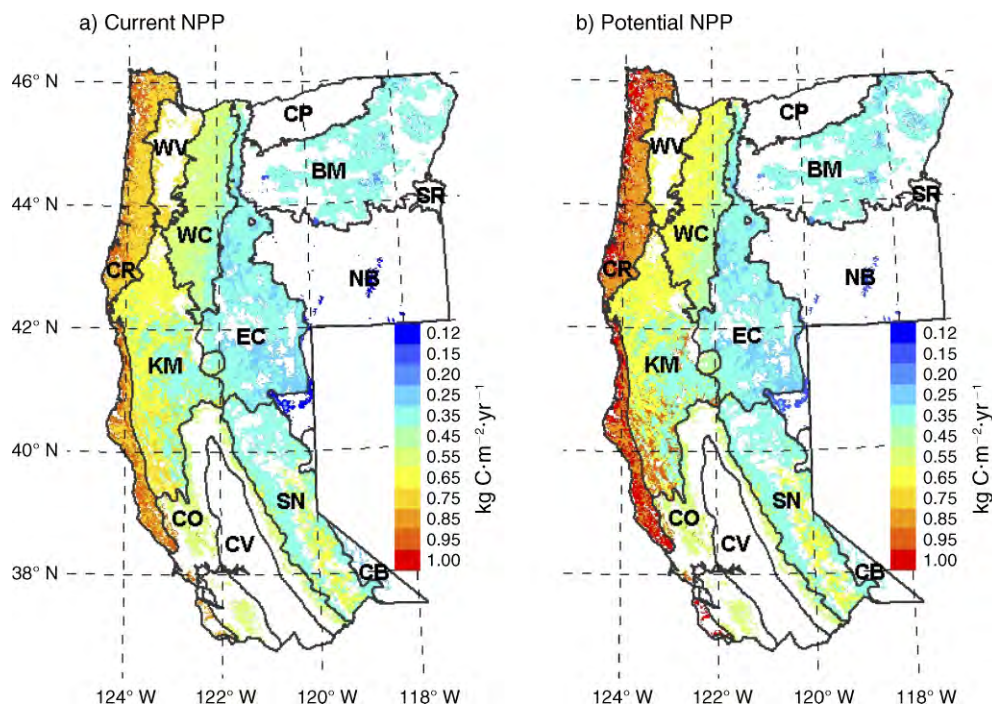
Fig. 9. Current and potential NPP by forest type within ecoregion. Potential NPP was calculated using the mean trend maximums of NPP by forest type (Appendix: Table A6).

(Harmon et al. 1996) and there are significant emissions from older products decomposing in landfills that tend to offset this carbon sink. Furthermore, with full carbon accounting there is a large carbon cost to the initial conversion of a landscape dominated by old forests to one dominated by young forests (Harmon et al. 1990).

Managing forests to maximize NPP may not have a positive effect on maximum regional carbon storage. Maximum NPP is reached much earlier in stand development than maximum biomass in all ecoregions (by hundreds of years) and limiting biomass accumulation to rotation ages based on maximum NPP would reduce land-based stocks. Examining current and potential productivity across the region is important when deciding regions where accumulation of biomass stores is most efficient. Highly productive ecoregions with infrequent fire such as the Coast Range are most likely to reach the potential stocks if managed for maximum biomass accumulation.

Modeling implications

As this study reaffirms, the processes driving forest carbon balance vary with stand age and forest type. Therefore estimates of regional carbon fluxes from modeling efforts depend in large part on our ability to accurately characterize stand age and forest type across the region (Turner et al. 2007). Including forest types as species groups in model parameterization could help further constrain model output for a specific area. Spatially explicit maps of age can be derived from

remote imagery (Cohen et al. 2002), but the accuracy of these maps is dependent on pronounced canopy structural development (Law et al. 2006). Until accurate age maps can be developed for the entire region, the distribution of forest age among inventory plots is uniquely valuable in developing probabilistic-based maps of age (e.g., Ohmann et al. 2007).

Ecological studies have identified patterns of response in biomass and productivity with age that are based on homogeneous forests growing under ideal conditions, and in this study, we found that those patterns are more evident in the upper bounds of biomass or productivity for a given age rather than mean response. The structure of many process models used for scaling NEP is often developed based on ecological trends elucidated from field studies on idealized study plots or the upper bounds. Depending on exactly how a process model is structured, it may be best to parameterize it with curves fit to the upper maximum of the inventory data because these trends reflect the unconstrained behavior of vegetation in a given ecoregion. This is especially true for trends such as age-related mortality, age-related allocation, and age-related declines in NPP that need to be explicitly enforced, because these high-order trends fail to otherwise emerge in standard simulations. Mean trends in the inventory data, on the other hand, have a different, but equally important value to modeling since after incorporating the constraints of disturbance and climate across a region, model output is best validated against the mean trends apparent in the inventory data.

TABLE 3. Forest carbon stocks in different regions.

Region	Stocks (kg/m^2)	Components	Source
Oregon and N. California	6.5–19†,‖	total tree biomass	this study
Oregon and N. California	17–70‡	total tree biomass	this study
West Coast USA	9.1–13.5†,‖	total tree biomass	Woodbury et al. (2007)
PNW, Coast Range	63‡	total tree biomass	Smithwick et al. (2002)
PNW, Cascades	58‡	total tree biomass	Smithwick et al. (2002)
PNW, Eastside	12‡	total tree biomass	Smithwick et al. (2002)
Northern Rockies	8†	total tree biomass	Hicke et al. (2007)
Colorado Rockies	<8†	total tree biomass	Hicke et al. (2007)
Mid-Atlantic USA	16–29‡	total tree biomass	Jenkins et al. (2001)
Mid-Atlantic USA	7–11†,‖	total tree biomass	Woodbury et al. (2007)
Eastern USA	7.5–25§	aboveground live tree biomass	Brown et al. (1999)
Russia	4.6–24§	total tree biomass	Krankina et al. (2005)
Canada	1–15†	total forest live biomass	Kurz and Apps (1999)
Latin America	13.5†	aboveground live tree biomass	Houghton (2005)
Tropical Asia	11.5†	aboveground live tree biomass	Houghton (2005)

Notes: Values reported as a range are the total estimates across different forest types or ecoregions. PNW is the Pacific Northwest.

† Includes foliage and coarse roots, but excludes fine roots.
‡ Reported as maximum levels for the study area.
§ Minimum to maximum levels across study area.
‖ Mean levels across ecoregions.
¶ From small-scale ecological studies (not inventory).

For this reason we advocate the separate characterization of mean and upper bound trends in federal inventory data.

Comparison with other studies

Our estimates of live and dead biomass, NPP, and mortality compare favorably with other regional studies (Table 3). Most recent federal inventory estimates of state total live and dead tree (excluding fine roots) carbon per unit area were 9–11 kg C/m^2 (Woodbury et al. 2007) in Oregon and California. Our results produce a mean of 12 kg C/m^2 (including fine roots) for both states, but range from 3 to 24 kg C/m^2 across the ecoregions. Hicke et al. (2007) reports a range of 4–20 kg C/m^2 for mean live tree biomass (excluding fine roots) at the county level in Oregon and California. When compared with the IDB estimates of ecoregion means, aboveground woody biomass varied by 5–10%.

Smithwick et al. (2002) reported that Oregon tree (including snags) and understory biomass in very old undisturbed stands averaged 63, 58, and 12 kg C/m^2 in the Coast Range, West Cascades, and East Cascades, respectively. Our estimates of the upper bounds of tree (not including snags) and understory biomass (Fig. 2) were 55, 50, and 20 kg C/m^2 for each of these ecoregions. Our East Cascade upper bounds estimate is much higher, but is based on a much larger number of plots over a more extensive area. Our aboveground wood biomass increments averaged 0.17 kg C·m^{-2}·yr^{-1} in the East Cascades to 0.50 kg C·m^{-2}·yr^{-1} in the Coast Range. In comparison with other regions, maximum live tree biomass averaged 10 kg C/m^2 in softwood forest types and wood biomass increment averaged 0.24 kg C·m^{-2}·yr^{-1} in the mid-Atlantic region (Jenkins et al. 2001). In tropical regions, total tree live-tree biomass averaged a 15 kg C/m^2 (Sierra et al. 2007) in Columbia

and ranged from 11 to 13 kg C/m^2 (aboveground biomass only) across all tropical forests (Houghton 2003).

Conclusions

The potential to store additional carbon in Pacific Northwest forests is among the highest in the world because much of the area has forests that are long-lived (e.g., Douglas-fir) and maintain relatively high productivity and biomass for decades to centuries. In Oregon and Northern California (4.4 × 10^7 ha), total live biomass of forests is estimated at 2.71 ± 0.28 Pg C, mean ± SD (mean of 12 kg C/ha) in the period 1991–1999. Total dead biomass (which does not include fine woody debris or litter stocks) of forests in the region was 0.51 ± 0.19 Pg C, and total NPP was 0.109 ± 0.001 Pg C/yr, mean ± SD. The majority of live and dead biomass (~65%) is on public lands (53% of forested land). Trends in NPP with age vary among ecoregions, which suggests caution in generalizing that NPP declines in late succession. Contrary to commonly accepted patterns of biomass stabilization or decline, biomass was still increasing in stands over 300 years old in the Coast Range, the Sierra Nevada and the West Cascades, and in stands over 600 years old in the Klamath Mountains. If forests were managed for maximum carbon sequestration total carbon stocks could theoretically double in the Coast Range, West Cascades, Sierra Nevada, and East Cascades and triple in the Klamath Mountains (Fig. 8). Our results indicate that Oregon and California forests are at 54% of theoretical maximum levels (3.2 ± 0.34 Pg C vs. 5.9 ± 1.34 Pg C) given the absence of stand-replacing disturbance. These theoretical levels are calculated using the mean trend of the data and account for variation in site quality, climate, and partial disturbance (i.e., thinning, insect outbreaks, non-cata-

strophic fire). An increase of 15% may be possible in just 50 years. However, these levels (if reached) may be unstable in high-frequency fire regions. A more realistic management approach to increase carbon storage on the landscape would be to increase rotation ages by 30–50 years or reduce the acreage that is harvested in areas more likely to reach the theoretical levels (Coast Range, West Cascades, Klamath Mountains). Initial conditions are important when considering management options. Maintaining mature and old forests that already store large amounts of carbon is a mitigation option, as suggested by the IPCC (2007).

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APPENDIX

Ecoregion, ownership, and forest type summary tables and statistics (*Ecological Archives* A019-008-A1).

Effects of Partial Harvest on the Carbon Stores in Douglas-fir/Western Hemlock Forests: A Simulation Study

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ABSTRACT

The STANDCARB 2.0 model was used to examine the effects of partial harvest of trees within stands on forest-related carbon (C) stores in a typical Pacific Northwest *Pseudotsuga/Tsuga* forest. For harvest rotation intervals of 20 to 250 years the effect of completely dispersed (that is, a checkerboard) versus completely aggregated cutting patterns (that is, single blocks) was compared. The simulations indicated that forests with frequent, but partial removal of live trees can store as much C as those with complete tree harvest on less frequent intervals. Stores in forest products generally declined as the fraction of live trees harvested declined and as the interval between harvests increased. Although the proportion of total system stores in forest products increased as the frequency of harvests and proportion of trees removed increased, this did not offset the reduction in forest C stores these treatments caused. Spatial arrangement of harvest influenced tree species composition profoundly; however, the effects of aggregated versus dispersed cutting patterns on C stores were relatively small compared to the other treatments. This study indicates that there are multiple methods to increase C stores in the forest sector including either increasing the time between harvests or reducing the fraction of trees harvested during each harvest.

Key words: carbon sequestration; carbon management and dynamics; disturbance; forest products; simulation modeling; Pacific Northwest.

INTRODUCTION

Forests are a critical part of the biological carbon (C) cycle and their management may contribute to stabilizing the concentration of the greenhouse gas C dioxide in the atmosphere (Pacala and Socolow

2004). Forests have great potential to store C (Post and others 1990; Dixon and others 1994), but the degree to which this potential is being met is uncertain. Clearing forests for agriculture and other land uses (Houghton and others 1983; Hall and Uhlig 1991), harvesting for commercial forest products (Cooper 1983; Houghton and others 1983; Harmon and others 1990), and the removal of non-commercial products such as fuel wood (Brown and others 1991; Houghton 1991) have decreased the amount of C stored in forests. Conversely, afforestation, fertilization, and protection from fire and insects have increased the amount of C stored

by forests (Tans and others 1990; Kauppi and others 1992; Cias and others 1995; Pacala and others 2001). Globally, forests are a net sink of C from the atmosphere (Gurney and others 2002), although very little of this current sink is due to actions directly related to deliberate C sequestration policies. If forests are to be deliberately managed to sequester additional C then the impact of various management approaches needs to be better understood.

Although timber harvest has generally reduced global terrestrial stores of C, the local effect of this activity is highly dependent on the initial conditions. Conversion of older forest to younger forests has generally been shown to release C to the atmosphere (Cooper 1983; Cropper and Ewel 1987; Harmon and others 1990; Dewar 1991; Schulze and others 2000; Harmon and Marks 2002). On the other hand, the creation of plantations on non-forest lands will generally increase C stores even when harvest occurs (Kauppi and others 1992; Kershaw and others 1993; Sedjo and Solomon 1991; Richter and others 1999; Johnsen and others 2001). Simply put, the net effect of harvest is dependent on whether the average C store in the initial condition is larger or smaller than the average C store in the harvested system (Harmon 2001).

The amount of C stored in a landscape is influenced by the interval between harvests and the fraction of C removed by disturbances. In general, the longer the interval between disturbance and less effective the disturbance is in removing C, the greater the average C store there is in a forested landscape (Smithwick and others 2007). A common result of most simulation studies is that as the interval between disturbances increases, the average C store in the system increases (Dewar 1991; Dewar and Cannell 1992; Harmon and Marks 2002). Thus, when timber harvest shortens the interval between disturbances a decline in average forest C stores can be expected. However, when a timber harvest system protects a forest from a more frequent disturbance (for example, fire) the average C store can increase with harvest (Kurz and others 1998; Seely and others 2002). Less clear is the effect of disturbances in removing C, in part, because during harvest, some of the removed C is stored offsite in forest products. In the case of site preparation practices, the more C removed in fires the lower the amount of C stored (Harmon and Marks 2002). Wildfires generally remove far less onsite C from forests than timber harvest (Tinker and Knight 2000), although as stated above some fraction of the harvest removal is stored offsite.

Typically 30–50% of the harvested C is lost in manufacturing and initial use, a loss that is larger than could be expected from even the most extreme forest fire (Harmon and others 1996; Tinker and Knight 2000; Campbell and others 2008).

Partial harvest of trees within a forest stand may also store more C than complete harvest (Dewar and Cannell 1992; Thornley and Cannell 2000), but there are uncertainties concerning the realism of these simulation studies. This is because most simulation models do not assess the impact of harvest pattern on foliage regrowth. Minimal impact of harvest pattern within a stand is likely in two cases: (1) complete removal of the trees, and (2) minor removals (for example, <2% per year). In the first case, there are no remnant trees to affect foliage development, whereas in the second case foliage losses can be replaced by minor amounts of growth. However, in the case of partial harvest the situation is potentially more complicated. Given that trees have a maximum crown width, there are levels of foliage removal that cannot be replaced by horizontal regrowth of existing trees. Replacing foliage in this situation depends on establishment and regrowth from below the canopy. This means that the remaining trees can influence the rate of foliage recovery by reducing light reaching the regenerating trees. Moreover, because the remaining trees affect light levels in the understory this can lead to changes in species composition, which can also influence C stores.

In this article, the STANDCARB model is used to examine the effects of partial harvest for forest stands on the dynamics of C in a hypothetical forest stand (Harmon and Marks 2002; Smithwick and others 2003). STANDCARB is a suitable choice to examine this aspect of forest management because it was designed to overcome common ecosystem model restrictions by incorporating the features of a gap simulation model (Urban and Shugart 1992) with an ecosystem process model (Harmon and others 1990). In STANDCARB, trees are simulated in a spatially explicit grid work and are assumed to have a maximum crown diameter with taller trees influencing the light environment of underlying and adjacent trees. Moreover, multiple tree species can exist in a stand with their abundance determined by their light environment. Given that tree species can also have different C-related parameters (for example, decomposition rates), it is therefore possible for changing species composition to change C dynamics. These features allow a more realistic examination of more complicated scenarios such as partial harvest within a stand. Our specific objective was to examine the effect of partial harvest of

live tree C on forest-related C stores. We examined a range of intervals between harvests and considered the C stores within the forest itself as well as in the forest products derived from timber harvest. We also examined the effect of cutting patterns by contrasting completely dispersed (that is, a checkerboard) versus completely aggregated patterns (that is, contiguous blocks).

MATERIALS AND METHODS

STANDCARB Model Overview

General aspects of STANDCARB are described by Harmon and Marks (2002) and Smithwick and others (2003). Detailed documentation of the model including example input and output files is available from http://andrewsforest.oregonstate.edu/lter/pubs/webdocs/models/standcarb2.htm. Below we review general aspects of the model and modifications of version 2.0.

STANDCARB simulates the accumulation of C over succession in mixed-species and mixed-aged forest stands. The model is parameterized for stands in the Pacific Northwest and as in many other C models it does not include the effects of nutrient cycling. This means that we assume that nutrient stores will not be influenced by the treatments enough to lead to major changes in site productivity. STANDCARB uses difference equations on an annual time step for all variables, except those

used to estimate the effects of climate on tree establishment, growth, and decomposition. These climate-related variables are calculated on a monthly time step. Spatially, STANDCARB is designed to simulate the dynamics of a number of cells within a stand. Each cell represents the area occupied by a single, mature tree (in these particular simulations this is an area of approximately 0.04 ha), although, depending on age, a cell can represent either a cohort of trees or a single tree. Within a cell spatial arrangement of trees is not considered; however, the relative height of trees in a cell determines the degree of interaction among cells.

STANDCARB uses a number of levels of organization to estimate changes in C stores within a stand (Figure 1). A stand is comprised of a number of *cells*, each of which contains up to four *layers* of vegetation, seven detritus pools, and three stable (that is, soil, wood, and foliage) pools. Four layers of vegetation can occur in each cell: (1) upper trees, (2) lower trees, (3) shrubs, and (4) herbs. There is only one species in the upper and lower trees in each cell; these can be the same or different species. Each of the layers can potentially have seven live *parts*: (1) foliage, (2) fine roots, (3) branches, (4) sapwood, (5) heartwood, (6) heart-rot, and (7) coarse roots. Each of the live parts of each layer contributes material to corresponding detritus or dead *pools* with the exception that heartwood and
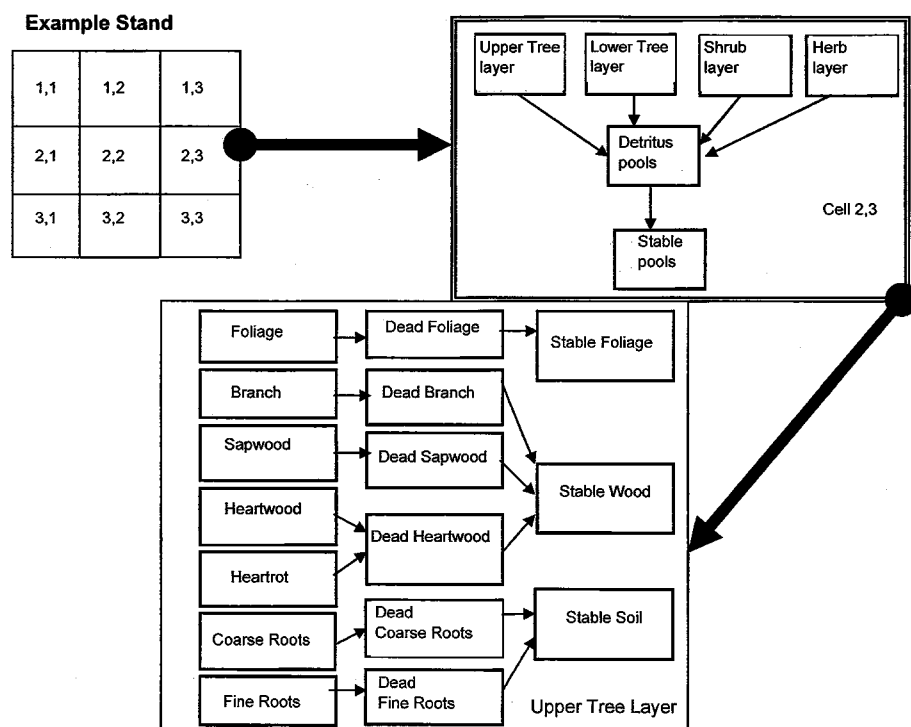

Figure 1. Conceptual structure of STANDCARB model showing the relationship among the stand, cells, plant layers, layer parts, detritus, and stable pools.

heart-rot contribute to the dead heartwood pool. Finally, all the detritus pools in a cell add material to either stable foliage, stable wood, or stable soil pool. Although stable pools lose C, their dynamics are much slower than that of the dead pools; stable pools thus represent highly decomposed materials.

Changes in STANDCARB 2.0

As with the original version, STANDCARB 2.0 contains 11 major modules that perform specific functions (Figure 2). The following section describes the modifications relative to version 1.0 of the model (Harmon and Marks 2002).

1. The time a cell takes to switch from a cohort to a single upper tree is now determined stochastically once the minimum age has been reached, whereas in version 1.0 the switch occurred once this minimum was reached.
2. The maximum tree height is now a function of site index.
3. The light coming into a cell is reduced by shading from surrounding cells, whereas in version 2.0 a small proportion of direct light is allowed to pass through neighboring cells to account for the sun flecks passing through minor openings.
4. The productivity of the trees in version 2.0 changes with age, so that as trees reach their maximum height their production declines by an amount set by the user. This accounts for the fact that production in older forests appears to be limited relative to younger forests (Acker and others 2000, 2002). Although there is some dispute about the exact cause of this phenomenon, there is little doubt it occurs in many types

of forests (Ryan and others 1997).

5. Heart-rot is now formed from heartwood after trees reach a minimum age, although the year heart-rot begins to form in a particular cell is stochastic.
6. Dead sapwood and dead heartwood are now separated into standing and downed material to account for the different microclimates of these two positions.
7. Dead pools are now tracked using a cohort structure for each year's input for a cell to account for the fact that a period of decomposition is required before stable materials are formed. Once a lag time, which is modified by climatic conditions, has been exceeded a dead pool cohort is stochastically transferred to the appropriate stable pool.
8. There are now three ''stable'' pools instead of one. Dead foliage is transferred to stable foliage (that is, the organic horizon), aboveground dead wood pools to stable wood, and dead fine and coarse roots to stable soil.

Model Calibration

Given that the purpose of our simulation experiments was to predict the relative effects of partial harvest on potential C stores, we calibrated STANDCARB to represent a common mixed-species forest in the Pacific Northwest. We therefore make no claims that the absolute levels of C stores are being predicted *a priori*. The simulated stands represent a mixture of two species, each with different potentials to store C (Table 1). The model determines how the mixture of species changes over succession, with Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) dominating early and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) dominating later in succession. The parameters of these species (Supplementary Table 1) are based on values estimated for Douglas-fir and western hemlock, with the latter storing approximately 6% less C on average (Mg ha^{-1}) than the former (Table 1). Therefore a species with higher potential is eventually replaced by one with lower C stores potential.
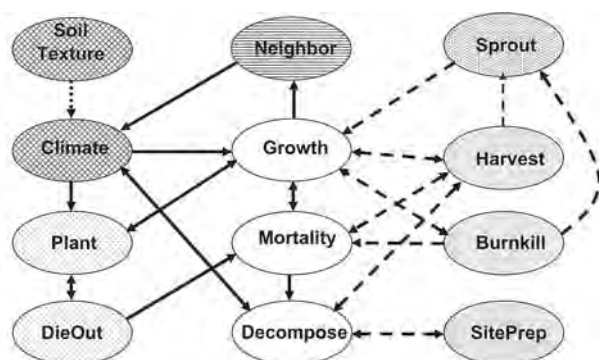

Figure 2. Major modules in STANDCARB model. Crosshatching indicates modules controlling driving variables; stippled population processes; horizontal lines cell to cell interactions; open ecosystem processes; and shaded disturbance processes.

Table 1. Model Predictions of Steady-State C Stores (Mg C ha^{-1}) of Each of the Two Species as well as the Mixed Forest Used in the Simulations

Species	Live	Dead	Stable	Total
Western hemlock	311	147	98	556
Douglas-fir	320	139	109	568
Mixed forest	323	151	102	578

Table 2. Character of the Site Used in the Simulations in Terms of Climate, Radiation, and Soil

Variable	Value
Mean annual temperature	8.6°C
Minimum monthly temperature	−1.5°C
Maximum monthly temperature	18.3°C
Mean annual precipitation	230 cm
Mean monthly diffuse radiation	157 cal cm^{-2} day^{-1}
Mean monthly direct radiation	186 cal cm^{-2} day^{-1}
Soil/rooting depth	100 cm
Soil texture	Loam
Coarse fragments (>2 mm)	None

We also assumed that wood-related production would decline 40% when trees reached their maximal height, a value consistent with observations (Acker and others 2000, 2002).

The climate and radiation data used to drive the simulations were from the H. J. Andrews Experimental Forest, a typical Oregon Cascade site (Table 2). The soil was a loam, with no coarse fragments larger than 2 mm in diameter, and a depth of 1 m. Live biomass accumulation rates of the species were calibrated to match those predicted from yield tables (McArdle and Meyer 1930; Barnes 1962) for productive sites (Site Class 3). Stores in detritus and soils were calibrated to older published data (Grier and Logan 1977), but also match those of other more recent studies of forests that had been disturbed catastrophically by fire 400 to 500 years ago (Harmon and others 2004; Smithwick and others 2002).

Forest Products Stores

Although our primary purpose in these simulation experiments is to examine the potential effect of various practices on C stores in forests, C is stored in forest products following timber harvest. Therefore, we examined two contrasting cases: (1) maximal and (2) minimal storage in long-term products. The former, termed the high storage system was simulated by assuming that 75% of the harvest would be converted to a mix of various long-term forest products that lost C at an average rate of 0.01 y^{-1}. The latter, termed the low storage system, was simulated by assuming 50% of the harvest would be converted to long-term forest products that lost C at an average rate of 0.02 y^{-1}. We assumed that manufacturing efficiencies would not change over the course of the simulations.

Simulation Experiments

For each of the simulation experiments there were five replications of each treatment and these were averaged for analysis. We used a 20 × 20 grid for these simulations, which given the size of our cells would represent a stand of approximately 11.5 ha. We assumed that the height of trees in the cells surrounding the simulation cells was the average of the cells within the simulation. We assumed that trees would fully stock the stand within 5 years, the legal requirement under the state of Oregon's Forest Practices regulations. Except in the case of the first experiment without major disturbances, we allowed the model to run 500 years before disturbance was introduced. To estimate the mean C stored by the particular system, we averaged the C in each aggregate pool being considered (live, dead, stable, ecosystem total, forest products, and total system) over a series of rotations once the values oscillated about a long-term average (that is, a stationary time series in which the starting and ending points of each cycle are the same). The time required to reach a stationary time series increased with the interval between harvests; we ran simulations until at least three stationary cycles were reached and we averaged at least three cycles to reduce the effects of minor variations caused by the stochastic nature of the model. Although a long simulation time was required to produce a stationary time series, the purpose was not to project changes in C stores thousands of years in the future; rather the intent was to calculate a representative average C store. Given the different light requirements of the two species we examined, it is likely that harvesting different amounts and in different patterns would result in different species mixtures. We therefore also calculated the minimum, maximum, and average proportion of tree biomass that was comprised of Douglas-fir.

In the first experiment we allowed the simulation to proceed for 2000 years without a major disturbance. Although this situation would be extremely rare in nature, it allowed us to see how the model would respond over the long-term.

In a second and main set of experiments, we examined the effect of partial cutting in a series of simulations in which live trees in 20, 40, 60, 80, or 100% of the cells had all the stem-related pools cut. This is quite different than in most models, because in those a fraction of the live C is removed over the entire area. Although our treatment had a similar effect, the remaining live mass had an influence on the growth of the next generation of forest through shading, which can have a significant effect on C stores (Smithwick and others 2003). This also altered the microclimate for the dead and stable pools. For all these simulations, we assumed that harvests did not involve the removal of any dead

material and that 95% of the stem-related pools that were cut were removed. We also examined the arrangement of harvest patterns. In a preliminary test (not reported here), we examined the effect of harvesting 20% of the cells over a range of rotation intervals. This indicated a continuous change as we proceeded from 1, 2, 4, 8, and 80 separate harvest blocks out of a possible 400. We therefore contrasted a totally aggregated harvest pattern (contiguous blocks) with a totally dispersed pattern (single cells in a checkerboard of harvest-no harvest) within a stand. Note that our simulations affected all the live trees in the cells being harvested and none of the trees in cells that were not harvested. For each amount of harvest and pattern, we examined harvest rotation intervals of 20, 40, 60, 80, 100, 120, 140, 160, 180, 200, and 250 years.

RESULTS

Experiment 1: No Major Disturbance Simulation

When major disturbances were excluded, the amount of C in all aggregate pools except the stable pool increased for the first 200–300 years (Figure 3). The aggregate stable pool initially decreased because of the temporal lag in the formation of this form of C. After reaching a peak in C stores, all the C pools declined to a long-term steady-state store after 600–700 years of age. The cause of this decline was related to the live pool dynamics; decreases in that pool limited the stores in the dead and stable pools. The decrease in live pools was caused in part by the decline in woody part production related to tree height/age and the appearance of heart-rots in the older trees. The cause of the mid-successional peak in C stores was mostly likely due to the lower light compensation point of
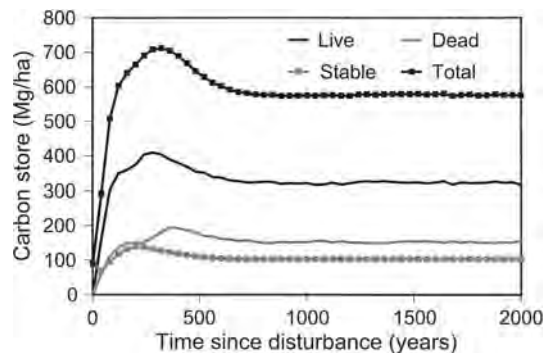
western hemlock which allows more overall C uptake by living trees; both Douglas-fir and western hemlock were present in intermediate stand ages. When these species were run separately this peak in live and dead C was not evident. The steady-state averages of the pools also indicated that the mixed-species forest could store slightly more C than a single species forest (Table 1).

Experiment 2: Partial Cutting Simulations

For both the aggregated and the dispersed cutting pattern within a stand, the average amount of live C increased as the interval between disturbances increased (Figure 4). In general, the less live C that was harvested, the larger the average live C store. These differences were greatest for the shortest intervals between harvests, with a 20% harvest system containing 5.5 to 6 times the average live C of a 100% harvest system. For aggregated cutting patterns, partial cutting stored more live C than 100% harvest for all the rotation intervals examined; however, for the dispersed cutting pattern the 100% harvest started to store more than some of
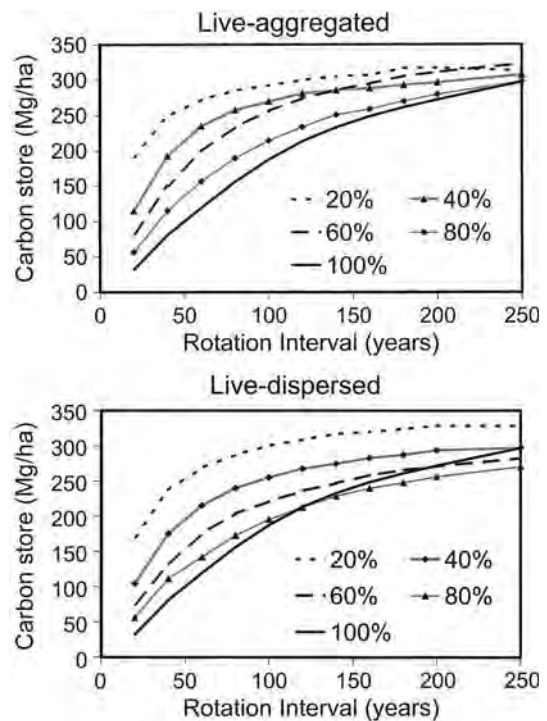

Figure 3. Live, dead, stable, and total C stores for mixed Douglas-fir/western hemlock forest without major disturbance.

Figure 4. Average live C stores over a harvest rotation interval for different levels of removal (percent of cells harvested) and cutting patterns. Aggregated represents one contiguous harvest block and dispersed represents a checkerboard pattern of harvest–no harvest.

the partial dispersed cutting patterns when the interval between harvests exceeded 100 years. The causes of these patterns were several-fold. First, by leaving live C in the forest at all times, partial harvests increased the overall average. Second, as the length of the interval between harvests increased, the differences in heights between the harvested and non-harvested cells also increased. With a dispersed cutting pattern the tallest trees were adjacent to the smallest trees and therefore the limitations due to shading were highest. Although these interactions occurred in aggregated cutting patterns, there were also cases in which small trees were next to small trees and tall ones next to tall ones. This meant that the height difference between harvested and non-harvested cells was less important and may explain why the live C stores for partial harvests with 40–60% live tree removals converged after 125 years. For the 100% cut, these differences in height among cells were minimal, so that at very long intervals of harvest there were more trees with optimal lighting.

Average dead C stores were higher for the aggregated cutting patterns, but the basic trends were quite similar to the dispersed cutting patterns (Figure 5). Unlike the case for live C, there did not

seem to be an interaction between the pattern of cutting and the amount cut, with higher levels of harvest leading to lower average dead C stores. The difference between the 20 and 100% harvest was largest for the shortest interval between harvests; for a 20-year interval the 20% harvest had 1.7 times the average dead store of the 100% harvest.

The stable C store seemed the most sensitive of all the pools in terms of cutting patterns, with aggregated patterns generally storing more on average than dispersed patterns (Figure 6). In the case of the aggregated pattern, the harvest rotation interval that produced the maximal average stable C store was between 20 and 100 years, with the rotation interval needed to reach the maximum increasing as the proportion harvested increased. In the case of the dispersed patterns, there were also maxima, but these appeared to generally occur at rotation intervals of less than 40 years. There are several possible causes for the differences between the two partial cutting patterns, but the most likely is the differences in live C between the two patterns. Live C generally was higher for the aggregated patterns. It was also likely that the environment for decomposition was altered, with
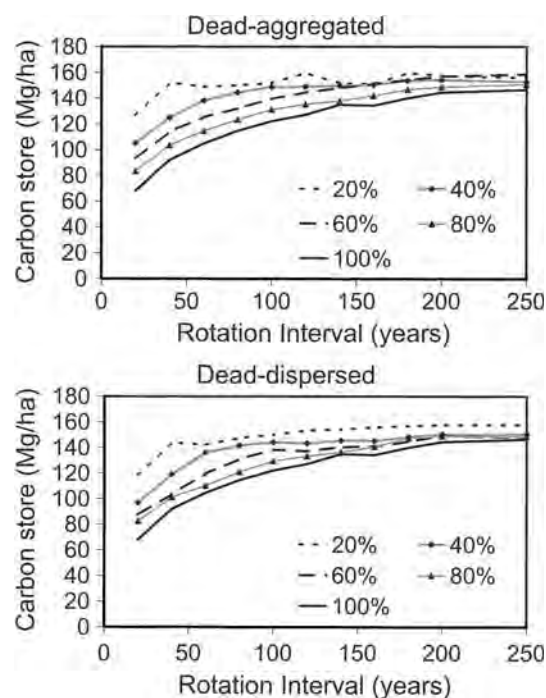

Figure 5. Average dead C stores over a harvest rotation interval for different levels of removal (percent of cells harvested) and cutting patterns. Aggregated represents one contiguous harvest block and dispersed represents a checkerboard pattern of harvest–no harvest.
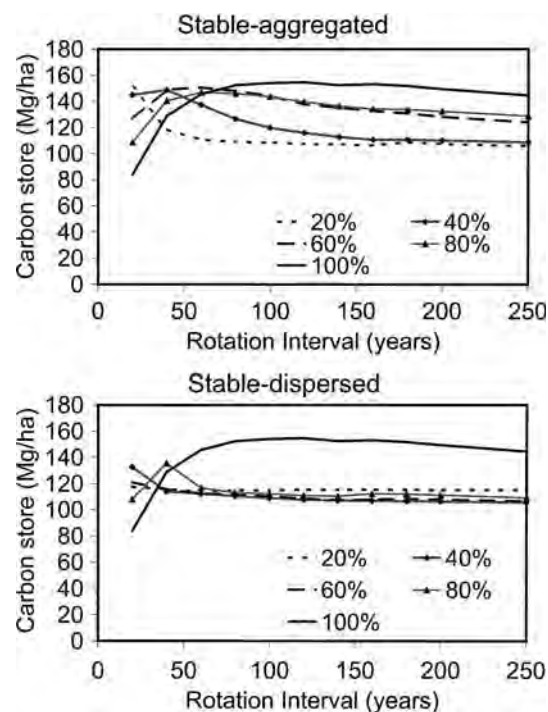
Figure 6. Average stable or soil C stores over a harvest rotation interval for different levels of removal (percent of cells harvested) and cutting patterns. Aggregated represents one contiguous harvest block and dispersed represents a checkerboard pattern of harvest–no harvest.

Figure 7. Average total ecosystem C stores over a harvest rotation interval for different levels of removal (percent of cells harvested) and cutting patterns. Aggregated represents one contiguous harvest block and dispersed represents a checkerboard pattern of harvest–no harvest.
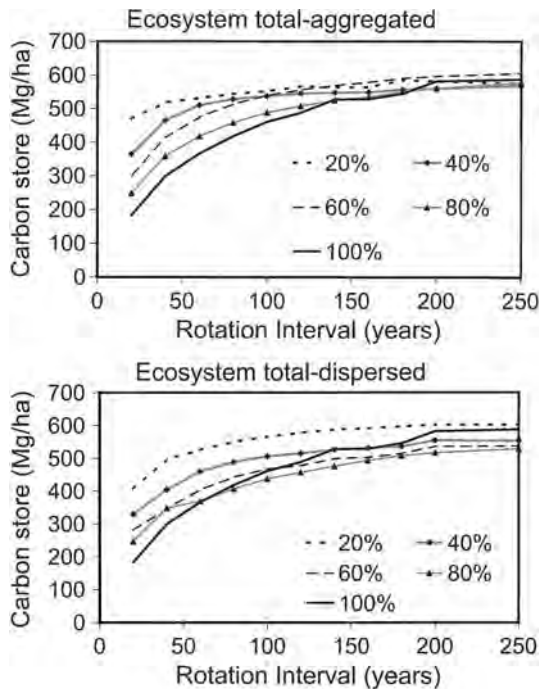
the dispersed cutting having a more favorable environment for decomposition.

The average ecosystem total C increased with the interval between harvests (Figure 7). As with live C there was an interaction between the proportion of cells harvested and the pattern of harvest. In the case of aggregated cutting patterns, the lower the proportion of cells harvested, the more C stored in the system on average at least up to a rotation interval of 150 years. These differences decreased as the interval between disturbances increased and in the case of a 20-year harvest rotation the 20% harvest stored 2.2 times more than the 100% harvest. Once the rotation interval exceeded 200 years there were minor differences between the amounts harvested. For the dispersed cutting patterns, the 100% harvest began to surpass the C amounts in some of the partial harvest systems once the interval exceeded 60 years, and after intervals of 180 years and longer the 100% harvest exceeded all but the 20% harvest system. The difference between the aggregated and dispersed patterns was largely due to the response of the live and stable C pools.

The shape of the response of average C stored in forest products to rotation interval was similar regardless for the pattern of harvest or the forest
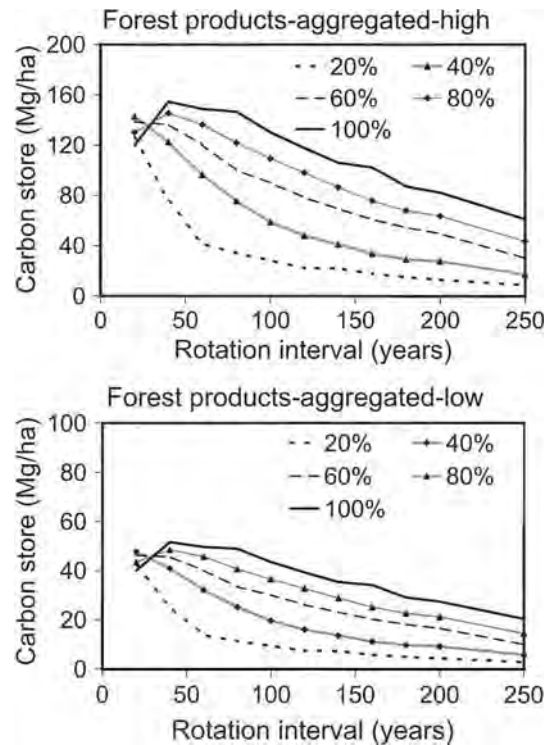

Figure 8. Average forest products stores over a harvest rotation interval for different levels of removal (percent of cells harvested) for the aggregated cutting pattern. Low represents a system in which 50% of the harvested C is converted to long-term forest products with losses of 2% per year; high represents a system in which 75% is converted to long-term forest products with losses of 1% per year. Results for the dispersed cutting pattern were nearly identical.

product storage system examined (Figure 8). As expected, converting more harvest into forest products that lasted longer lead to a larger average C store (that is, approximately a threefold difference) than the converse. Changing the proportion harvested lead to a complex response, with the different proportions having different times to yield the maximal forest products store. In general, the rotation interval to store the maximal amount of forest products increased as the proportion of harvest increased. Specifically, the interval to produce a maximum for the 20 and 40% harvest appeared to be less than 20 years, for 60% harvest it appeared to be 20 years, and for 80 and 100% harvest it appeared to be 40–50 years. Once the maximum was reached, the average store in forest products declined gradually for all the cases except the 20% harvest, which declined very sharply for rotation intervals between of 20 and 60 years. Several factors contributed to the patterns simulated. On one hand, as the forests aged there was more live C to

Table 3. Average Mass of Live C Removed Each Harvest (Mg C ha^{-1} per Harvest) for Different Proportions of Live Harvest, Rotation Intervals, and Cutting Patterns (Aggregated Versus Dispersed)

Rotation interval (years)	Proportion of live stem harvested (%)								
	Aggregated					Dispersed			
	20	40	60	80	100	20	40	60	80
20	35	32	39	36	33	35	40	39	35
40	41	53	74	78	83	40	69	78	77
60	33	77	95	109	121	33	76	99	111
80	36	79	109	130	151	35	77	107	131
100	35	81	119	146	173	35	74	114	143
120	35	77	126	155	188	34	75	116	147
140	36	72	126	157	197	34	69	104	148
160	36	70	127	158	203	35	68	104	149
180	36	70	131	163	208	34	69	107	151
200	34	73	132	168	217	34	69	111	157
250	35	70	126	175	245	35	67	106	145

harvest, and in this set of simulations this would have occurred until a forest age of at least 300 years (Figure 3). Therefore increasing the interval between harvests increased the amount harvested each time (Table 3). On the other hand, increasing the interval between harvests increased the time forest products were lost without replacement and tended to reduce the average amount harvested per year. This lowered the average C store in the forest products pool. Reducing the proportion of cells harvested effectively reduced the amount harvested, but this was countered by the fact, at least for very short rotations, that those trees that were not harvested become larger by the time of the next harvest. Eventually, this effect of increasing the

average size of trees was negligible. For a 100% harvest system, all the trees were generally the same size, and the maximum tree size was reached at a longer interval between harvests.

The response of the average system total C differed with the degree of aggregation in the cutting pattern (Figure 9). For all the management systems examined, there was an increase in average system C stores as the interval between harvests increased. The addition of forest products narrowed the difference between the partial and complete harvests. In the case of the low forest products storage system, these increases were not enough to counter decreases in ecosystem C stores. In the case of the high forest products storage system, these increases
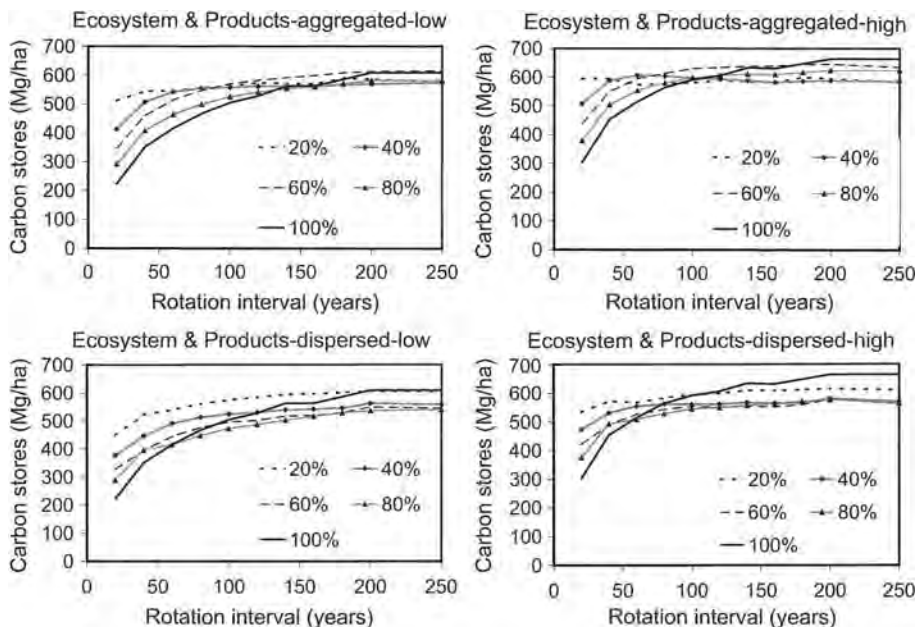
Figure 9. Average total system (ecosystem and forest products) C stores over a harvest rotation interval for different levels of removal (percent of cells harvested) and cutting patterns. Aggregated represents one contiguous harvest block and dispersed represents a checkerboard pattern of harvest–no harvest. Low represents a system in which 50% of the harvested C is converted to long-term forest products with losses of 2% per year; high represents a system in which 75% is converted to long-term forest products with losses of 1% per year.

were enough to counter decreases in ecosystem C stores for the 100% harvest system once the rotation interval exceeded 125–175 years. At the shortest intervals between harvests the 20% harvest stored approximately two times more than the 100% harvest systems. For the aggregated harvest patterns, average total systems stores became very similar once rotation intervals exceeded 160 years, although it should be noted that the stores in the 100% harvest high forest products storage system continued to increase. For the dispersed harvest patterns, harvesting a lower proportion of cells resulted in more C stored in the system for short rotation intervals. However, in comparison to 100% harvest, dispersed patterns of harvest began to store less C once the interval exceeded 60 years, and at an interval of 200 years the 100% harvest generally stored more.

The amount and pattern of harvest had major effects on tree species composition (Figure 10). In the case of complete harvest (that is, 100%),

Figure 10. Proportion of the stand live C comprised of Douglas-fir (*P. menziesii*) for different levels of removal (percent of cells harvested) and cutting patterns. The *bars* represent the range and the *white line* represents the average over a rotation interval. Aggregated represents one contiguous harvest block and dispersed represents a checkerboard pattern of harvest–no harvest.

Douglas-fir remained the dominant tree species for rotation intervals from 20 to 200 years. For partial harvests, there were three trends: (1) as the proportion of cells harvested increased, the proportion of Douglas-fir increased; (2) as the interval between harvests increased, the proportion of Douglas-fir decreased; and (3) aggregated patterns of harvest lead to higher amounts of Douglas-fir than dispersed ones. Perhaps the most dramatic example of the latter was for 20% harvest with a 20-year rotation interval. When the harvest pattern was aggregated Douglas-fir comprised 70% of the live tree C. However, for a dispersed harvest pattern with the same amount and interval of harvest, Douglas-fir averaged 12% of the live tree C. In fact there were few combinations of harvest amount and interval that lead to Douglas-fir comprising more than 30% of the live C at any time between harvests when the dispersed harvest system was used.

Discussion

Our simulation experiments examined the effects of rotation length and proportion of trees harvested on forest C stores. Our results match those of an earlier study using STANDCARB (Harmon and Marks 2002), but here we more thoroughly examined the effects of partial harvest and the interactions of trees of different ages. The current experiments show that partial harvest of live trees may lead to higher C stores than complete harvest, especially when the interval between harvests is short. We found that arrangement of the partial harvests employed within a stand (that is, aggregated versus dispersed) had a major effect on species composition, but had smaller effects on C stores than the proportion removed or the rotation interval.

Given that our model included the effect of remnant trees on the rest of the system, it would appear that the presence of live trees does not reduce net primary production (NPP) or increase decomposition enough to overcome the effect of leaving more live C in the forest throughout the rotation. Partial harvesting can damage remaining trees and increase their mortality rates for several years (Beese and Bryant 1999). Our model did not consider these effects and although inclusion of them would have made the model more realistic, it is unlikely that they would have been large enough to alter the general conclusions. For example, there was at least a twofold difference between a 20% versus a 100% harvest system when a 20-year interval harvest was used. NPP of the surviving trees would have to be permanently reduced two-
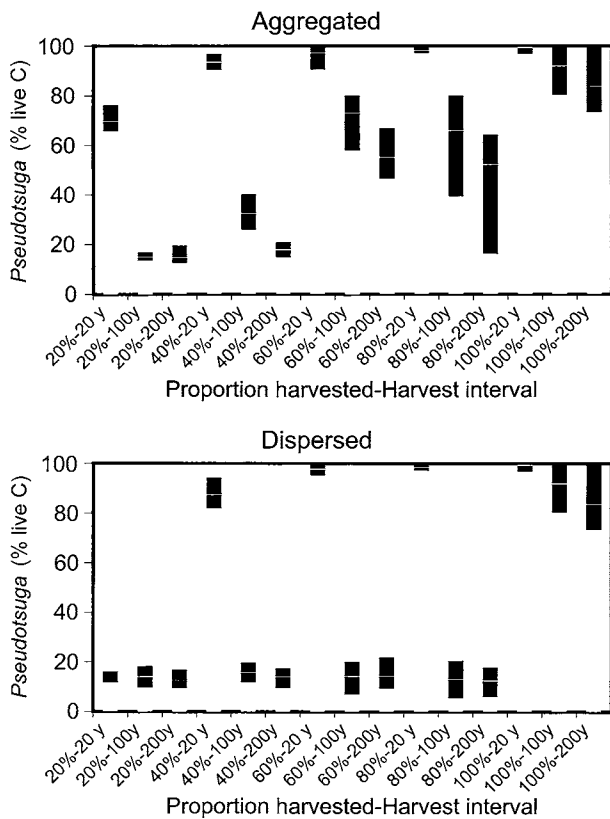
fold for these two systems to store the same amount of C. Beese and Bryant (1999) report that 5–25% of the remaining trees died within a 3-year period when 95% of the tree volume was harvested, but these tended to be the intermediate crown class indicating less biomass would have been lost. Our simulations were probably more similar to the patch cuts examined by Beese and Bryant (1999) which had a 3-year cumulative mortality of 1%, a value less than that used for the annual mortality in our simulations.

Our model did not include nutrient cycling and this probably influenced the impact of short rotations on C stores. Our results would be representative as long as the site preparation treatments and harvest rotation intervals do not change the overall nutrient availability of the forest. Given the fact only tree stems were removed in our simulations, nutrient status should not have declined greatly. However, the results of others (for example, Seely and others 2002) indicate that for short rotations there could be a 10% drop in C stores associated with nutrient limitations. Thus the differences we estimated between short and longer rotations as well as partial versus complete tree stem harvest are likely to be conservative.

Our result that increasing rotation length increases forest C stores has been found in numerous studies. In Finland, both Liski and others (2001) and Pussinen and others (2002) found that longer rotation lengths stored more C in forests than shorter ones. This was also true in a larch-dominated boreal forest in China (Jiang and others 2002), western Canadian boreal forests (Seely and others 2002), forests in the United Kingdom (Dewar and Cannell 1992; Thornley and Cannell 2000), and tropical plantations (Schroeder 1992). At a very general level, this result is due to two facts related to inputs versus removals of C. First, the average input through photosynthesis generally increases as rotation interval increases. Although in our model gross primary production decreased when forests reached their maximum height at around 200–300 years, this did not alter this basic trend in average NPP over the interval between disturbances. Thus the longer the rotation interval, the higher the average NPP input to the system. Second, the proportion of C removed also controls the steady-state or in our case the average store of C (Olson 1963). In the case of harvest of live C, the longer the interval between harvests the lower the effective proportion of C removed per harvest. This removal effect would also be true for site preparation treatments such as prescribed fires that remove C from dead stores.

Although the majority of studies have examined complete harvest of trees, several have examined partial harvests. When clearcuts are employed, whole tree harvest reduces forest C stores more than removal of just the stem (Jiang and others 2002). Thinning within stands between complete harvests generally decreases C stores in live and total forest C stores (Dewar and Cannell 1992; Balboa-Murias and others 2006). Partial harvest within a stand leads to intermediate levels of C stores in forests (Thornley and Cannell 2000) between the unharvested system and the complete harvest system. Both results make sense relative to the balance of inputs versus outputs described above. Thinning with complete harvest reduces the input and also increases the proportion of C removed during a rotation interval. Partial harvests at the stand level also reduce average inputs through NPP and also increase the proportion of C removed relative to the no harvest system, but to a far less degree than the complete harvest system.

Despite the preponderance of evidence that short intervals between clearcut harvests of stands store less C in forests than long ones, there are those promoting the use of short rotation plantation systems to sequester additional atmospheric C. To some degree this originates from the failure to correctly scale time-specific stand results to the long-term, broader scale (Harmon 2001). Although it may be true that particular ages of young forest remove more C from the atmosphere than older (or younger) ages, one needs to consider the C balance of all age classes present in a disturbance regime. Forests lose C immediately after disturbance; the amount is dependent on the amount of legacy C remaining in the form of dead material and in soil as well as the rate at which the new forest reestablishes (Harmon in press). This initial period of C loss often offsets the later period of C uptake. Because one cannot have older forests without younger forests, it is quite possible for the later periods of gains to be completely countered by the earlier period of C loss. Indeed, that is why in all systems analyzed to date, C stores in forests oscillate about a mean over the rotation unless one is transitioning from one disturbance regime to another (Dewar and Cannell 1992; Smithwick and others 2007). If a shorter rotation is replaced by a longer one, the amount of C stored in the system increases. If a longer rotation is replaced by a shorter one, then the amount of C stored in the system decreases. Thus, replacement of a long natural disturbance cycle by short rotation forestry generally leads to decreases in C stores in the forest system (Harmon and others 1990). Conversely,

replacement of a short natural disturbance cycle by a longer harvest cycle generally leads to increases in C stores (Kurz and others 1998; Seely and others 2002).

Another possible explanation for suggesting short rotation harvests is that this will increase C stores related to forest products. Our results and those of others (Dewar and Cannell 1992; Liski and others 2001; Pussinen and others 2002; Seely and others 2002) show that the average amount of C harvested per year decreases once a peak is reached at relatively short rotations that are near the culmination of mean annual increment. If forest products could be created without C loss in manufacturing and use, then harvesting forests at the culmination of mean annual increment would maximize C stores in forest systems provided enough time elapses. Unfortunately, with the possible exception of biofuels which directly count as fossil fuel offsets, almost all forest products suffer significant losses in manufacture and use (Harmon and others 1996). Those studies that have included forest products in the analysis, including ours, have found that forest products do not comprise a large fraction of the forest systems C stores (Dewar and Cannell 1992; Pussinen and others 2002; Seely and others 2002; Harmon and Marks 2002). The fraction of total forest system C stores comprised by forest products increases as rotation interval decreases, but these are relative gains and not gains in the entire system; they just offset a fraction of the losses occurring in the forest.

Our assumption that a constant fraction of harvest, regardless of rotation length, is converted to long-term stores might be challenged, as shorter intervals between harvests may lead to a smaller fraction being converted to long-term forest products (Bourque and others 2007). This would mean, for example, that the total system C stores for the 100% cut for short rotation intervals would be closer to the low storage system results; conversely for the longer rotation intervals the values would be closer to the high storage system results. On the other hand, more small trees are being converted to long-term products with improved milling technology, which might tend to make the conversion rates to long-term products similar to that of larger trees. For partial harvests, short intervals between partial stand harvests may produce trees that can be converted into long-term forest products. Rather than model all these complexities, we chose to simulate two extreme cases which can be used to model any set of assumptions regarding the effect of tree size on the storage of forest products. Moreover, although a more realistic model might modify the exact shape of the curves, it would be unlikely to change the basic conclusion that forest products are not the major share of total system C stores.

Although we estimated the stores in forest products, we did not include the so-called substitution effects of using wood versus other more energy intensive materials for construction. As pointed out by Hennigar and others (2008), there is little consensus on the values to be used (that is, they vary 10-fold). The other issue is that these estimates represent maximal values that assume that all future buildings will be primarily constructed of materials other than wood. Thus, it counts the substitution effect over an over even when a wooden building is replaced by a wooden building. Although this assumption simplifies calculations, it does not necessarily lead to reliable estimates of the most likely substitution effect over time. If we counted the substitution effect based on the sustainable store of wooden buildings and assumed half the harvest was devoted to building construction, then the additional ''store'' due to substitution would have been in the range of 10 to 100% of the average forest products store we reported depending on the substitution effect assumed (0.2–2 Mg C for each Mg C of building stores). Although this would have made the total systems' curves in Figure 9 flatter, with the shorter rotations more similar to the longer ones, it would not have made the partial harvests store less C than the 100% harvest. This is because the forest stores for all harvest systems are highest for the shorter rotation intervals.

There are conditions in which forest management can increase C stores relative to existing systems. Intermediate rotation intervals can increase C stores over longer rotations when there is a major decline in live C stores in older forests (Johnsen and others 2001). It is clear that altering species can greatly change the mean C store over a rotation and replacing a slowly growing species with one with rapid growth can decrease the time to reach a given C store (Dewar and Cannell 1992; Liski and others 2001; Hennigar and others 2008). Similarly, increasing nutrient availability through fertilization or improved soil management can also decrease the time to reach a given C store (Pussinen and others 2002; Jiang and others 2002; Seely and others 2002). Although we are unaware of any examination of the effect of changing decomposition rates on forest C stores, reducing decomposition rates would likely increase C stores, thereby slowing decomposition rates may also reduce rates of nutrient cycling and ultimately reduce inputs through NPP.

Our simulation experiments ignored the effect of changing climate. Climate change is likely to have multiple effects on rates of nutrient cycling, decomposition, and disturbance. The exact effect is likely to be very much dependent on the site in question. Simulating a warmer climate in Finland, Pussinen and others (2002) found that forest C stores decreased largely because increased losses in soil C were not offset by increases related to NPP. A warmer and potentially drier climate may also lead to the increased disturbance by fire, drought, and insects. Breshears and Allen (2002) have raised the possibility that increasing ecosystem C stores now may result in major releases in the future when disturbance increases. An analysis by Kurz and others (2008) indicated that changes in disturbance regimes associated with climate change are likely to shift Canada's managed forest from a C sink to a source in the next decades. We agree that had we factored increasing rates of disturbance into our analysis, our stores estimates would have been lower. The degree of decrease would be a function of how synchronous these disturbances are on a broad scale. If these disturbances do occur synchronously on a broad scale, then the reduction in C stores in forests could be substantial.

CONCLUSIONS

Our simulation modeling study indicates there are multiple ways to achieve similar C stores in the forest system. Our studies and those of others have shown that increasing the interval between disturbances, in our case harvests, increases the C stores in the forest ecosystem (that is, live, dead, stable, or soil). Harvesting at short intervals will result in more stores in forest products; however, because of the potentially high C losses during manufacturing, this increase in forest products stores does not completely offset the losses from the forest ecosystem. C stored in the entire forest system (that is, ecosystem and forest products) is lower when the interval between harvests is decreased. Frequent partial harvest of forest stands can store as much C in the entire forest system as long intervals between complete harvests of trees in a stand. This occurs because with partial harvest in a stand, the live C store is not reduced to zero and thus remains higher for a longer period in the rotation than when complete harvest occurs. With complete harvest of the trees in a stand, the live C store has to accumulate from zero stores. Partial harvest of trees may also lead to reduced C stores under certain conditions. This would include a change in the species present as well as the amount of damage caused to the remaining trees. We did not examine the latter effect, but this likely could be reduced if precautions were taken during harvest, although this would increase harvesting costs. In our simulations, there was a major change in species present under partial versus complete harvest within a stand. The increased presence of western hemlock under partial harvest of stands in our simulations should have lead to increased losses from live trees as the amount of heart-rot in that species is considerably higher than for Douglas-fir. However, we found relatively small differences in C stores when the species mixture of forests changed. Nonetheless, our findings on partial harvest of stands could vary in other systems depending on the degree the species change and the difference in species characteristics controlling C stores. The generality of our findings should therefore be tested in other mixed-species forest stands.

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Public land, timber harvests, and climate mitigation: Quantifying carbon sequestration potential on U.S. public timberlands

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Abstract

Scientists and policy makers have long recognized the role that forests can play in countering the atmospheric buildup of carbon dioxide (CO_2), a greenhouse gas (GHG). In the United States, terrestrial carbon sequestration in private and public forests offsets approximately 11% of all GHG emissions from all sectors of the economy on an annual basis. Although much of the attention on forest carbon sequestration strategy in the United States has been on the role of private lands, public forests in the United States represent approximately 20% of the U.S. timberland area and also hold a significantly large share (30%) of the U.S. timber volume. With such a large standing timber inventory, these forested lands have considerable impact on the U.S. forest carbon balance. To help decision makers understand the carbon implications of potential changes in public timberland management, we compared a baseline timber harvest scenario with two alternative harvest scenarios and estimated annual carbon stock changes associated with each. Our analysis found that a ''no timber harvest'' scenario eliminating harvests on public lands would result in an annual increase of 17–29 million metric tonnes of carbon (MMTC) per year between 2010 and 2050—as much as a 43% increase over current sequestration levels on public timberlands and would offset up to 1.5% of total U.S. GHG emissions. In contrast, moving to a more intense harvesting policy similar to that which prevailed in the 1980s may result in annual carbon losses of 27–35 MMTC per year between 2010 and 2050. These losses would represent a significant decline (50–80%) in anticipated carbon sequestration associated with the existing timber harvest policies. If carbon sequestration were valued in the marketplace as part of a GHG offset program, the economic value of sequestered carbon on public lands could be substantial relative to timber harvest revenues.
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Keywords: Public timberland; Forestry; Climate change; Carbon sequestration

1. Introduction

Forest ecosystems play an important role in the global carbon cycle, absorbing large amounts of atmospheric carbon dioxide (CO_2) through photosynthesis and emission of CO_2 to the atmosphere through respiration, decomposition, and disturbances such as timber harvesting, fire, pest infestations, and land use change. Globally, terrestrial ecosystems are a net carbon sink[1] because removals and storage of CO_2 from the atmosphere (about 2300 million metric tonnes of carbon [MMTC] per year) exceed emissions (1600 MMTC per year) (IPCC, 2000). Most of the terrestrial sink is in forests. The global carbon balance masks some regional disparities; for instance, tropical forests are a source of emissions as deforestation outpaces regrowth, while the reverse is true currently in temperate forests, which are a net sink. The latest data for the United States indicate that land use, land use change, and forestry (predominately forest) comprises a net carbon sink of over 210 MMTC per year, offsetting about 11% of the country's GHG emissions (U.S. EPA, 2006).[2]

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[1] A carbon pool is a net sink if, over a certain time interval, more carbon is flowing into the pool than is flowing out of the pool. Conversely, a carbon pool can be a net source of CO_2 emissions if less carbon is flowing into the pool than is flowing out of the pool.

[2] Note that EPA data are reported in teragrams (million metric tonnes) of CO_2 equivalent (Tg CO_2). One ton of carbon equals 3.667 tons of CO_2.

EXHIBIT 12

Expanding the area of land in forest cover, avoiding deforestation, and managing existing forests to store carbon in ecosystem stocks for longer periods by increasing the length of time between harvests can increase the net size of the carbon sink or, in some cases, turn a source into a sink. This has been recognized in the global and domestic policy arenas as a mix of mandatory and voluntary initiatives have sprung forth in the last decade that incentivize expansion of carbon sinks as a climate mitigation strategy. In the United States, much of the emphasis has been on incentives to expand carbon sinks on private lands (U.S. EPA, 2005; Lewandrowski et al., 2004; Richards and Stokes, 2004; McCarl and Schneider, 2001; Adams et al., 1999; Stavins, 1999; Plantinga et al., 1999). The more limited work regarding estimates of public lands' contribution to the U.S. carbon sink pertains to the projection of the status quo or business-as-usual case or BAU (Turner et al., 1995; Smith and Heath, 2004) or to regional contributions (e.g., Alig et al., 2006). Yet public timberlands constitute a sizable share of the U.S. forest resource in terms of both land area and timber volume (see Section 2) and thereby provide a potentially important resource to manage for climate change mitigation.

This paper departs from the literature by examining public timberlands' forest carbon sequestration potential at a national scale, not only under BAU conditions, but also under changes in forest management. The change in public forest management addressed in this paper is the level of allowable timber harvests, with two alternative scenarios to BAU defining the range of options from no timber harvest (elimination of all timber harvests on public timberlands) to a return to the historically high harvest period of the 1980s. Public land managers could consider other forms of forest management, such as modified rotations and intensive management of inputs, but those remain outside the scope of this paper.

The next section of the paper provides a brief overview of the public forestland resources in the United States, followed by a description of the data and methods used in the analysis and presentation of results for public timberlands. The paper ends with policy conclusions that can be drawn from the study and suggestions for future work.

2. Public timberland in the United States

The contiguous 48 (C48) states have approximately 228 million acres of public forests. Approximately 80% of this land, or 182 million acres, is in federal ownership (W.B. Smith et al., 2004; J. Smith et al., 2004). States, counties, and municipalities own the remaining 46 million acres; approximately 61% (138 million acres) of the public forestland is classified as *timberland* because it meets site productivity criteria and is not withdrawn from timber utilization by statute or administrative regulation.[3] Public timberland in the C48

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[3] Timberland is defined as forestland that can produce 20 ft[3] of industrial wood per acre per year in naturally regenerated stands and that is not withdrawn from timber utilization by statute or administrative regulation (W.B. Smith et al., 2004; J. Smith et al., 2004).
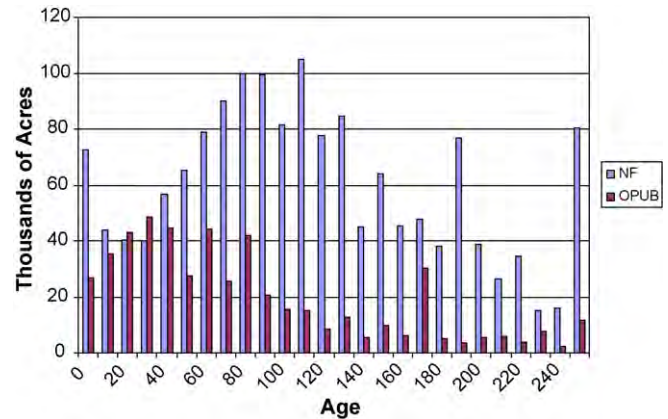

Fig. 1. Distribution of national forests and other public lands acres by age class: 2000.

states is concentrated in the West (west of the 100th meridian), which holds about 80% of U.S. public forestland. The top six states in order of public timberland area are Oregon, Idaho, Montana, California, and Colorado/Washington (tie).

Although the public owns a significant share of U.S. timber resources, they contribute a much smaller fraction of total U.S. timber removals. Public timberlands held 41% of growing stock inventory in 2001. The largest concentration of public timberlands is on National Forest (NF) lands, which alone held 30% of U.S. timber growing stock in 2001 (W.B. Smith et al., 2004; J. Smith et al., 2004). However, public timberlands produced only 8% of the U.S. timber removals in 2001, with NF lands providing just 2% of U.S. timber removals in 2001. Public policy makers have reduced timber harvests in favor of other nontimber outputs (e.g., wildlife, recreation, watershed protection, scenic amenities) since the late 1980s (Wear and Murray, 2004). Note that annual mortality is larger in volume than growing stock removals on both NF and other public (OPUB) timberlands, while net growth volume is at least two times the amount of mortality volume for those ownerships, leading to a net accumulation of growing stock and carbon. For example, in the case of NF timberlands, many acres are in young age classes with relatively rapid growth. However, public timberlands hold a relatively large share of the nation's older timber on timberland, especially on NFs, as shown in Fig. 1.

3. Analysis scenarios

Current management of U.S. public forestlands centers on a mix of environmental and socioeconomic objectives. For example, the Northwest Forest Plan (NWFP) covers almost 25 million acres and addresses northern spotted owl population and habitat, marbled murrelet population and habitat, late successional old-growth habitat, watershed conditions, and socioeconomic characteristics. Monitoring efforts are also underway to evaluate the success of the NWFP in achieving its objectives based on new scientific knowledge on key topics that include old-growth forest habitat, watersheds, and rural economies. Currently, carbon sequestration is more a by-product

than a primary management objective of the plan, but that could change with the renewed interest in climate change mitigation at the federal level in the United States (Paltsev et al., 2007).

For this analysis, we characterize a baseline (referred to as the BAU timber harvest scenario) and compare and contrast annual carbon stock changes associated with two alternative timber harvest scenarios. The baseline scenario for public timberlands identified by Mills and Zhou (2003) was derived from the USDA Forest Service's (USFS's) Washington office and represents expectations at that time based on guidelines of USFS policy. Timber harvests are drawn from a characterization that we call a "removals scenario" after Mills and Zhou (2003) and were allocated according to the number of acres in each age class (see below). Regeneration volumes were based on ATLAS model (Mills and Kincaid, 1992) projections of forest inventory (see below for details).

The first alternative scenario, "no harvest," eliminates timber harvest completely and thereby reflects nontimber forest management objectives in the extreme. NF timber stands are assumed to grow without any timber harvest-related disturbances for the next 100 years. Mills and Zhou (2003) assumed that other naturally occurring disturbances such as fire, insects and diseases, and other natural mortality would remove timber volume and require the natural regeneration of an additional 140,000 acres annually. This acreage number came from the average rate of acres disturbed in the 10 years preceding the publication of "Projecting National Forest Inventories for the 2000 Resources Planning Act (RPA) Timber Assessment" by the USDA Forest Service (Mills and Zhou,

2003). The disturbed acres were taken from the two dominant forest types, those occupying the largest acreage. Within the two dominant forest types, disturbed acreage was removed from every age class above the minimum harvest age for the ATLAS model.

The second alternative, "high-harvest/pre-1989" scenario, follows timber harvest levels as depicted in the 1989 USFS's Timber Assessment (USDA Forest Service, 1990), the most recent period of timber harvesting on public timberlands that is above historical averages. These timber harvest levels, as reported in the 1989 RPA Assessment, for NFs came from the forest plans in effect or drafted in 1987 in response to the National Forest Management Act of 1976. NFs at that time provided about two-thirds of timber harvests from public timberlands, and NF timber harvest was assumed to increase by about 400 million ft^3, from 2.3 billion in 1986 to 2.7 billion by 2040. The 1986–2040 projected harvest levels took into consideration the anticipated impacts at that time of the Threatened and Endangered Species Act of 1973. The scenarios are intended to convey differences in forest carbon and carbon that is disposed of off-site – in products, landfills, and energy use – under different timber harvest assumptions.

As shown in Fig. 2, the BAU timber harvests per decade from public timberlands in 2010 range from 15 to 20 billion ft^3 during the period of the analysis. Approximately two-thirds of the harvests come from other public timberlands (see Fig. 2a and b), a reverse of the relative contributions of the two major sources of public timber harvest in 1986. In contrast, the pre-1989 scenario harvests per decade are significantly higher and
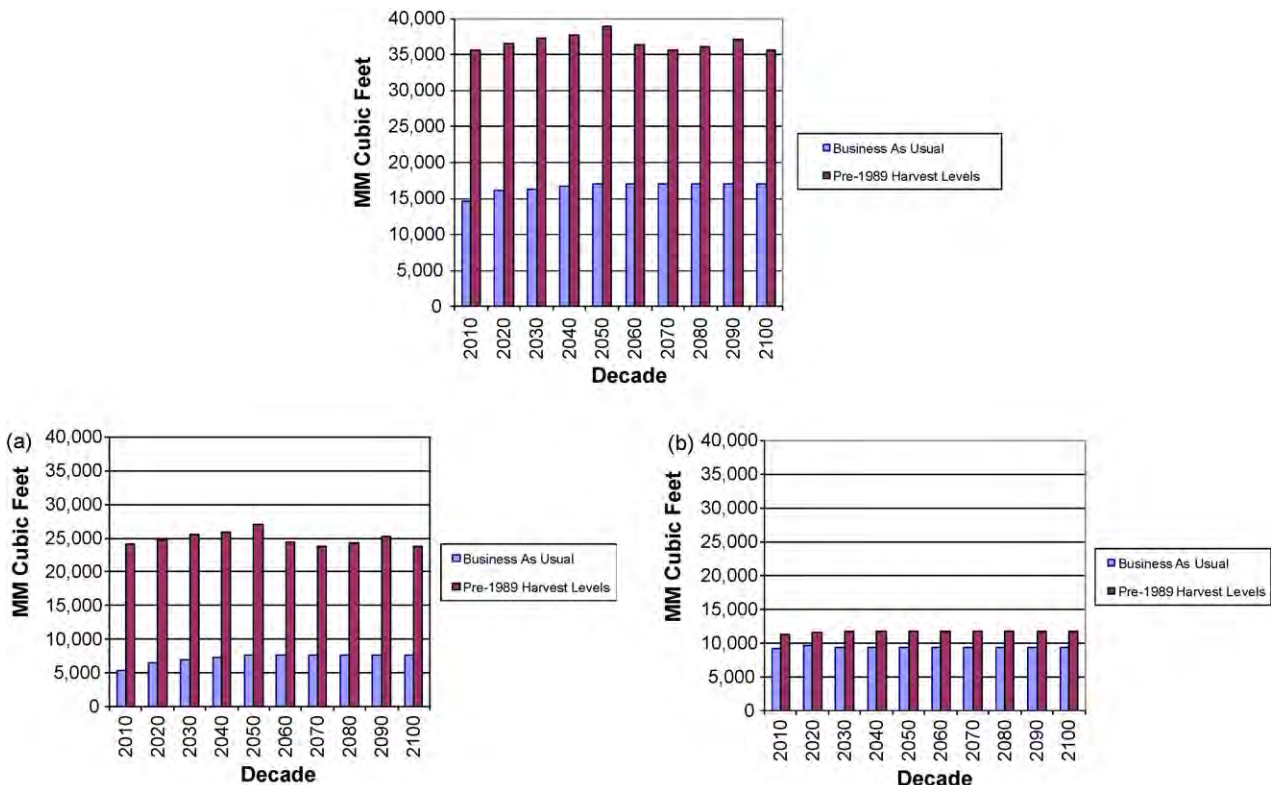

Fig. 2. Total public timberland harvests by decade and scenario 2010–2100. This includes harvests from (a) National forests and (b) other public lands.
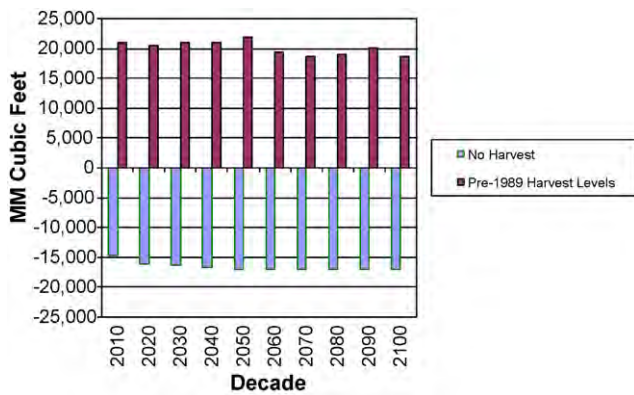
Fig. 3. National forests and other public lands: changes from BAU harvest volume by scenario.

range from 35 to 40 billion ft^3. Timber harvests in these scenarios increasingly rely on NF lands, with approximately two-thirds of the decades' total harvests coming from NFs.

As shown in Fig. 3, the no-harvest scenario reduces public timber harvests by approximately 15 billion ft^3 per decade. Presumably, this scenario will increase carbon stocks by avoiding carbon losses associated with converting standing forests into wood products. In contrast, the pre-1989 scenario increases baseline public timber harvest levels by approximately 20 billion ft^3 per decade. As a result, carbon losses will increase as more timber is removed. Our analysis is designed to estimate, compare, and contrast annual carbon stock changes associated with the two radically different timber harvest scenarios.

4. Data and methods

Simulating public forest management requires data specific to public timberlands on a range of variables, including land class, timberland area, forest type, timber yields for specified land management trajectories, growing stock or biomass volume by age class, site productivity, and regeneration yields. These data also need to be linked to data or models that quantify the relationship between these variables and carbon storage.

4.1. Timberland inventory

Public timberland data were obtained from ATLAS modeling used in the 2000 RPA Timber Assessment. We assembled the inventory data, along with existing and regenerated timberland yield projections for NF aggregates, using strata identical to those used in the private timberland tables in the Forest and Agricultural Sector Optimization Model-Greenhouse Gases, or FASOMGHG (McCarl et al., 2005; Adams et al., 1996). Data for projections came from USFS Forest Inventory Analysis (FIA) permanent sample plots. Collected data for NF and OPUB timberlands were stratified by region, ownership, forest type, and age class (Mills and Zhou, 2003). Assembling inventory data included identifying public timberland area and growing stock volumes by age, land class, region, forest type, site class, and broad management intensity

class. Timber growth and yield relations were developed from a broad cross section of field plots. In ATLAS, timber management intensity classes correspond to a specific regime of silvicultural treatments to represent a regional average response for a particular forest type. The management intensity classes are initially populated with a timberland inventory derived from forest survey plots. Empirically derived parameters dictate forest stand development in terms of net growing stock volume as the ATLAS model simulates growth, timber harvesting, and regeneration. The ATLAS modeling approach has been applied in regional and national timber resource assessments, for modeling of changes on both private and public timberland.

Mills and Zhou (2003) provided public timberland data, based on USFS Forest Inventory and Analysis (FIA) plots. We used 5-year age classes to represent public timberlands, up to ages of 250+ in all regions except the South, where the oldest age class was 90+ for the generally younger forests held there. Some Northeast and South Central plots did not have age class data assigned by the FIA units; for these plots age was assigned using a method that considers volume and stocking.

Age class is one of the parameters used to calibrate the yield functions that determine volume; another parameter is region. Nine timber supply regions were designated to categorize the United States described in Mills and Zhou (2003). These regional designations help organize forest area into areas of similar growth characteristics, making the model more accurate than if only one yield function were used for the entire United States.

Across all regions, forestland was aggregated into softwood and hardwood forest-type groups. In the Pacific, Rocky Mountain, Lake States, and Corn Belt regions, all land with trees over 250 years old was aggregated into the age cohort of >250. In the Southern regions, land with trees 90 years or older was aggregated into the uppermost age cohort of >90. In the Southeast and South Central regions, ATLAS was unable to project yields of older stands for the entire 100-year time horizon. In the older stands, the total volume within the strata was used to extrapolate yield curves throughout the projection period. Based on data limitations, each stand in the inventory was assigned a medium site class. Public timberland only occurred on the FORONLY ("forest only") land class, areas not suitable or not available for conversion to crop or pasture. Because of this limitation, no conversion is allowed to agriculture on public land, which, regardless of whether it is biophysically feasible to do so, is not likely to occur for legal and political reasons.

Timber management intensity on NF timberland consists of three categories: a low intensity of even-age management, uneven-age management, and reserved (Mills and Zhou, 2003). Other public timberlands only had the low intensity of timber management. With a low intensity of timber management, no significant intermediate stand treatments are assumed to occur between stand establishment and final harvest.

Timber stands are final harvested over a range of stand ages. The uneven-age regime allows partial cutting (Mills and Zhou, 2003), where a treatment removes a portion of timber volume to reflect a stand subject to multiple entries. Timberland in a

reserved class is not available for timber harvest, but growth of the reserved stands is projected forward in time. The number of acres assigned to these regimes was derived from a survey of NF regional silviculturists (Mills and Zhou, 2003). The majority of the NF acres are assigned to either the partial cutting or reserved classes.

Timber yield estimation for regenerated stands was based on the ATLAS model approach. ATLAS calculates regeneration failures by region and used lagged yields to reflect failed cases. ATLAS has acres remain in the youngest timber age class for an extra 5 years for the South or 10 years elsewhere. Lacking data on pre- and postdisturbance forest types, all regenerated stands returned to the same forest type from which they originated in the same proportions of hardwood and softwood as they had before disturbance.

Assumptions concerning future harvest patterns and land base changes included that the public timberland area does not change over the planning horizon. All clear-cut harvested acres are regenerated as a single stratum with the other harvested acres in that same period and region. Harvests are distributed according to area in each age class; no age class or management intensity is excluded from harvest except for reserved acres.

4.2. Carbon projection methods

Our analysis calculates the stocks and flows (fluxes) of carbon on public timberlands in the United States, including NF and OPUB lands. These estimates are based on USFS projections of future timberland inventories and timber harvest levels, forest carbon accounting equations of the USFS FORCARB2 model (see below), and wood product accounting methods based on the previous work of Smith et al. (2006). As shown in Fig. 4, the carbon accounting framework separates forest carbon calculations into two parts: the accumulation of forest ecosystem carbon as forested stands mature before harvest and the disposition of carbon into various destination pools after the point of harvest. We discuss each component below.

4.2.1. Forest ecosystem carbon accumulation before harvest

On-site carbon accounting closely mirrors the FORCARB2 system used by the USFS in their aggregate assessments of

Fig. 4. Carbon accounting framework.

forest carbon sequestration. Using this framework, carbon accumulates in four pools and we describe each below:

- trees
- understory
- forest floor and coarse woody debris
- soil

4.2.1.1. Trees. In FORCARB2, tree carbon is a function of two factors: merchantable timber volume and parameters of a forest volume-to-biomass model developed by USFS researchers (Smith et al., 2003). Merchantable volume, by age, on each representative stand is obtained from the timber growth and yields tables in the ATLAS model described above. Tree carbon includes live and standing dead tree carbon and is calculated using the parameters of the forest volume-to-biomass model equations for live and dead tree mass densities (above and below ground) in Smith et al. (2003).[4] Birdsey's (1992) assumption that mass of wood is approximately 50% carbon is used to derive the associated quantity of carbon:

$$C^{\mathrm{R}} = \left(\frac{D^{\mathrm{L}} + D^{\mathrm{D}}}{U^{\mathrm{B}}} \right) \times 0.5, \tag{1}$$

where live and dead tree biomass are computed as

$$D^{\mathrm{L}} = F^{\mathrm{w}} \times \left(G^{\mathrm{vbw}} + (1 - \exp^{(-V^{\mathrm{T}}/H^{\mathrm{vbw}})}) \right) \tag{2}$$

$$D^{\mathrm{D}} = D^{\mathrm{L}} \times A^{\mathrm{vbw}} \times \exp^{(-(V^{\mathrm{T}}/B^{\mathrm{w}})^{C^{\mathrm{vbw}}})}. \tag{3}$$

The variables in these equations are reported in Table 1.

4.2.1.2. Understory. Understory vegetation is the smallest component of total carbon stock and includes all live vegetation except trees larger than seedlings. In this analysis, understory carbon is a fixed fraction of live tree carbon based on published ratios reported by the U.S. EPA (2003). Weighted ratios for regions/forest types are created using forestland area data reported by the USDA Forest Service (Miles, 2003).

$$C^{\mathrm{U}} = \frac{D^{\mathrm{L}}}{U^{\mathrm{B}}} \times 0.5 \times R^{\mathrm{Uw}} \tag{3}$$

The variables in this equation are defined in Table 2. The weighted parameters used are reported in Table 3.

4.2.1.3. Forest floor and coarse woody debris. Forest floor carbon constitutes the third largest carbon storage pool, but this pool is much smaller than tree or soil carbon pools. Smith and Heath (2002) developed a model for estimating forest floor carbon mass, which forms the basis for the forest floor carbon estimates used here. Their model's definition of forest floor excludes coarse woody debris (CWD) materials (i.e., pieces of dead wood that are not attached to trees). CWD includes large woody material fallen or cut and left from live and standing
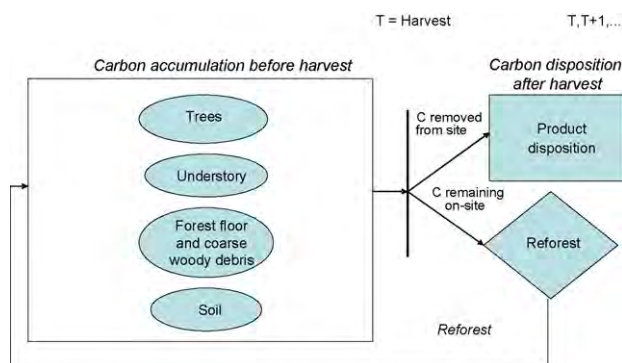
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[4] The parameters used are weighted for the economic model's (McCarl et al., 2005) region/forest-type designations. Forestland area data reported in the RPA Assessment (Miles, 2003) are used to calculate the appropriate weights.

Table 1
Tree carbon variables and parameters

Symbol	Description	Source
D^L (Mg C/ha)	Live tree mass density (above and below ground)	See Eq. (2)
D^D (Mg C/ha)	Dead tree mass density (above and below ground)	See Eq. (3)
C^R (Mg C/acre)	Total tree carbon	See Eq. (1)
V^T (m³/ha)	Total timber volume	–
F^{vbw}, G^{vbw}, H^{vbw}	Weighted live tree density parameters from volume-to-biomass equations	Smith et al. (2003) Table 3 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
A^{vbw}, B^{vbw}, C^{vbw}	Weighted dead tree mass density parameters from volume-to-biomass equations	Smith et al. (2003) Table 4 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
U^B (1 hectare (ha) = 2.471 acres)	Units conversion factor	–

Mg C = megagram ("metric" tonne) of carbon equivalent m³ = cubic meters of timber volume.

dead trees with a diameter of at least 7.5 cm (W.B. Smith et al., 2004; J. Smith et al., 2004). CWD accumulates over the life of a forested stand. At the time of harvest, a relatively large component of CWD may be left on site, which decays over time as the next rotation of trees grows. To account for effects of growth, mortality, disturbance, and decay of carbon in this material, we assumed CWD is a fixed fraction of tree carbon. Published ratios of CWD carbon to live tree carbon reported by the U.S. EPA (2003) were weighted for regions/forest types using forestland area data reported by the USDA Forest Service (Miles, 2003). This formulation of the CWD model clearly has limitations because CWD dynamics depend on the time since harvest and the amount of dead wood left after the disturbance. Although we view the results of the simulations using the current CWD model as fairly robust, given the relatively small factor that CWD plays in stand dynamics over time, the CWD model likely underestimates CWD stocks. Future CWD modeling work could adopt methods similar to recently published work (Smith et al., 2006).

The model for net accumulation of forest floor carbon is a continuous and increasing function of age. The rate of accumulation eventually approaches zero (i.e., a steady-state level of forest carbon):

$$C^{FFA} = \left(\frac{A^{ffw} \times \text{age}}{B^{ffw} + \text{age}} \right) / U^B \tag{4}$$

The variables in this equation are defined in Table 4.

Forest floor carbon mass following clear-cutting is assumed to begin at the level of carbon for a mature forest, and decay is described using an exponential function of time and average mature forest floor carbon mass:

$$C^{FFR} = (C^{ffw} \times \exp^{-(\text{age}/D^{ffw})}) / U^B \tag{5}$$

The variables in this equation are defined in Table 5.

For CWD, we report the weighted parameters used in Table 6.

4.2.1.4. Soil. Although the soil carbon pool is the second largest carbon storage pool in aggregate in the United States (Birdsey and Heath, 1995), Heath et al. (2002) note that little change in soil carbon occurs if forests are regenerated after harvest. This analysis assumed that all public timberland harvested returns to forest after harvest (i.e., no land is deforested), as is consistent with a mandate to manage and protect public forests. As a result, we assumed soil carbon on public timberland remains at a steady-state value (i.e., there is no change in soil carbon stock in the analysis) for the entire period of analysis.

4.2.2. Carbon disposition after harvest

At the time of harvest, some timber is removed from the site and used to make pulpwood-based products such as paper and sawlog-based products such as lumber, veneer, and

Table 2
Understory carbon variables and parameters

Symbol	Description	Source
C^U (Mg/acre)	Total understory carbon	See Eq. (3)
D^L (Mg/ha)	Live tree mass density (above and below ground)	See Eq. (2)
U^B (1 hectare (ha) = 2.471 acres)	Units conversion factor	–
R^{Uw} (%)	Weighted ratio of understory carbon to live tree carbon	U.S. EPA (2003) Table O-2 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6

Table 3
Weighted ratio of understory to live tree carbon (%)

Region	Softwood	Hardwood	Planted pine	Natural pine	Oak pine	Douglas fir	Bottomland hardwood	Upland hardwood	Other softwoods
Northeast	2.6	2.2	NA	NA	NA	NA	NA	NA	NA
Lake states	2.1	2.4	NA	NA	NA	NA	NA	NA	NA
Corn Belt	2.1	2.4	NA	NA	NA	NA	NA	NA	NA
Southeast	NA	NA	6.8	6.8	4.4	NA	2.2	4.4	NA
South central	NA	NA	5.9	5.9	4.4	NA	2.2	3.7	NA
Rocky mountain	5.7	9.2	NA	NA	NA	NA	NA	NA	NA
Pacific northwest west	2.0	4.5	NA	NA	NA	2.0	NA	NA	3.2
Pacific northwest east	3.0	4.5	NA	NA	NA	NA	NA	NA	NA
Pacific southwest	5.0	2.9	NA	NA	NA	NA	NA	NA	NA

Source: Author calculations using U.S. EPA (2003) and forestland area data from RPA (Miles, 2003).

Table 4
Forest floor carbon variables and parameters: net accumulation

Symbol	Description	Source
C^{FFA} (Mg/acre)	Total forest floor carbon net accumulation	See Eq. (4)
Age (years)	Age of stand	–
A^{ffw}, B^{ffw}	Weighted forest floor carbon model coefficients	Smith and Heath (2002) Table 4 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
U^{B} (1 hectare (ha) = 2.471 acres)	Units conversion factor	–

Table 5
Forest floor carbon variables and parameters: decay of forest floor carbon mass existing prior to clear-cut

Symbol	Description	Source
C^{FFR} (Mg/acre)	Total forest floor carbon, residual	See Eq. (5)
Age (years)	Age of stand	–
C^{ffw}, D^{ffw}	Weighted forest floor carbon mass coefficients	Smith and Heath (2002) Table 4 weighted by forestland area data from RPA (Miles, 2003) Tables 5 and 6
U^{B} (1 hectare (ha) = 2.471 acres)	Units conversion factor	

panels. These products are then used to produce goods and services such as furniture, housing, and printed materials that are put into use for some period of time. The ultimate disposition over time of harvested carbon removed from the site depends on the products produced, their end uses, and the period of time elapsed since they were harvested and turned into product. Carbon in logging residue left on site is tracked separately in the forest floor carbon pool described above.

The wood product carbon accounting method used here is based on early versions of recent product accounting work (Smith et al., 2006). The modified approach uses calculation methods that are distinguished by the starting point of the harvest input (e.g., roundwood harvests or primary products produced). Because future NF and OPUB timberland inventories and timber harvest levels are expressed in terms of roundwood harvested rather than primary products produced,

we used the roundwood harvests approach to track the fate of product carbon in the following pools:[5]
- products in use (sink),
- landfills (sink),
- energy (source or sink), and
- emissions (source).

Note, our primary analysis treats wood products allocated to the energy pool as a source of GHG emissions. However, we have also included calculations that treat energy uses as a sink

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[5] In contrast, the FASOMGHG economic model (McCarl et al., 2005), which incorporates the carbon accounting methods described herein and applies them to estimate forest carbon sequestration at the national and regional levels in the United States, includes production technologies that convert roundwood harvests into primary products. Therefore, FASOMGHG's product accounting system uses the alternative starting point for product carbon calculations (i.e., quantities of primary products produced).

Table 6
Weighted ratio of coarse woody debris (CWD) to live tree carbon (%)

Region	Softwood	Hardwood	Planted pine	Natural pine	Oak pine	Douglas fir	Bottomland hardwood	Upland hardwood	Other softwoods
Northeast	12.3	11.2	NA	NA	NA	NA	NA	NA	NA
Lake states	14.1	10.8	NA	NA	NA	NA	NA	NA	NA
Corn belt	14.1	10.8	NA	NA	NA	NA	NA	NA	NA
Southeast	NA	NA	23.9	23.9	17.3	NA	21.8	24.3	NA
South central	NA	NA	18.6	18.6	17.3	NA	15.7	15	NA
Rocky mountain	12.6	26.7	NA	NA	NA	NA	NA	NA	NA
Pacific northwest west	11.9	3.9	NA	NA	NA	11.9	NA	NA	15.4
Pacific northwest east	14.8	3.9	NA	NA	NA	NA	NA	NA	NA
Pacific southwest	13.0	11.4	NA	NA	NA	NA	NA	NA	NA

Source: Author calculations using U.S. EPA (2003) and forestland area data from RPA (Miles, 2003).

for GHG emissions, assuming that biomass energy sources from the forest sector substitute for fossil fuel energy sources and serve as an offset for those emissions.

To calculate product carbon, we used cubic feet of roundwood harvested, divided into pulpwood or sawtimber products using yield tables, and converted volumes harvested into metric tonnes of carbon using factors reported in earlier versions of Smith et al. (2006). These factors include the average specific gravity, an upward adjustment to account for bark (1.18), and the carbon content of wood (0.5). Next, we allocated the carbon into the wood product pools (see Fig. 5) according to years since harvest and the disposition patterns. Examples of these patterns for the Southeast region are reported in Table 7.[6]

5. Results

Carbon sequestrations for U.S. public timberlands under the three scenarios (BAU, no harvest, and high harvest/pre-1989) are presented in Tables 8–10 respectively. Results are reported separately for all public timberlands and subcomponents (NF and OPUB) and for forest ecosystem carbon and wood product carbon. The projection time period is 10 decades, starting in 2010 and running through 2110. Tables 8–10 report detail for the first 5 decades, but summary totals are provided below for all 10 decades in the projection (Figs. 6–8). All carbon quantities are reported in average annual change in carbon stocks for that period, also known as annual flux.

Under the BAU scenario, public timberlands sequester, on average, 50 MMTC annually during the first 5 decades. This estimate ranges from 65 to 40 MMTC between 2010 and 2050, and decline after that (Fig. 6). The annual carbon flux occurs primarily in the ecosystem carbon pools of public forests prior to harvest (NF and OPUB), and the remainder is associated with postharvest wood and paper product sequestration. The ecosystem fluxes range between 82 and 92% of the total flux depending on decade and whether energy is treated as a credit. NFs account for

over 60% of the annual carbon flux for all public timberlands. In 2030, for example, we estimated a total annual forest carbon flux of 33 million metric tonnes for NF timberlands compared with 15 million metric tonnes for OPUB timberlands (see Table 8). As shown in Fig. 7, over 85% of the NF forest carbon flux occurs in the West. The Rocky Mountain region accounts for 41%, followed by the Pacific northwest west (23%) and Pacific southwest (21%).

Table 8 and Fig. 6 display a positive but declining sequestration rate for public timberlands under BAU, with sequestration levels highest in the first decade and falling after that. The magnitudes of stock changes are consistent with the estimates for public forests in Smith and Heath (2004), although they do exhibit slightly different trends. These patterns reflect recent dynamics in the way public lands have been managed. Many of the current forest stands on public timberland today were regenerated after the heavier timber harvest periods of the 1960s–1980s. The net growth in such forest stands eventually slows down considerably as the stands age. Together with the recent slowdown in timber harvest levels, the age distribution of the public timberland stands will
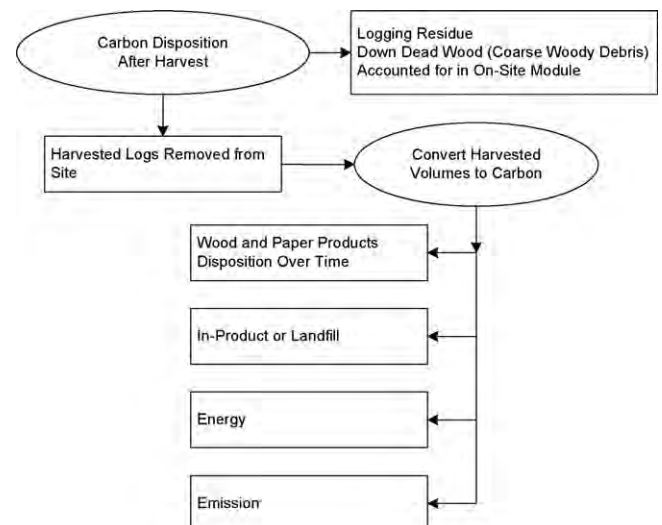

Fig. 5. Wood and paper product carbon disposition.

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[6] Data for other regions are available upon request.

Table 7
Example of disposition patterns of harvested wood by region and harvest type, 100-Year period: southeast[a]

Region	Type	Product	Disposition	Years after harvest										
				0	10	20	30	40	50	60	70	80	90	100
Southeast	Softwood	Pulpwood	Products	0.30	0.07	0.05	0.04	0.03	0.03	0.03	0.03	0.03	0.03	0.02
Southeast	Softwood	Pulpwood	Landfills	0.00	0.16	0.16	0.16	0.10	0.14	0.14	0.13	0.12	0.11	0.11
Southeast	Softwood	Pulpwood	Energy	0.44	0.45	0.45	0.46	0.46	0.46	0.46	0.46	0.46	0.46	0.46
Southeast	Softwood	Pulpwood	Emissions	0.26	0.32	0.34	0.35	0.41	0.37	0.38	0.39	0.40	0.41	0.41
Southeast	Softwood	Sawtimber	Products	0.47	0.28	0.24	0.21	0.18	0.17	0.15	0.14	0.13	0.13	0.12
Southeast	Softwood	Sawtimber	Landfills	0.00	0.13	0.16	0.17	0.18	0.19	0.19	0.19	0.18	0.18	0.18
Southeast	Softwood	Sawtimber	Energy	0.38	0.40	0.40	0.40	0.40	0.40	0.41	0.41	0.41	0.41	0.41
Southeast	Softwood	Sawtimber	Emissions	0.15	0.19	0.20	0.22	0.24	0.24	0.25	0.26	0.28	0.28	0.29
Southeast	Hardwood	Pulpwood	Products	0.30	0.07	0.05	0.04	0.04	0.03	0.03	0.03	0.03	0.03	0.03
Southeast	Hardwood	Pulpwood	Landfills	0.00	0.16	0.16	0.15	0.15	0.14	0.13	0.12	0.12	0.11	0.10
Southeast	Hardwood	Pulpwood	Energy	0.39	0.40	0.41	0.41	0.41	0.41	0.41	0.41	0.41	0.41	0.41
Southeast	Hardwood	Pulpwood	Emissions	0.31	0.37	0.38	0.40	0.40	0.42	0.43	0.44	0.44	0.45	0.46
Southeast	Hardwood	Sawtimber	Products	0.27	0.12	0.08	0.07	0.06	0.05	0.05	0.04	0.04	0.04	0.04
Southeast	Hardwood	Sawtimber	Landfills	0.00	0.11	0.13	0.14	0.14	0.14	0.13	0.13	0.13	0.13	0.12
Southeast	Hardwood	Sawtimber	Energy	0.42	0.43	0.43	0.44	0.44	0.44	0.44	0.44	0.44	0.44	0.44
Southeast	Hardwood	Sawtimber	Emissions	0.31	0.34	0.36	0.35	0.36	0.37	0.38	0.39	0.39	0.39	0.40

[a] These are proportions of the harvested stock allocated to each pool in the years following harvest. Column totals may not sum to one due to independent rounding.

Table 8
Annual stock changes: business-as-usual scenario (MM metric tonnes of carbon, MMTC, unless otherwise specified)

Decade	Forest carbon			Disposition of wood product carbon					Total in wood products		Total carbon stock change	
	Existing	Regenerated	Total in forest	Cumulative harvest volume since 2000 (MM cf)[a]	Decade harvest (MM cf)	Products	Land-fills	Energy	Without energy credit	With energy credit	Without energy credit	With energy credit
All public lands												
2010	55.1	4.5	59.7	28,009	14,695	2.5	2.5	2.5	5.0	7.5	64.6	67.1
2020	45.2	10.1	55.3	44,159	16,150	2.5	2.5	2.5	5.0	7.5	60.3	62.8
2030	34.0	13.5	47.5	60,477	16,318	2.2	2.2	2.2	4.3	6.5	51.9	54.0
2040	20.1	17.5	37.5	77,236	16,759	2.1	2.1	2.1	4.1	6.2	41.7	43.7
2050	15.7	20.4	36.2	94,239	17,003	1.9	1.9	1.9	3.8	5.6	39.9	41.8
National forests												
2010	50.0	1.2	51.2	9,424	5,394	1.2	1.2	1.2	2.5	3.7	53.7	54.9
2020	35.3	2.9	38.2	15,912	6,488	1.4	1.4	1.4	2.7	4.1	40.9	42.3
2030	28.6	4.1	32.8	22,862	6,950	1.3	1.3	1.3	2.6	3.9	35.4	36.6
2040	22.1	5.6	27.6	30,253	7,391	1.3	1.3	1.3	2.5	3.8	30.2	31.4
2050	17.6	7.1	24.7	37,888	7,635	1.2	1.2	1.2	2.4	3.5	27.0	28.2
Other public lands												
2010	5.2	3.3	8.4	18,585	9,301	1.2	1.2	1.2	2.5	3.7	10.9	12.2
2020	9.9	7.2	17.2	28,247	9,662	1.1	1.1	1.1	2.3	3.4	19.4	20.6
2030	5.4	9.4	14.8	37,615	9,368	0.9	0.9	0.9	1.7	2.6	16.5	17.4
2040	−2.0	11.9	9.9	46,983	9,368	0.8	0.8	0.8	1.6	2.4	11.5	12.3
2050	−1.9	13.4	11.5	56,351	9,368	0.7	0.7	0.7	1.4	2.1	12.9	13.6

[a] The cumulative harvest for periods includes all harvests for the previous decades plus the current decade.

shift to older stands in the coming decades and the growth rate will slow.[7]

A comparison of timber harvest scenarios illustrates the carbon storage trade-offs that policy makers face when consi-

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[7] One possible change to this growth projection is the effect of a changing climate. As shown in various studies at different spatial scales (Sohngen and Mendelsohn, 1998; Alig et al., 2002; Abt and Murray, 2001), future changes in climate can affect the growth and species distribution of forests in ways that are either favorable or unfavorable, depending on location.

dering alternative timber harvest levels from public forests. As shown in Fig. 8, moving from the baseline to a no-harvest regime leads to a significant increase in the carbon sequestered on public timberlands. Our estimates suggest an annual increase (above baseline) of 17–29 MMTC per year between 2010 and 2050, approximately a 40–50% increase in carbon storage depending on the decade. Interestingly, this is just below the 55–57% additional carbon sequestration reported by Harmon et al. (1990) when looking at the carbon sequestration potential of maintaining old-growth stands versus converting to

Table 9
Annual stock changes: no-harvest scenario (MM metric tonnes of carbon, MMTC, unless otherwise specified)

Decade	Forest carbon			Disposition of wood product carbon					Total in wood products		Total carbon stock change	
	Existing	Regenerated	Total in forest	Cumulative harvest volume since 2000 (MM cf)	Decade harvest (MM cf)	Products	Land-fills	Energy	Without energy credit	With energy credit	Without energy credit	With energy credit
All public lands												
2010	93.3	0.0	93.3	0	0	0.0	0.0	0.0	0.0	0.0	93.3	93.3
2020	85.5	0.0	85.5	0	0	0.0	0.0	0.0	0.0	0.0	85.5	85.5
2030	76.1	0.0	76.1	0	0	0.0	0.0	0.0	0.0	0.0	76.1	76.1
2040	61.0	0.0	61.0	0	0	0.0	0.0	0.0	0.0	0.0	61.0	61.0
2050	57.3	0.0	57.3	0	0	0.0	0.0	0.0	0.0	0.0	57.3	57.3
National forests												
2010	64.3	0.0	64.3	0	0	0.0	0.0	0.0	0.0	0.0	64.3	64.3
2020	52.2	0.0	52.2	0	0	0.0	0.0	0.0	0.0	0.0	52.2	52.2
2030	46.8	0.0	46.8	0	0	0.0	0.0	0.0	0.0	0.0	46.8	46.8
2040	41.1	0.0	41.1	0	0	0.0	0.0	0.0	0.0	0.0	41.1	41.1
2050	36.9	0.0	36.9	0	0	0.0	0.0	0.0	0.0	0.0	36.9	36.9
Other public lands												
2010	29.0	0.0	29.0	0	0	0.0	0.0	0.0	0.0	0.0	29.0	29.0
2020	33.3	0.0	33.3	0	0	0.0	0.0	0.0	0.0	0.0	33.3	33.3
2030	29.4	0.0	29.4	0	0	0.0	0.0	0.0	0.0	0.0	29.4	29.4
2040	19.9	0.0	19.9	0	0	0.0	0.0	0.0	0.0	0.0	19.9	19.9
2050	20.4	0.0	20.4	0	0	0.0	0.0	0.0	0.0	0.0	20.4	20.4

sustained harvesting of stands under rotational forestry. Sequestration under the no-harvest scenario in the first 5 decades would offset between 1 and 2% of total CO_2 emissions in the United States at current levels and is equivalent to removing the emissions of about 13–24 million cars per year. Most of the additional sequestration occurs within NFs (see Table 9). This rate of additional carbon sequestration declines over time (Fig. 8).

In contrast with the no-harvest scenario, increasing the baseline harvest levels to pre-1989 levels leads to a significant decrease in the carbon sequestered in public forests. Our estimates suggest losses ranging from 27 to 35 MMTC per year

Table 10
Annual stock changes: pre-1989 harvest levels (MM metric tonnes of carbon, MMTC, unless otherwise specified)

Decade	Forest carbon			Disposition of wood product carbon					Total in wood products		Total carbon stock change	
	Existing	Regenerated	Total in forest	Cumulative harvest volume since 2000 (MM cf)[a]	Decade harvest (MM cf)	Products	Land-fills	Energy	Without energy credit	With energy credit	Without energy credit	With energy credit
All public lands												
2010	5.6	9.1	14.8	69,470	35,630	7.5	7.5	7.5	14.9	22.4	29.7	37.1
2020	−3.1	17.7	14.6	105,975	36,504	6.6	6.6	6.6	13.2	19.8	27.8	34.3
2030	−17.7	25.5	7.8	143,301	37,327	6.0	6.0	6.0	12.0	18.0	19.9	25.9
2040	−29.9	32.2	2.3	181,085	37,784	5.3	5.3	5.3	10.5	15.8	12.9	18.1
2050	−33.1	36.1	3.0	220,015	38,929	5.0	5.0	5.0	10.0	15.0	13.1	18.1
National forests												
2010	2.7	5.8	8.6	47,200	24,220	5.6	5.6	5.6	11.1	16.7	19.7	25.3
2020	−11.0	10.1	−0.8	72,040	24,840	5.0	5.0	5.0	10.1	15.1	9.2	14.3
2030	−19.2	15.4	−3.8	97,562	25,522	4.6	4.6	4.6	9.2	13.8	5.4	10.0
2040	−24.3	19.0	−5.3	123,472	25,910	4.1	4.1	4.1	8.1	12.2	2.9	6.9
2050	−27.3	21.3	−6.0	150,527	27,055	4.0	4.0	4.0	8.0	12.0	1.9	5.9
Other public lands												
2010	2.9	3.3	6.2	22,270	11,410	1.9	1.9	1.9	3.8	5.7	10.0	11.9
2020	7.8	7.6	15.4	33,935	11,664	1.6	1.6	1.6	3.1	4.7	18.5	20.1
2030	1.6	10.1	11.7	45,739	11,804	1.4	1.4	1.4	2.8	4.2	14.5	15.9
2040	−5.6	13.2	7.6	57,613	11,874	1.2	1.2	1.2	2.4	3.6	10.0	11.2
2050	−5.7	14.8	9.1	69,487	11,874	1.0	1.0	1.0	2.0	3.0	11.1	12.1

[a] The cumulative harvest for periods includes all harvest for the previous decades plus the current decade.
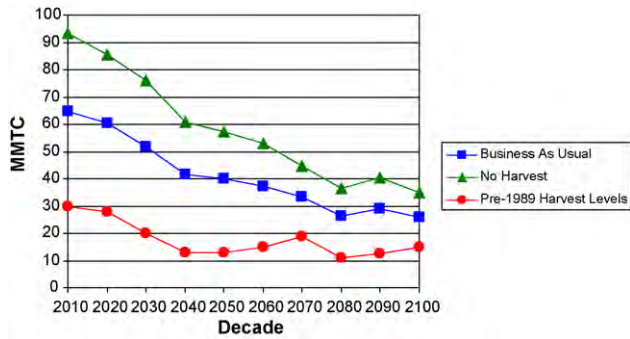
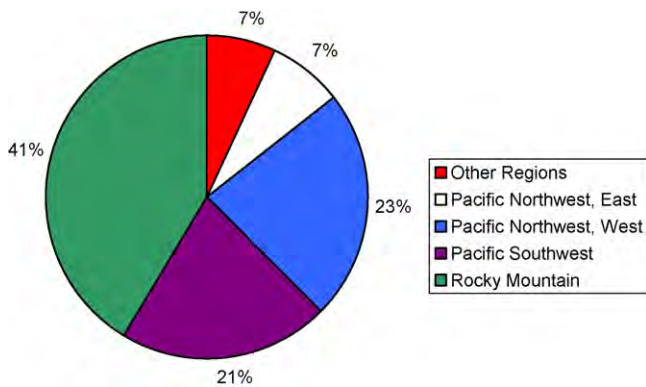Fig. 6. Annual carbon sequestration in all public lands by scenario.

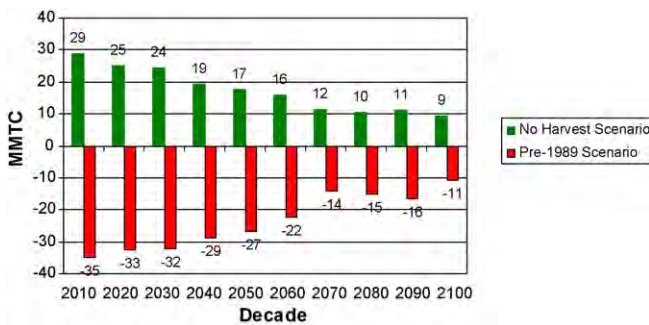Fig. 7. Distribution of annual NF carbon stock changes by region: BAU scenario (2030).

Fig. 8. Comparison of annual carbon stock changes with business-as-usual scenario.

between 2010 and 2050, approximately a 50–80% decline in carbon storage from BAU depending on the decade. This tempers some in the last 5 decades as the regrowth from harvested stands contribute more strongly to the sequestration rate. The vast majority of timber harvests come from NF timberlands: an average of 26 billion ft^3 are harvested per decade in NFs compared with 12 billion ft^3 in OPUB timberlands. As a result, returning to these high timber harvest levels would make NFs a net source of emissions between 2020 and 2050. Although OPUB timberlands continue to be carbon sinks, the annual carbon stock changes in forests are substantially lower than in the BAU case. Carbon losses associated with the more intense harvesting scenario are reduced to some degree

through carbon storage in wood and paper products. Our estimates suggest that wood and paper products sequester between 10 and 15 MMTC per year. If we treat energy uses as a sink for GHG emissions, assuming energy use substitutes for other energy sources and serves as an offset for those emissions, our wood and paper product sequestration estimates increase another 5–7 MMTC per year, rising between 15 and 22 MMTC per year total, depending on the decade (see Table 10).

It is instructive to view these results in terms of the potential monetary value of sequestered carbon in the different scenarios. Payments for carbon sequestration can be viewed as part of a potential broader system to offset emissions of CO_2 and other GHGs. CO_2 emission credits are currently being traded for between US$ 15 and 30 per metric tonne (Mg) of CO_2 equivalent on the EU's Emissions Trading System (EU ETS). Translating to units of carbon, this is about US$ 55–110 per Mg C. Although forest carbon sequestration is not currently traded in the EU ETS, this range provides some sense, perhaps an upper range, of its monetary potential if sequestered carbon on public timberlands were included in a trading mechanism.[8] At this price range, the annual value of carbon sequestered on public timberlands under BAU ranges from US$ 2.2 to 7.1 billion, depending on the decade. However, GHG compensation schemes that include forest carbon offsets might not consider BAU sequestration to be creditable, focusing instead on carbon that is additional to BAU (Murray et al., 2007). We can estimate that the additional amount of carbon sequestered under the no-harvest scenario would be between US$ 0.9 and 3.2 billion per year, and foregone carbon revenue would be between US$ 1.5 and 3.9 billion per year under the pre-1989 harvest scenario. By contrast, timber harvest revenues on public lands in 2005 were approximately US$ 800–900 million (Adams, 2006). One should note that these revenue comparisons do not capture all relevant aspects of welfare. A more complete comparison would capture effects on consumer and producer surplus and thereby the net benefits to society of each harvesting plan. That is beyond the scope of this study. The revenue comparisons here, however, do indicate relative trade-offs between timber and carbon revenue that might be expected under different management regimes.

6. Conclusions

For decades, public timberlands have been managed for multiple uses and ecosystem services including timber, range, wildlife habitat, watershed protection, recreation, and visual amenities. More attention in recent years has been placed on establishing and maintaining forest carbon sinks to help regulate atmospheric GHGs and climate, but little empirical work at a national scale has estimated the biophysical potential

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[8] Rather than evaluating its revenue potential in a greenhouse gas trading market, another perspective is the social cost of carbon remaining in the atmosphere. This measures the value of climate change damages caused by carbon accumulation in the atmosphere and thus the marginal benefit of carbon removed from the atmosphere. The most recent IPCC assessment report provides a range of values for social cost of carbon at about US$ 43 per tonne C or about US$ 12 per tonne CO_2 (IPCC, 2007).

of modifications in public timberland management to sequester more carbon. This paper addresses that gap by combining data on public timber inventories, timber harvest scenarios, and carbon accounting to quantify the accumulation of carbon on public timberlands and in wood product stocks from harvested timber under three scenarios: BAU, no harvest, and high harvest (equivalent to the 1980s). Findings suggest that under BAU, public timberlands will continue to sequester carbon through the next century, though at a diminishing rate. The BAU accumulation of carbon occurs because of the age class and growth dynamics of the current inventory of public timberland, which has experienced timber harvest levels in the recent past that are substantially lower than the preceding decades. These changes in timber harvest were done for a wide variety of ecological and economic reasons, but a by-product of these efforts was an increase in public timberlands' positive contribution to global climate regulation.

Variations in BAU in either direction – elimination of harvests altogether or a substantial ramp-up in public harvests to levels of 20 years ago – could substantially alter the annual carbon balance of public timberlands, at least 50% in either direction. Each action would have opportunity costs in terms of the economic and ecological value of the corresponding changes in market and nonmarket ecosystem services, but a market for sequestered carbon could alter the balance considerably with public sequestration worth potentially billions of dollars in value per year. Although markets for carbon are in their nascent stages and the level of future carbon prices are highly uncertain, public decision makers should nonetheless consider the economic value of carbon when developing national, regional, and forest-level targets for timber harvests and other public timberland outputs.

This study provides a rough estimate of the potential from a relatively few, though wide-ranging, timber harvest policy alternatives. Forest and carbon management, however, is much more subtle than simply determining how much to harvest. Many forest management decisions from the time of stand establishment through mid-rotation treatments to the timber harvest decision could be affected with carbon sequestration as a more accentuated objective. Of particular interest is the link between carbon management, fire management, and biofuel production, each of which can have a profound impact on the carbon balance, ecological integrity, and economic value of the forest. One research need is a better understanding of how such linkages are affected by the stochastic nature of certain disturbances such as fires. Future research should carefully evaluate these trade-offs and opportunities at regional, landscape, and individual forest scales.

Acknowledgements

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Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests

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From analysis of published global site biomass data (*n* = 136) from primary forests, we discovered (*i*) the world's highest known total biomass carbon density (living plus dead) of 1,867 tonnes carbon per ha (average value from 13 sites) occurs in Australian temperate moist *Eucalyptus regnans* forests, and (*ii*) average values of the global site biomass data were higher for sampled temperate moist forests (*n* = 44) than for sampled tropical (*n* = 36) and boreal (*n* = 52) forests (*n* is number of sites per forest biome). Spatially averaged Intergovernmental Panel on Climate Change biome default values are lower than our average site values for temperate moist forests, because the temperate biome contains a diversity of forest ecosystem types that support a range of mature carbon stocks or have a long land-use history with reduced carbon stocks. We describe a framework for identifying forests important for carbon storage based on the factors that account for high biomass carbon densities, including (*i*) relatively cool temperatures and moderately high precipitation producing rates of fast growth but slow decomposition, and (*ii*) older forests that are often multiaged and multilayered and have experienced minimal human disturbance. Our results are relevant to negotiations under the United Nations Framework Convention on Climate Change regarding forest conservation, management, and restoration. Conserving forests with large stocks of biomass from deforestation and degradation avoids significant carbon emissions to the atmosphere, irrespective of the source country, and should be among allowable mitigation activities. Similarly, management that allows restoration of a forest's carbon sequestration potential also should be recognized.

Eucalyptus regnans | climate mitigation | primary forest | deforestation and degradation | temperate moist forest biome

Deforestation currently accounts for ≈18% of global carbon emissions and is the third largest source of emissions (1). Reducing emissions from deforestation and degradation (REDD) is now recognized as a critical component of climate change mitigation (2). A good understanding of the carbon dynamics of forests (3) is therefore important, particularly about how carbon stocks vary in relation to environmental conditions and human land-use activities. Average values of biomass carbon densities for the major forest biomes (4) are used as inputs to climate-carbon models, estimating regional and national carbon accounts, and informing policy debates (5). However, for many purposes it is important to know the spatial distribution of biomass carbon within biomes (6) and the effects of human land-use activities on forest condition and resulting carbon stocks (refs. 3 and 7 and www.fao.org/forestry/site/10368/en).

Primarily because of Kyoto Protocol rules (ref. 8; http://unfccc.int/resource/docs/convkp/kpeng.pdf), interest in carbon accounting has been focused on modified natural forests and plantation forests. It has been argued that primary forests, especially very old forests, are unimportant in addressing the climate change problem because (*i*) their carbon exchange is at equilibrium (9, 10), (*ii*) carbon offset investments focus on planting young trees as their rapid growth provides a higher sink capacity than old trees, and/or (*iii*) coverage and hence importance of modified forest is increasing. Recent research findings have countered the first argument for all 3 major forest biomes (namely, tropical, temperate, and boreal forests) and demonstrated that old-growth forests are likely to be

functioning as carbon sinks (11–13). The long time it takes new plantings to sequester and store the amount of carbon equivalent to that stored in mature forests counters the second argument (14). The third argument about the unimportance of old forest in addressing climate change relates, in part, to the diminishing extent of primary forest caused by land-use activities (15) and associated depletion of biomass carbon stocks (16). However, significant areas of primary forest remain (17), and depleted carbon stocks in modified forests can be restored.

It is useful to distinguish between the carbon carrying capacity of a forest ecosystem and its current carbon stock. Carbon carrying capacity is the mass of carbon able to be stored in a forest ecosystem under prevailing environmental conditions and natural disturbance regimes, but excluding anthropogenic disturbance (18). It is a landscape-wide metric that provides a baseline against which current carbon stocks (that include anthropogenic disturbance) can be compared. The difference between carbon carrying capacity and current carbon stock allows an estimate of the carbon sequestration potential of an ecosystem and quantifies the amount of carbon lost as a result of past land-use activities.

This study re-evaluates the biomass carbon densities of the world's major forest biomes based on a global synthesis of site data of biomass measurements in forest plots from publicly available peer-reviewed articles and other reputable publications. Site data were selected that (*i*) provided appropriate measurements of biomass and (*ii*) sampled largely mature and older forests to provide an estimate of carbon carrying capacity. The most reliable nondestructive source of biomass carbon data are from field measurements of tree and dead biomass structure at sites that sample a given forest type and condition. These structural measurements are converted to biomass carbon densities by using allometric equations. Standard national forestry inventories contain site data but they are not always publicly available and their suitability for estimating carbon stocks at national and biome-levels has been questioned (5, 6).

We identify those forests with the highest biomass carbon densities and consider the underlying environmental conditions and ecosystem functions that result in high carbon accumulation. These results (*i*) provide a predictive framework for identifying forests with high biomass carbon stocks, (*ii*) help clarify interpretation of average forest biome values such as those published by the Intergovernmental Panel on Climate Change (IPCC), and (*iii*) inform policies about the role of forests in climate change mitigation.

Australian *Eucalyptus regnans* Forests Have the World's Highest Biomass Carbon Density

Evergreen temperate forest dominated by *E. regnans* (F. Muell.) (Mountain Ash) in the moist temperate region of the Central

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EXHIBIT 13

Fig. 1. *E. regnans* forest with midstory of *Acacia* and understory of tree ferns. The person in the bottom left corner provides a scale.

Highlands of Victoria, southeastern Australia has the highest known biomass carbon density in the world. We found that *E. regnans* forest in the O'Shannassy Catchment of the Central Highlands (53 sites within a 13,000-ha catchment) contains an average of 1,053 tonnes carbon (tC)·ha^{-1} in living above-ground biomass and 1,867 tC·ha^{-1} in living plus dead total biomass in stands with cohorts of trees >100 years old sampled at 13 sites. We examined this catchment in detail because it had been subject to minimal human disturbance, either by Indigenous people or from post-European settlement land use. We compared the biomass carbon density of the *E. regnans* forest with other forest sites globally by using the collated site data (Table S1). No other records of forests have values as high as those we found for *E. regnans*.

Our field measurements and calculations revealed that maximum biomass carbon density for a *E. regnans*-dominated site was 1,819 tC·ha^{-1} in living above-ground biomass and 2,844 tC·ha^{-1} in total biomass from stands with a well-defined structure of overstory and midstory trees (see Fig. 1) consisting of multiple age cohorts with the oldest ≈250+ years (19). There was substantial spatial variability in total biomass carbon density across the sites in the catchment within an ecologically mature forest type, ranging from 262 to 2,844 tC·ha^{-1}. Unexpectedly, we found the highest values were from areas experiencing past partial stand-replacing natural disturbances.

In February 2009, extensive areas of the O'Shannassy Catchment and elsewhere in the Central Highlands of Victoria were burned in a major conflagration. We will be undertaking a major survey of the network of permanent field sites in the catchment (20) to assess changes in postfire carbon stocks. It will be important that these sites are not subject to postfire salvage logging over the coming years to prevent the extensive removal of dead biomass carbon (21).

Some Temperate Moist Forest Types Can Have Higher Biomass Carbon Density Than Both Boreal and Tropical Forests

Average values of the collated global site biomass data from largely mature or primary forests were much higher for the sampled
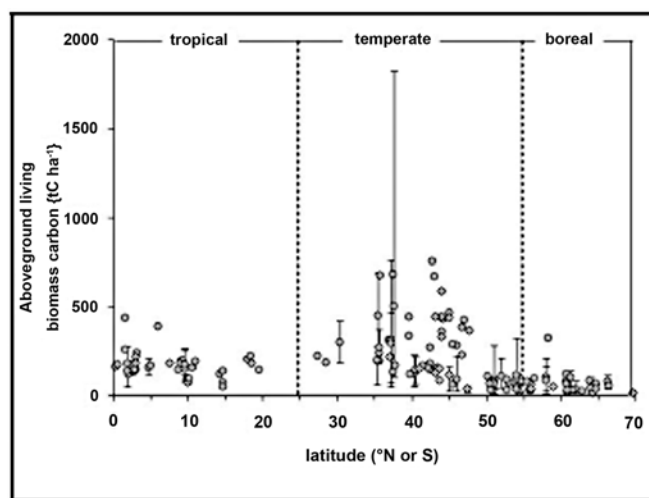

Fig. 2. Global forest site data for above-ground biomass carbon (tC·ha^{-1}) in relation to latitude (north or south). Points are values for individual or average of plots, and bars show the range in values at a site. The O'Shannassy Catchment has a mean of 501 tC·ha^{-1} and ranges from 104 to 1,819 tC·ha^{-1}. The highest biomass carbon occurs in the temperate latitudes.

temperate moist forests ($n = 44$) than they were for the sampled tropical ($n = 36$) and boreal ($n = 52$) forests, where n is the number of sites in each forest biome (Table S1) (Fig. 2). The locations of the global site biomass data are shown in Fig. S1. They do not represent all forest types or environmental conditions within a given biome (reflecting the difficulty of finding published field data) and therefore are insufficient to calculate biome spatial averages. We related site values of above-ground living biomass carbon (tC·ha^{-1}) and total biomass carbon (tC·ha^{-1}) to temperature and precipitation (Fig. 3).

Fig. 3 shows that temperate moist forests occurring where temperatures were cool and precipitation was moderately high had the highest biomass carbon stocks. Temperate forests that had particularly high biomass carbon density included those dominated by *Tsuga heterophylla*, *Picea sitchensis*, *Pseudotsuga menziesii*, and *Abies amabilis* in the Pacific Northwest of North America [range in living above-ground biomass of 224−587 tC·ha^{-1} and total biomass of 568–794 tC·ha^{-1} (22–25)]. A synthesis of site data for the Pacific Northwest gave an average for evergreen needle leaf forest of 334 tC·ha^{-1} (26), and this is used as the continental biome value by the IPCC (4). An upper limit of biomass accumulation of 500–700 tC·ha^{-1} in the Pacific Northwest of the United States has been derived from an analysis of global forest data of carbon stocks and net ecosystem productivity in relation to stand age (11, 27). In New Zealand, the highest biomass carbon density reported is for *Agathis australis* [range in living above-ground biomass of 364−672 and total biomass of 400–982 tC·ha^{-1} (28)]; and a synthesis based on forest inventory data gave a mean of 180 tC·ha^{-1} with a range in means for forest classes of 105–215 tC·ha^{-1} (29). In Chile, the highest biomass carbon densities reported are for *Nothofagus*, *Fitzroya*, *Philgerodendron*, and *Laureliopsis* [range in living above-ground biomass 142–439 and total biomass of 326–571 tC ha^{-1} (30–33)].

IPCC Tier-1 Biome Default Values

IPCC biome default values are shown in Table 1 alongside the published global site biomass data (Table S1). The site data were averaged for each biome but they are not equivalent to a spatial average for each biome. The comparison helps identify biomes where site averages differ significantly from default values. The biome-averaged values of the global site biomass carbon data were 2.5–3 times higher than the IPCC biome default values for warm and cool temperate moist forests (Table 1). The IPCC default
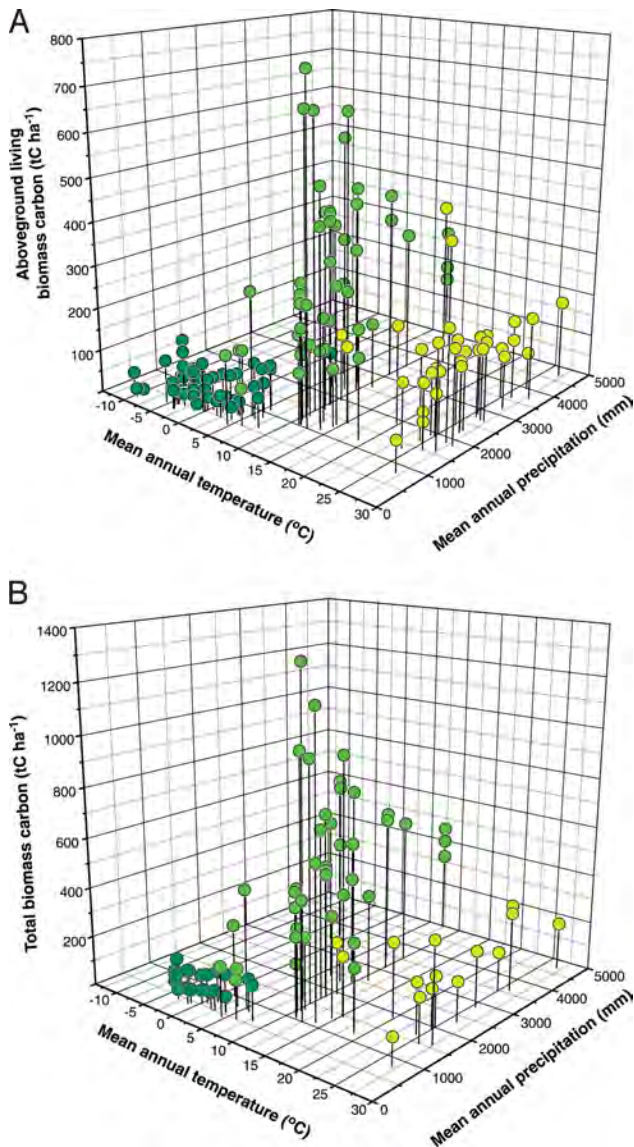
Fig. 3. Global forest site data for above-ground living biomass carbon (tC·ha⁻¹) (*A*) and total biomass carbon (tC·ha⁻¹) (*B*), in relation to mean annual temperature and mean annual precipitation for the site. Site data are shown in relation to their distribution among biomes of boreal (dark green), temperate (midgreen), and tropical (light green) forests. The highest biomass carbon density occurs in cool, moderately wet climates in temperate moist forest biomes. Some sites had values for above-ground living biomass carbon but not dead biomass, so there was no value for total biomass carbon.

values were <1 SD from the averaged site values. Average site data were comparable with IPCC default values for tropical and boreal biomes. However, the IPCC biome default value for tropical moist forest was marginally <1 SD from the averaged site values. Also, the site data for the boreal biome reflected higher above-ground living biomass carbon values but lower below-ground plus dead biomass carbon values compared with the IPCC default values (Table 1).

The differences between the collated global site biomass data and IPCC biome default values for temperate moist forests reflect the diversity of forest ecosystem types considered under the temperate biome category. Biome default values likely under-represent Southern Hemisphere evergreen temperate moist forest types and do not distinguish forest condition caused by land-use history (5). The differences between site biomass data and IPCC default values for boreal forests could reflect the effect of land-use history and fire on carbon stocks at the site level.

Toward a Predictive Framework for High Biomass Carbon Forests

We developed a framework for identifying forests with high biomass carbon stocks based on an understanding of underlying mechanisms and using the *E. regnans* forests as an example. The factors in the framework include (*i*) environmental conditions, (*ii*) life history and morphological characteristics of tree species, and (*iii*) the impacts of natural disturbance such as fire and land-use history. It is the interactions and feedbacks among these factors that influence vegetation community dynamics and ultimately lead to very high carbon densities.

Derivation of Carbon Stocks. Stock of carbon represents the net exchange of carbon fluxes in an ecosystem (net ecosystem exchange). In living biomass, the carbon stock is determined by the balance between the fluxes of carbon gain by photosynthetic assimilation by the foliage [gross ecosystem production (GEP)] and carbon loss by autotrophic respiration, which results in net primary productivity (NPP). In the total ecosystem (living plus dead biomass plus soil), the carbon stock is determined by the balance between the fluxes of carbon gain by NPP and carbon loss by decomposition of dead biomass and heterotrophic respiration. Ecosystem carbon stocks vary because environmental conditions influence the carbon fluxes of photosynthesis, decomposition, and autotrophic and heterotrophic respiration differently (34).

Environmental Conditions. The key climatic variables of precipitation, temperature, and radiation are broadly correlated with vegetation structure and function (35, 36), although such empirical correlations do not necessarily reveal underlying biochemical processes or the dependence of these processes on environmental factors (37). Climatic influences on photosynthesis include effects of (*i*) irradiance and temperature on carboxylation rates, (*ii*) temperature and soil water status on stomatal conductance and thus diffusion of CO_2 from the atmosphere into the intercellular air spaces, and (*iii*) temperature-dependent nitrogen uptake (37). The climatic conditions and relatively fertile soils of the Central Highlands of Victoria favor rapid growth of *E. regnans* (>1 m·yr⁻¹ for the first 70 years), and these trees eventually become the world's tallest flowering plant (up to 130 m) (38).

Both dark respiration and maintenance respiration are temperature dependent (37). Soil respiration is correlated with temperature and water availability, although substrate also has an important influence (34). Rates of coarse woody biomass decomposition have been found to decrease with lower temperatures in temperate forests (39) and are also related to wood density, chemistry, and size (40–42).

Climatic conditions that favor higher rates of GEP relative to rates of respiration and decomposition should, other factors being equal, lead to larger biomass carbon stocks. Table 2 gives the average and range in climatic conditions (annual precipitation and temperature) for the global site data from Table S1 and compares estimates of GEP (34) and decomposition rates (*k*) (42). Estimates of the climate conditions and derived variables are also shown for *E. regnans* forests in the Central Highlands of Victoria. Temperate forests are characterized by higher rates of GEP than boreal forests but lower decomposition rates than tropical forests. There is considerable variation evident in rates of carbon fluxes within each forest biome, along with overlap between biomes.

Life History and Morphological Characteristics of Tree Species. *E. regnans* can live for ≈450 years, with stem diameters up to 6 m (38, 43). In our analysis, the stands of *E. regnans* with high values of biomass carbon density were at least 100 years old. *E. regnans* wood density is high (450–550 g·cm⁻³) (44), so that biomass is greater for a given volume. Limited crown development in *E. regnans* (through crown shyness or reduced crown area caused by abrasion of growing tips by neighboring crowns) and the isolateral leaf form of this

Table 1. Average published site data (from Table S1) for biomass carbon (tC·ha⁻¹) of each forest biome (mean, standard deviation, and number of sites) and default biomass carbon values (IPCC; refs. 4 and 66)

Domain	Climate region	Above-ground living biomass carbon, tC·ha⁻¹		Root + dead biomass carbon, tC·ha⁻¹		Total living + dead biomass carbon, tC·ha⁻¹	
		Average site data	Biome default value*	Average site data	Biome default value†	Average site data	Biome defaul value
Tropical	Tropical wet	171 (61) $n = 18$	146	76 (72) $n = 7$	67	231 (75) $n = 7$	213
	Tropical moist	179 (96) $n = 14$	112	55 (66) $n = 5$	30	248 (100) $n = 5$	142
	Tropical dry	70 $n = 1$	73	41 $n = 1$	32	111 $n = 1$	105
	Tropical montane	127 (8) $n = 3$	71	52 (6) $n = 3$	60	167 (17) $n = 3$	112
Subtropical	Warm temperate moist	294 (149) $n = 26$	108	165 (75) n = 20	63	498 (200) $n = 20$	171
	Warm temperate dry		75		65		140
	Warm temperate montane		69		63		132
Temperate	Cool temperate moist	377 (182) $n = 18$	155	265 (162) $n = 18$	78	642 (294) $n = 18$	233
	Cool temperate dry	176 (102) $n = 3$	59	102 (77) $n = 3$	62	278 (173) $n = 3$	121
	Cool temperate montane	147 $n = 1$	61		63	153 $n = 1$	124
Boreal	Boreal moist	64 (28) $n = 28$	24	37 (16) $n = 14$	75	97 (34) $n = 14$	99
	Boreal dry	59 (36) $n = 24$	8	25 (12) $n = 9$	52	84 (39) $n = 9$	60
	Boreal montane		21		55		76

The site data represent an average and variance of point values whereas the default values represent a spatial average. The site data have been taken from mature and older forests with minimal human land use impact whereas the default values do not distinguish between natural undisturbed forest and regenerating forest nor forest age (unless <20 years). Domain and climate region classification are according to Table 4.5 and defined in Table 3A.5.2 (4).
*Default values are from the IPCC (4). Above-ground biomass from Table 4.7 (4) averaged across continents for each ecological zone. Carbon fraction in above-ground biomass [Table 4.3 (4)].
†Default values are from the IPCC (4, 66). Litter carbon stocks [Table 3.2.1 (66)]. Ratio of below- to above-ground biomass [Table 4.4 (4)]. Dead wood stocks [Table 3.2.2 (66)].

species enable high levels of light to penetrate the forest floor, allowing luxuriant understory layers to grow (45). Eucalypt foliage is evergreen and minimum winter temperatures in the Central Highlands are moderate, so *E. regnans* trees can grow all year. Similarly, evergreen temperate forests of the Pacific Northwest of North America with high biomass have been found to photosynthesize throughout the year (46).

Natural Disturbance Such as Fire. Fire affects vegetation structure and biomass carbon stocks at multiple spatial scales, such as the landscape, stand, and individual tree levels. Fire can kill but not combust all of the material in trees, leading to much of the biomass carbon changing from the living biomass pool to the standing dead and fallen dead biomass pools. The amount of carbon lost from the forest floor and the soil profile may vary depending on ecosystem type, fire regimes, and postdisturbance weather conditions (47). The dead biomass then decays as the stand grows (48). Slow decomposition rates can therefore result in large total carbon stocks

of dead biomass and regrowing living biomass. A study of temperate forests along a subalpine elevation gradient in the United States estimated coarse woody debris turnover time to be 580 ± 180 years (39). Large amounts of coarse woody debris biomass are also typical of old-growth forests of the Pacific Northwest of North America (40).

Unlike the majority of eucalypt species, *E. regnans* does not regenerate by epicormic growth or sprouting from lignotubers after a wildfire. Rather, a tree is killed if its canopy is completely scorched by fire. It then sheds seeds that germinate in the postfire ash-bed conditions (49). In the Central Highlands of Victoria, wetter sites on lower slopes and shaded aspects support longer fire intervals and less intense fires, leading to a greater probability of multiaged stands (50). Whether environmentally controlled or the result of stochastic processes, past partial stand-replacing wildfires produce younger cohorts of fast-growing *E. regnans* trees, mixed with an older cohort of living and dead trees, together with rejuvenating the understory of *Acacia* spp. and other tree species (Fig. 1).

Table 2. Comparison of mean and range climatic conditions for boreal, temperate, and tropical forest biomes based on the global site data (Table S1 and Fig. 3)

Condition	Mean annual temperature, ° C	Total annual precipitation, mm	GEP, g CO₂ m⁻² y⁻¹	k, year⁻¹
Boreal: mean	−0.6	581	822	0.01
Minimum	−10.0	213	382	0.01
Maximum	8.0	2,250	1,228	0.03
Temperate: mean	9.9	1,850	1,318	0.04
Minimum	1.5	404	923	0.02
Maximum	18.9	5,000	1,740	0.08
Tropical: mean	23.6	2,472	1,961	0.12
Minimum	7.2	800	1,190	0.03
Maximum	27.4	4,700	2,140	0.17
E. regnans: mean	11.1	1,280	1,374	0.04
Minimum	7.0	661	1,181	0.03
Maximum	14.4	1,886	1,529	0.06

Shown is the climatic profile for *E. regnans* calculated by Lindenmayer *et al.* (65). GEP is estimated from a regression correlation derived from flux tower data as a function of mean annual temperature by Law *et al.* (34). k is the decomposition rate constant of coarse woody debris calculated from an empirical relationship derived by Chambers *et al.* (42) using forest biome characteristic temperatures.

Land-Use Activity. The final reason for high biomass carbon densities in *E. regnans* forests is a prolonged absence of direct human land-use activity. The O'Shannassy Catchment has been closed to public access for >100 years to provide water for the city of Melbourne. It had an almost complete absence of Indigenous land use before European settlement. Natural disturbances have included wildfire, windstorms, and insect attacks. Logging has been excluded, including postwildfire salvage logging that removes large amounts of biomass in living and dead trees (thus preventing the development of multiple age cohorts) (21, 51, 52).

Some types of temperate moist forests that have had limited influence by human activities can be multiaged and do not necessarily consist exclusively of old trees, but often have a complex multiaged structure of multiple layers produced by regeneration from natural disturbances and individual tree gaps in the canopy (53). Net primary production in some types of multiaged old forests has been found to be 50–100% higher than that modeled for an even-aged stand (54). Both net primary production and net ecosystem production in many old forest stands have been found to be positive; they were lower than the carbon fluxes in young and mature stands, but not significantly different from them (55). Northern Hemisphere forests up to 800 years old have been found to still function as a carbon sink (11). Carbon stocks can continue to accumulate in multiaged and mixed species stands because stem respiration rates decrease with increasing tree size, and continual turnover of leaves, roots, and woody material contribute to stable components of soil organic matter (56). There is a growing body of evidence that forest ecosystems do not necessarily reach an equilibrium between assimilation and respiration, but can continue to accumulate carbon in living biomass, coarse woody debris, and soils, and therefore may act as net carbon sinks for long periods (12, 57–59). Hence, process-based models of forest growth and carbon cycling based on an assumption that stands are even-aged and carbon exchange reaches an equilibrium may underestimate productivity and carbon accumulation in some forest types.

Large carbon stocks can develop in a particular forest as a result of a combination and interaction of environmental conditions, life history attributes, morphological characteristics of tree species, disturbance regimes, and land-use history. Very large stocks of carbon occur in the multiaged and multilayered *E. regnans* forests of the Central Highlands of Victoria. The same suite of factors listed above operate, to varying degrees, across other evergreen temperate forests, particularly in the northwestern United States, southern South America, New Zealand, and elsewhere in southeastern Australia. Collectively, they provide the basis of a generalized framework for predicting high biomass carbon density forests. However, construction of a quantitative predictive model inclusive of all factors is complicated by a lack of process understanding (37), knowledge of species life history characteristics and dynamics, and many interactions and feedback effects (60).

Climate Change Policy Implications

Our results about the magnitude of carbon stocks in forests, particularly in old forests that have had minimal human disturbance, are relevant to negotiations under the United Nations Framework Convention on Climate Change (UNFCCC) concerning reducing emissions from deforestation and forest degradation. In particular, our findings can help inform discussions regarding the roles of conservation, sustainable management of forests and enhancement of forest carbon stocks (ref. 61; http://unfccc.int/resource/docs/2007/cop13/eng/06a01.pdf#page=8). Conserving forests with large stocks of biomass from deforestation and degradation avoids significant carbon emissions to the atmosphere, irrespective of the source country, and should be among allowable mitigation activities negotiated through the UNFCCC for the post-2012 commitment period. Similarly, where practical, management that allows restoration of a forest's carbon sequestration potential should be a recognized mitigation activity.

Our insights into forest types and forest conditions that result in high biomass carbon density can be used to help identify priority areas for conservation and restoration. The global synthesis of site data (Fig. 3 and Table 2) indicated that the high carbon densities of evergreen temperate forests in the northwestern United States, southern South America, New Zealand, and southeastern Australia should be recognized in forest biome classifications.

Concluding Comments

Our findings highlight the value of field-based site measurements in characterizing forest carbon stocks. They help reveal the variability within forest biomes and identify causal factors leading to high carbon densities. Further analyses of existing site data from forests around the world, along with new field surveys, are warranted to improve understanding of the spatial distribution of biomass carbon inclusive of land-use and fire history.

Methods

Biomass of *E. regnans* Forest. The 13,000-ha O'Shannassy Catchment (37.62° S, 145.79° E) has a mean annual rainfall of 1,670 mm, mean annual temperature of 9.4 °C, and annual radiation of 178 W·m^{-2}. Average elevation of the catchment is 830 m, and the area has a generally southerly aspect. Soils are deep red earths overlying igneous felsic intrusive parent material. These are fertile soils with high soil water-holding capacity and nutrient availability compared with most forest soils in Australia. The vegetation is classified as tall eucalypt forest with small pockets of rainforest. The forest is multilayered with an overstory of *E. regnans*, a midstory tree layer of *Acacia dealbata*, *A. frigiscens*, *Nothofagus cunninghamii*, and *Pomaderis aspera*, and a tall shrub layer that includes the tree ferns *Cyathea australis* and *Dicksonia antarctica*.

Inventory sites were established by using a stratified random design to sample the range in dominant age cohorts across the catchment. Stands were aged by a combination of methods, including historical records of disturbance events, tree diameter–age relationships, and cross-checking with dendrochronology. Ages of understory plants ranged from to 100 to 370 years, as determined by radiocarbon dating (62). Different components of the ecosystem survive and regenerate from various previous disturbance events. All living and dead plants >2 m in height and >5 cm in diameter were measured at 318 10-m × 10-m plots nested within 53 sites (each measuring 3 ha) within the catchment. Tree size ranged from 486-cm diameter at breast height (DBH) to 84 m in height (Fig. 1).

Living and dead biomass carbon for each site were calculated by using an allometric equation applied to the inventory data for the individual trees in the plots. The equation related biomass to stem volume and wood density. A reduction factor was included in the equation to account for the reduction in stem volume caused by asymmetric buttresses, based on measurements of stem cross-sections and the area deficit between the actual wood and the perimeter derived from a diameter measurement (43). A second reduction factor was included in the equation to account for decay and hollows in stems of *E. regnans* calculated as a proportion related to tree size. Trees >50 cm DBH begin to show signs of internal decomposition, and by 120 cm DBH actual tree mass is ≈50% of that predicted from stem volume (52). Accounting for decay is an important aspect of estimating biomass from allometric equations derived from stem volume that requires further research, but that is overcome by using direct biomass measurements for the derivation of the allometric equations. Selection of trees for measurement that cover the full range of conditions is also important. Unlike many allometric equations developed for forest inventory purposes, the equation used here was calculated from data representing ecologically mature *E. regnans* trees. Carbon in dead biomass was calculated by using this allometric equation for standing stems with a reduction for decay. Coarse woody debris on the forest floor was measured along 100-m transects (63). The structure of stands with high biomass was described by a bimodal frequency distribution of tree sizes that represented different age cohorts. The maximum amount of biomass carbon occurred in tree sizes 40–100 and 200–240 cm DBH. A lack of comparable high-quality soil data meant we could not provide estimates of below-ground carbon stocks nor consider associated soil carbon dynamics.

Our analyses of biomass carbon stocks used a combination of techniques including field inventory data, biomass measurements, and understanding of carbon cycling processes, as has been recommended by the IPCC (64). The relationship between reflectance from spectral bands, leaf area index, and biomass accumulation is not linear. This is exemplified by the relatively low leaf area of *E. regnans* for the high biomass accumulation in the stemwood of these tall trees. Hence, it is important that all of these types of information are used to estimate biomass carbon stocks and that models are well calibrated with site data, rather than relying solely on remote sensing.

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Global Site Biomass Data. Data on forest biomass were obtained from the literature where biomass was calculated from individual plot data at sites that represent largely mature or primary forest with minimal human disturbance (Table S1). The data were categorized into forest biomes (defined by the IPCC; Table 4.5 in ref. 4). We used field plot data that were available in the published literature as they constitute the most reliable primary data sources. We did not use modeled estimates of biomass carbon or regional estimates derived from forest inventory data and expansion factors to derive wood volume and biomass. A carbon concentration of 0.5 gC·g^{-1} was used where only biomass

data were provided. Where site information was not given, latitude and longitude were obtained from Google Earth (http://earth.google.com) by using the described site location, and mean annual temperature and precipitation were obtained from a global dataset (www.cru.uea.ac.uk/cru/data/tmc.htm). Little or no information was provided by most of the publications concerning how internal decay in trees was accounted for in the biomass estimates. Hence, our estimates of biomass of *E. regnans* that were reduced to account for decay are considered conservative compared with the global site data.

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Review

How strongly can forest management influence soil carbon sequestration?

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Abstract

We reviewed the experimental evidence for long-term carbon (C) sequestration in soils as consequence of specific forest management strategies. Utilization of terrestrial C sinks alleviates the burden of countries which are committed to reducing their greenhouse gas emissions. Land-use changes such as those which result from afforestation and management of fast-growing tree species, have an immediate effect on the regional rate of C sequestration by incorporating carbon dioxide (CO_2) in plant biomass. The potential for such practices is limited in Europe by environmental and political constraints. The management of existing forests can also increase C sequestration, but earlier reviews found conflicting evidence regarding the effects of forest management on soil C pools. We analyzed the effects of harvesting, thinning, fertilization application, drainage, tree species selection, and control of natural disturbances on soil C dynamics. We focused on factors that affect the C input to the soil and the C release via decomposition of soil organic matter (SOM). The differentiation of SOM into labile and stable soil C fractions is important. There is ample evidence about the effects of management on the amount of C in the organic layers of the forest floor, but much less information about measurable effects of management on stable C pools in the mineral soil. The C storage capacity of the stable pool can be enhanced by increasing the productivity of the forest and thereby increasing the C input to the soil. Minimizing the disturbances in the stand structure and soil reduces the risk of unintended C losses. The establishment of mixed species forests increases the stability of the forest and can avoid high rates of SOM decomposition. The rate of C accumulation and its distribution within the soil profile differs between tree species. Differences in the stability of SOM as a direct species effect have not yet been reported.
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Keywords: Soil C dynamics; Forest management; Natural disturbance; C sequestration

Contents

EXHIBIT 14

1. Introduction

Forest ecosystems store more than 80% of all terrestrial aboveground C and more than 70% of all soil organic C (Batjes, 1996; Jobbágy and Jackson, 2000; Six et al., 2002a). The annual CO_2 exchange between forests and the atmosphere via photosynthesis and respiration is ≈ 50 Pg C/yr, *i.e.* 7 times the anthropogenic C emission. An increase in soil respiration would increase the CO_2 emissions from forest ecosystems. In order to mitigate climate change, more C should be sequestered in forest ecosystems and strategies for an adapted forest management are sought (Brown et al., 1996).

According to the Kyoto Protocol (KP), C sequestration in terrestrial sinks can be used to offset greenhouse gas emissions. Currently, European forests absorb 7 to 12% of European emissions with agricultural land being a source and forests a sink of CO_2 (Janssens et al., 2003). Several Europeans countries have so far failed to curtail their greenhouse gas emissions and may rely on the inclusion of terrestrial C sinks in order to meet their emission reduction targets. The Kyoto Protocol states in Article 3.3 that "net changes in greenhouse gas emissions by sources and removals by sinks resulting from direct human-induced land-use change and forestry activities, limited to afforestation, reforestation and deforestation since 1990, measured as verifiable changes in carbon stocks in each commitment period, shall be used to meet the commitments". However, the ability to utilize afforestation as a tool to offset carbon emissions is constrained by available land area. The upper limit for afforestation projects in Europe has been estimated to be 20% of the agricultural land area (Cannell, 1999a). In several countries (*e.g.* Austria, Finland, Sweden, Switzerland) the forest cover is already 50% and further increases are unlikely. In countries with a low forest cover (*e.g.* Ireland, Denmark, Mediterranean countries), however, an increase in the forested area is on the political agenda. KP Article 3.4 allows the use of forest management for C sequestration up to nationally applicable limits (United Nations Framework Convention on Climate Change, 2002; Cannell, 2003; ECCP-Working group on forest sinks, 2003).

National Forest Inventories are used to assess the C sequestration in the aboveground biomass in the context of national greenhouse gas emission reports (Löwe et al., 2000). Measuring changes in soil C is more difficult because its spatial variability is high and soil C accumulation is a slow process (Conen et al., 2004). The rate of formation of stable SOM is between 2 and 12 kg C/ha/yr and much lower than the accumulation of C in the aboveground biomass of a moderately productive forest (Schlesinger et al., 2000). Experiments have found different effects of forest management activities on C sequestration (Johnson, 1992; Post and Kwon, 2000; Johnson and Curtis, 2001). Treatments such as thinning, harvesting, and fertilization modify soil C dynamics and different results can be explained by specific site and soil conditions. In this paper, we review the effects of forest management on C sequestration from the perspective of soil processes. We attempt to generalize about soil processes, that are affected by forest management, scrutinize forest management strategies with respect to their influence on soil C pools, and recommend activities that can lead to long-term C sequestration in forest soils.

2. The pool of soil organic carbon

2.1. Factors influencing the soil C pool

The soil C pool is determined by the balance between C input by litterfall and rhizodeposition on the one hand and the release of C during decomposition on the other side. The turnover of SOM depends on the chemical quality of the C compounds (labile or stable C), site conditions (climate), and soil properties (clay content, soil moisture, pH, nutrient status). Several of these factors are directly or indirectly influenced by forest management. The relative effect of temperature and chemical quality on the decomposition rate has received considerable attention (Trumbore et al., 1996; Liski et al., 1999; Giardina and Ryan, 2000; Knorr et al., 2005; Davidson and Janssens, 2006). The actual turnover rate differs between regions. In boreal peatland forests, excess soil moisture is a limiting factor; in both high elevation and boreal forests the short growing season limits the annual decomposition rate, whereas in mediterranean systems summer droughts inhibit the turnover of SOM.

In a warming world both the primary productivity and the decomposition of SOM accelerate and the soil C pool will move towards a new equilibrium. Forest soils respond more strongly than soils under other forms of land use (Schimel, 1995; Valentini et al., 2000; Rustad et al., 2001). A review of soil respiration experiments concluded that in the long run warming

will reduce the amount of SOM because soil respiration rates will be stimulated more than the productivity (Rustad et al., 2001). In cold regions the response is expected to be more pronounced (Cox et al., 2000; Kirschbaum, 2000). However, 10 years of experimental warming suggest that the loss of soil C is only a temporary effect, because only the labile soil C pool is exhausted (Jarvis and Linder, 2000; Melillo et al., 2002). The response of SOM to rising temperatures is still a subject of controversy, mainly owing to different assumptions on the heterogeneity of fractions of SOM (Kirschbaum, 2004; Powlson, 2005).

The chemical quality of SOM limits the rate of soil respiration (Giardina and Ryan, 2000; Liski et al., 2003). Labile C fractions are quickly mineralized when the temperature regime is appropriate, but the turnover of stable fractions of SOM such as organic compounds associated with the mineral soil is independent of the temperature (Trumbore et al., 1996; Hobbie et al., 2000). Soil microorganisms will acclimatize to changed conditions and the temperature sensitivity of soil respiration will decrease (Luo et al., 2001). Nevertheless, microbial processes are controlled by the quality and availability of substrate and by site properties such as nutrient availability and moisture supply. The substrate availability depends on litter input, the chemical bonding between SOM, and the mineral soil and the chemical structure of the organic compounds.

2.2. Stabilization of soil organic matter

The process of C stabilization is different from the process of accumulation. Accumulation is driven by site factors inhibiting soil respiration, such as excess soil moisture or low temperatures. For an increase of stable soil C pools it is necessary to identify sites where soil properties are conducive to C sequestration. An abundance of reactive surfaces of clay minerals and oxides, where C can form complexes with a low turnover rate, leads to the stabilization of C. The adsorption of organic matter at the mineral surface creates an intimate bond, which leads to an enduring stabilization (Torn et al., 1997; Torn et al., 2002; Hagedorn et al., 2003).

Processes that affect the aggregation of the soil also affect the C sequestration capacity. Stabilized SOM is found in micro-aggregates of the mineral soil. Stabilization of SOM can either be a consequence of the inherent recalcitrance of the molecules, bonding at oxide and clay mineral surfaces, or simply the inaccessibility of SOM for potential microbial grazers (Sollins et al., 1996; Six et al., 2002a,b). The surface accumulation of SOM is positively related to the C input. There are gradual differences between different clay minerals. The bonding of SOM to smectite is tighter than to kaolinite and its turnover time is twice as long (Wattel-Koekkoek et al., 2003). The chemical reaction is a surface condensation that forms stable bondings (Keil et al., 1994; Kennedy et al., 2002). Even over the longest available time series of soil data (150 years) from Russian grasslands, it was shown that the abundance of amorphous minerals was the single most important factor determining the size of the soil C pool. The decisive factor is the physical protection of C upon adsorption to the surface. Once C is

stabilized, the C pool does not change, even when marked differences in land use and climate occur. A comparison of recent data with archived soil material from the Russian steppe shows minimal changes over a century. Despite cultivation and global warming the recalcitrant C stock remained unchanged (Torn et al., 2002).

Stabilization of soil C is not strongly related to site productivity. ^{13}C tracer experiments have shown that the net accumulation of new tree-derived C can be greater in loamy soils with a low productivity than in fertile sandy soils with a high productivity (Hagedorn et al., 2003). This suggests that soil properties play a dominant role.

Soil C sequestration in peatlands is a special case of biochemical stabilization. Under anaerobic conditions the enzyme phenol oxidase is inactive, even when temperatures are rising (Freeman et al., 2001). Consequently, chemically labile SOM accumulates on this site. A change in land management, e.g., the drainage of peatland, can lift this biological constraint and increase the mobilization of SOM. Global warming also promotes drying of peatland and will partially mobilize this huge C pool (Goulden et al., 1998).

3. Afforestation — Kyoto Protocol article 3.3

Forests have a higher C density than other types of ecosystems (Bolin et al., 2000). The terrestrial C pool has been greatly reduced by human activities such as conversion of forests into agricultural land and urban areas. Among the consequences was a reduction of the soil C pool. The currently observed carbon sink is a reversal of past carbon losses (Erb, 2004; Lal, 2004). The *afforestation* of former agricultural land increases the C pool in the aboveground biomass and replenishes the soil C pool. Accumulation occurs until the soil reaches a new equilibrium between C input (litterfall, rhizodeposition) and C output (respiration, leaching). Recent reviews report that the average rate of soil C sequestration was 0.3 t C ha^{-1} yr^{-1} (range 0–3 t C ha^{-1} yr^{-1}) across different climatic zones (Post and Kwon, 2000). On average afforestation increases total C stocks by 18% over a variable number of years (Guo and Gifford, 2002). The initial C accumulation occurs in the forest floor. Its thickness and chemical properties vary with tree species (Vesterdal and Raulund-Rasmussen, 1998; Six et al., 2002a, see chapter 4).

Changes in soil C storage have been reported from a number of studies based on stand chronosequences, paired plots and repeated sampling. Results are quite diverse as soils may gain C, experience no change or even lose C following afforestation (Guo and Gifford, 2002; Vesterdal et al., 2002b). Carbon loss can occur in a brief period following afforestation, when there is an imbalance between C loss by soil microbial respiration and C gain by litterfall. Planting leads to soil disturbance and can stimulate the mineralization of SOM. These losses are not necessarily offset by the low C input by litterfall in a young plantation. Experimental evidence supports this theory. Carbon gains in the upper mineral soil of plantation forests can be offset by losses of old C from deeper parts of the soil (Bashkin and Binkley, 1998; Giardina and Ryan, 2002; Markewitz et al., 2002; Paul et al., 2002; Vesterdal et al., 2002a). In experiments in

South Carolina with *Pinus taeda*, 80% of the C accumulation occurred in the biomass, some accumulation was found in the forest floor and only a small amount ended up in the mineral soil (Richter et al., 1999). A synthesis of afforestation chronosequences in northwestern Europe suggested that soils can contribute about 30% of the total C sequestration in afforested ecosystems (Vesterdal et al., 2006). Mineral soils only sequestered C in two out of the six chronosequences. Radiocarbon analyses and ^{13}C tracer experiments showed that litter-derived C was moved into the mineral soil, but it remained unstabilized and was lost rapidly by decomposition (Trumbore, 2000; Hagedorn et al., 2003). The available long-term experiments found that after several decades more C is moved to the mineral soil (Jenkinson, 1991; Compton et al., 1998; Richter et al., 1999; Gaudinski et al., 2000; Post and Kwon, 2000; Hooker and Compton, 2003; Johnson et al., 2003; Paul et al., 2003; DeGryze et al., 2004).

Following afforestations soils accumulate less C and at a slower rate than the aboveground biomass. Conditions that are not conducive to soil microbial processes, such as sandy texture, low nutrient availability and low pH, can lead to the formation of a thick forest floor layer (Staaf, 1987; Vesterdal et al., 1995; Vesterdal and Raulund-Rasmussen, 1998). It is less certain how C sequestration in the mineral soil is affected by the soil type. In some cases, fertile and clayey soils stored more C, because the production of above- and belowground litter is high and because the formation of organo-minerals complexes protects SOM from decomposition (van Veen and Kuikman, 1990; Liski, 1995; Vogt et al., 1995). In other cases, poor mineral soils were reported to store more C, which was attributed to the slow decomposition and complex formation between organic molecules and metal ions (Vesterdal et al., 2006). In an assessment of soil C stocks in pure Norway spruce and mixed spruce-broadleaved stands on poor soils the C stocks were positively related to soil aluminum pools in an area with relatively poor soils (Berger et al., 2002), because decomposition of SOM is slow in acidic soils. However, the question of how the C stock of different soil types responds to afforestation is not yet resolved (Vejre et al., 2003).

Previous land use affects the C sequestration potential of afforested sites. Pasture soils already have high C stocks and high root densities in the upper part of the mineral soil, so afforestation has a small effect (Guo and Gifford, 2002; Römkens et al., 1999; Murty et al., 2002). Chronosequence studies from New Zealand on former pastures, northern Spain on arable land, and northern England on peatland found that soils initially lost, but later gained C (Romanyá et al., 2000; Halliday et al., 2003; Zerva et al., 2005). In contrast, croplands are more depleted in soil C, and have a greater potential to sequester soil C.

In conclusion, the rate of soil C sequestration is slower than changes in the aboveground C, and it takes decades until net gains occur in former arable soils. Forest floors accumulate C quickly, but most of it in a labile form and for a limited time.

4. Influence of tree species

Despite much research on the role of vegetation in soil formation, a general understanding of the extent of the effect of

tree species across site types has not yet been reached (Stone, 1975; Augusto et al., 2002; Binkley and Menyailo, 2005). Tree species affect the C storage of the ecosystem in several ways. Shallow rooting coniferous species tend to accumulate SOM in the forest floor, but less in the mineral soil, compared with deciduous trees. At identical biomass volumes, trees with a high wood density (many deciduous tree species) accumulate more C than trees with light wood (many coniferous species) (Table 1). Late-successional trees tolerate a higher stem density than pioneer species. Species that occupy different ecological niches can complement each other so that the biomass production of a mixed stand is higher than that for pure stands (Resh et al., 2002; Pretzsch, 2005). For the productivity of a forest over the entire rotation period, its stability against disturbance is important. In Central Europe, mixtures of beech and spruce are the better option, even if pure spruce stands have a higher growth rate (Pretzsch, 2005).

Table 1 shows the differences in soil C pools under common European tree species. Pine forests have remarkably low soil C pools, whereas beech forests have the highest soil and total C pools. It must be kept in mind that mean values for different species also represent site conditions where the species are dominant. For instance, Scots pine forests often grow on shallow and dry soils, which have low C stocks, whereas beech is found on more fertile soils (Callesen et al., 2003, Table 1).

The influence of tree species was studied in common garden experiments with replicated stands of the same species (Fyles et al., 1994; Binkley, 1995; Prescott et al., 2000). In Denmark, a study of seven species replicated at seven different sites along a soil fertility gradient focused on the forest floor C stock (Vesterdal and Raulund-Rasmussen, 1998). Lodgepole pine (*Pinus contorta*), Sitka spruce (*Picea sitchensis*) and Norway spruce had much higher C stocks than European beech (*Fagus sylvatica*) and oak (*Quercus robur*). Similarly, a German experiment showed more C in the forest floor under pine than under beech. This was attributed to the slower decay of pine and spruce litter compared with the litter of deciduous trees (Vesterdal and Raulund-Rasmussen, 1998; Fischer et al., 2002). It should be noted that the effects on the mineral soil are variable. An Austrian study showed higher soil C stocks in pure Norway spruce stands than in mixed spruce-broadleaf stands (Berger et al., 2002). An interaction between tree species and soil type was shown. On poor soils the admixture of spruce increased the soil C pool to a larger extent than on fertile soils. There is insufficient evidence of a consistent effect of tree species on mineral soil C stocks, but the establishment of a spruce forest

Table 1
Wood density of European tree species and median of C pools in European forests (de Vries et al., 2003)

Species	Wood density [kg/m^3]	Tree C [t/ha]	Soil C [t/ha]	$\sum$C [t/ha]
Pinus sylvestris (Scots pine)	490	60	62	122
Picea abies (Norway spruce)	430	74	140	214
Abies alba (Silver fir)	410	100	128	228
Fagus sylvatica (beech)	680	119	147	266
Quercus sp. (oak)	660	83	102	185

after beech leads to the release of C from parts of the mineral soil that is no longer penetrated by roots (Kreutzer et al., 1986). The rooting depth is relevant for soil C because root growth is a most effective way of introducing C to the soil (Jobbágy and Jackson, 2000; Rothe et al., 2002; Vesterdal et al., 2002a).

The conversion of Central European secondary Norway spruce plantations to mixed species forests has been proposed (Spiecker et al., 2004). The primary objective is to reduce storm damages and increase the stability of forests in a changing environment (von Lüpke, 2004; Pretzsch, 2005). Spruce forests generate a higher revenue than mixed species forests or pure beech stands, even when the higher production risk of spruce is taken into account (Assmann, 1961; Dieter, 2001). According to models the long-term C sequestration in Douglas fir (*Pseudotsuga menziesii*) and beech stands is higher than in Norway spruce stands (Burschel et al., 1993; Schöne and Schulte, 1999). In pine stands that have been underplanted with beech, the depth gradient of soil C was changed. In mixed pine–beech stands more C accumulated in deeper parts of the mineral soil, because beech roots reached deeper into the mineral soil. It remains to be seen if this C will be shifted into a stable pool. Nevertheless, the total soil C gain after conversion from pine to beech was low (Fischer et al., 2002).

In conclusion, the effect of tree species on forest floor C stocks is rapid. For the permanence of C sequestration it is more relevant to select tree species that increase the pool of stabilized C in the mineral soil. The driving process is the production of belowground biomass. However, little evidence for the size of this effect is available.

5. Stand management — Kyoto Protocol article 3.4

The thinning regime, the length of the rotation period, specific harvesting techniques, uneven-aged forest management, and continuous-cover forestry are management options with tangible economical and ecological consequences.

Thinning interventions increase the radial growth of the remaining trees at the expense of the total biomass and are not primarily aimed at maximizing C sequestration (Assmann, 1961; Sobachkin et al., 2005). Thinning changes the microclimate. Decomposition of forest floor C is temporarily stimulated because soils become warmer and possibly wetter due to reduced evapotranspiration and the soil C pool decreases (Piene and van Cleve, 1978; Aussenac, 1987). The stand microclimate returns to previous conditions unless the thinning intervals are short and intensities are high. Apart from the changed microclimate, litterfall is temporarily lowered in heavily thinned stands. This reduces forest floor accumulation and contributes to lower soil C stocks. The input of thinning residues into the soil may compensate for losses (de Wit and Kvindesland, 1999). Forest floor C stocks decreased with increasing thinning intensity in field studies in New Zealand, Denmark and the USA (Wollum and Schubert, 1975; Carey et al., 1982; Vesterdal et al., 1995). In the Danish study, forest floor C stocks were inversely related to the basal area, but the change in the forest floor C pool was smaller than its variation between experimental sites with different soil types (Vesterdal et al., 1995).

Less experimental evidence is available for the effect of thinning on the C pool in the mineral soil. The balance in forest soil C depends on the extent of the soil disturbance, the input of thinning residues into the soil and the rate of the litterfall. In an Austrian experiment of a Norway spruce stand, all thinning intensities decreased the C storage (Fig. 1). A thinning intervention in an experimental site with flux measurements in Finland did not result in a net release of C from the ecosystem, because the enhanced growth of the ground vegetation compensated for the reduced C sequestration of the tree layer and the increase of heterotrophic soil respiration was balanced by a decrease in autotrophic respiration of similar magnitude (Suni et al., 2003). In a Korean study, neither soil CO_2 efflux nor litter decomposition was increased with increasing thinning intensity (Son et al., 2004). Any effects on soil respiration rates were apparently overruled by root respiration as indicated by a positive relationship between stand density and soil CO_2 efflux.

Harvesting removes biomass, disturbs the soil and changes the microclimate more than a thinning operation. In the years following harvesting and replanting, soil C losses may exceed C gains in the aboveground biomass. The long-term balance depends on the extent of soil disturbance. Harvesting influences soil carbon in two contrasting ways: harvest residues left on the soil surface increase the C stock of the forest floor and disturbance of the soil structure leads to soil C loss. In a comparative study, harvesting turned forests into a C source because soil respiration was stimulated, or reduced to a lesser extent, than photosynthesis (Kowalski et al., 2004). A scheme of C dynamics after harvest shows the almost immediate C loss that is followed by a slow recovery of the C pool Fig. 2.

A review of harvesting techniques suggested that the effect on soil C is rather small, on average, and depends on the harvesting type (Johnson and Curtis, 2001). Whole-tree harvesting caused a small decrease in A-horizon C stocks, whereas conventional harvesting, leaving the harvesting residues on the soil, resulted in a small increase. Although soil C changes were noted after harvesting, they diminished over time without a lasting effect. In general, different harvesting methods had a far greater effect on ecosystem C due to its effect on the biomass of the regenerating
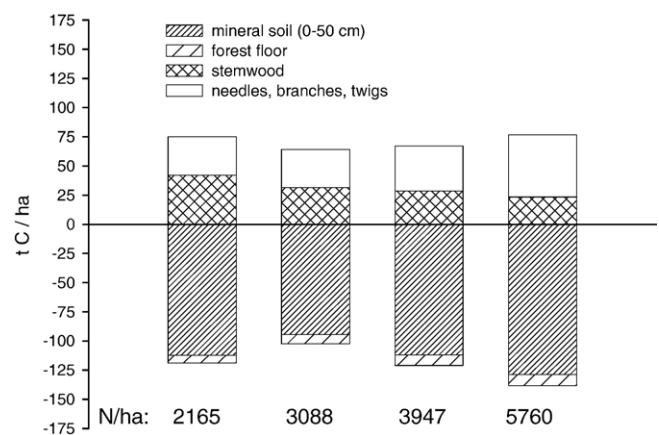

Fig. 1. Carbon in the aboveground biomass and the soil in a thinning experiment eight years after the intervention. "N" denotes the number of stems per ha (Hager, 1988).
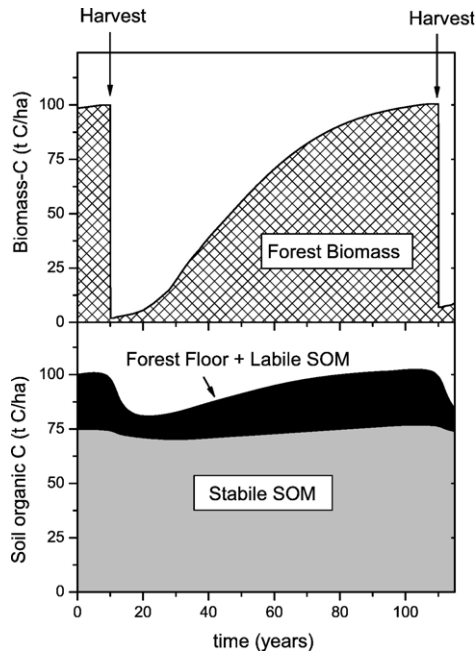
Fig. 2. Simulation of C dynamics in the aboveground biomass and the soil after harvesting. — Assumptions: Biomass-C stock typical for Central European Norway spruce forest; rotation period ≈ 100 years; 25% of SOM are labile, total SOM loss from literature (Olsson et al., 1996).

stand, and a weaker effect on soil C (Johnson and Curtis, 2001; Johnson et al., 2002).

Other researchers report large soil C losses after harvesting. Measurement of net ecosystem C exchange showed that for at least 14 years after logging, regenerating forests remained net sources of CO_2 owing to increased rates of soil respiration (Olsson et al., 1996; Schulze et al., 1999; Yanai et al., 2003). Reductions in soil C stocks over 20 years following clear cuts can range between 5 and 20 t C/ha and are therefore significant compared to the gain of C in biomass of the maturing forest (Pennock and van Kessel, 1997).

Continuous-cover forestry, including selective harvesting, resembles thinning with respect to its effect on the soil C pool, and is considered a possible measure to reduce soil C losses compared with clear-cut harvesting (ECCP-Working group on forest sinks, 2003).

An elongation of the *rotation period* has been proposed to foster C sequestration in forests. Old-growth forests have the

highest C density, whereas younger stands have a larger C sink capacity. After harvest operations, soil C pools in managed forests recover to the previous level. Short rotation lengths where the time of harvest is close to the age of maximum mean annual increment will maximize aboveground biomass production, but not C storage. Longer rotation periods imply that the disturbance frequency due to forest operations is reduced and soils can accumulate C (Schulze et al., 1999). Growth and yield tables suggest that stand productivity declines significantly in mature forest stands. However, even very old unmanaged forests can sequester large amounts of C. A 250-year old beech stand in the Hainich National Park (Central Germany) accumulated more than 4 t C/ha/yr (Knohl et al., 2003). A mature Siberian Scots pine forest and old-growth forests in the USA transferred a higher proportion of its C into the soil than in the early stages of the stand development and continuously increased the soil C stock (Harmon et al., 1990; Schulze et al., 2000). In Sitka spruce plantations in the UK all investigated C pools increased with a 20 year longer rotation, because the productivity of the forest remained very high (Kaipainen et al., 2004). The accumulation of C continues until the C gain from photosynthesis is larger than respiration losses. Late-successional species (*e.g.* beech, Norway spruce) are able to maintain high C sequestration rates for longer than pioneer tree species. Over-mature forest stands are not able to close canopy gaps created by natural mortality or thinning. Consequently the decomposition of SOM is enhanced and decreases the soil C pool.

Chronosequences of spruce in Norway and pine in Northern Germany showed an increase in the thickness of the forest floor layer with age, reaching a steady state after several decades (Sogn et al., 1999; Böttcher and Springob, 2001). No C changes with stand age were found in the mineral soil of the pine forest. A chronosequence of Norway spruce stands in Austria shows only a slight, statistically insignificant, C enrichment of the soil (Fig. 3).

Several modeling studies suggest that very long rotation lengths do not necessarily maximize the total C balance of managed forests (Cannell, 1999b; Liski et al., 2001; Harmon and Marks, 2002). In a simulation experiment of the effect of increased rotation length on C storage in Scots pine plantations in Finland, Germany, and Spain stand productivity declined, because the currently applied harvest age was already beyond the maximum annual increment. Soil C accumulated for several decades but leveled off. The main reason was the decline in
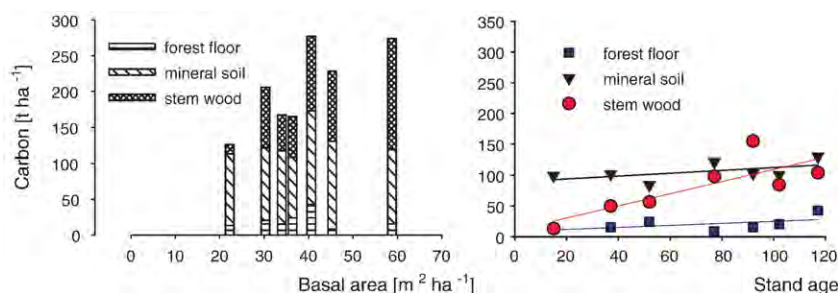

Fig. 3. Carbon pools a chronosequence of Norway spruce stands in Kobernauser Wald/Austria. (a) C pools versus stand basal area, and (b) temporal trend of C pools over stand age (Bauer, 1989).

aboveground litter production, which controlled the soil C pool (Kaipainen et al., 2004).

The elongation of the rotation period has consequences for the wood product market. Carbon that remains in the forest ecosystem cannot be built into wood products and cannot contribute to the substitution of fossil fuels (Schlamadinger and Marland, 1996). It therefore needs to be substantiated, in which types of forests are long rotation periods effective, and where greater volume growth rates in short- to medium-rotation length systems are a better choice.

We conclude that ageing of forests results in increasing C densities in management systems with longer rotation lengths, provided the harvest age is not beyond the age where the forest stand turns from a net sink to a source of C. The magnitude of the effect of increased rotation lengths depends on the current management practice. At the landscape level, longer rotation lengths with more old forests lead to higher C pools than short rotations with only young plantations. A conclusive summary of the long-term C accumulation in forests is still needed. Even when single old stands can sequester C at a high rate, it needs to be demonstrated that these forests are truly representative for the life time of the respective forest type within a given region. — Management interventions such as thinning add value to the stand, but remove biomass. The net effect for C is a loss. Nevertheless, thinning increases the stand stability and therefore offers an important control mechanism for the maintenance of C storage in ecosystems.

6. Disturbances — fire, storm and pest infestation

Recommendations for forest management need to consider the regional disturbance regime. Fire has always played an integral role in the structure and function of forest ecosystems, especially in seasonally dry forests (Fisher and Binkley, 2000). The policy of fire suppression can delay but cannot prevent wildfires over the long term. It leads to an apparent net C accumulation that in fact increases the risk of large C release during catastrophic fires. The role of fire in ecosystem C changes is not straightforward. Several experiments showed that wildfire had caused increases in soil C, which may be driven by the incorporation of charcoal into soils and new C inputs via post-fire N_2 fixation (Schulze et al., 1999; Hirsch et al., 2001; Johnson and Curtis, 2001; Johnson et al., 2004). However, N-fixing plants are not common to all fire-prone ecosystems.

In boreal and mediterranean forests wildfires impose natural limits on the rotation period. Owing to the fire cycle, Siberian forests which are younger than 40 years are a net C source because the rate of decay of forest floor material is larger than biomass accumulation. Forests between 40 and 100 years old are a strong net C sink (≈ 1 t C/ha/yr), older forests are a weak sink (≈ 0.2 t C/ha/yr) (Wirth et al., 2002). Wildfires in tropical forests are not common, but can have serious impacts on the global C cycle. Burning of forested peatlands of Indonesia in 2002 released an equivalent of 13 to 40% of the annual global C emissions from fossil fuels. No management options exist to affect the size of the C pool in tropical peatlands, but protection of these swamp–forest ecosystems is required (Page et al., 2002).

Climate change may increase the frequency and intensity of drought, especially in the Mediterranean and temperate zones. The impacts are site specific and difficult to predict. Water limitations will tend to affect tree growth negatively, but on the other hand the decomposition of soil C may be reduced (Hanson and Weltzin, 2000). Climate change also has an impact on forest pest infestations. A feedback mechanism between ozone, CO_2 and insect populations has been demonstrated in a FACE experiment in North America with aspen (*Populus tremuloides*) and mixed aspen–birch (*Betula papyrifera*) stands. Under changing conditions the population of insects and the frequency of diseases increased. Moreover, forests did not reach the anticipated productivity, either because of damage or the detrimental effect of ozone. The decreased biomass production lowered the rate of soil C formation significantly (Percy et al., 2002; Loya et al., 2003).

Storm damage may result in strongly increased amounts of coarse woody debris on the forest floor. Carbon dynamics after the disturbance are also affected by subsequent management decisions. In the case of a severe reduction in the value, the stand will be harvested and damaged timber will be salvaged. When only parts of the canopy are broken and the stand is already mature, it may be wise to continue the originally planned production cycle (Thürig et al., 2005). Uprooting of trees by windthrow destroys soil structure, which in turn makes protected C accessible for decomposers. Two years after a windthrow in European Russia, the whole ecosystem lost 2 t C/ha to the atmosphere over a 3-month summer period (Knohl et al., 2002).

In conclusion, disturbances consistently lead to the mobilization of C and present a potentially large C source. There are many interdependencies with management activities such as choice of tree species, regulation of stand structure, thinning intensity, and rotation length. Without forest management interventions, the importance of disturbances for C dynamics increases.

7. Improvement of site conditions

7.1. Nitrogen fertilization

Cycling of SOM is influenced by fertilization in contrasting ways. (1) Nitrogen fertilization stimulates tree growth, which potentially increases C inputs into soils through litterfall and rhizodeposition. Increases in tree growth and SOM content due to long-term N fertilization would support this assumption, but there are also reports about decreased root biomass under experimental N additions (Mäkipää, 1995; Eriksson et al., 1996; Andersson et al., 1998; Gundersen et al., 1998). (2) Fertilization increases the nutrient content of the litter material, which stimulates decomposition of SOM (Paul and Clark, 1989). In contrast there are indications that input of mineral N retards decomposition rates of old litter and recalcitrant SOM by suppression of ligninolytic enzymes of soil microbes and by chemical stabilization. Nitrogen stimulates the initial decomposition of fresh litter, but suppresses humus decay in later stages. Radiocarbon and ^{13}C tracer experiments indicated that N additions increased the fraction of old and stable humus in

soils, which may significantly affect soil C storage in the long run (Fog, 1988; Berg and Matzner, 1997; Magill and Aber, 1998; Berg and Meetemeyer, 2002; Neff et al., 2002; Franklin et al., 2003; Hagedorn et al., 2003).

A meta-analysis of 48 experiments from a wide geographical range reported the effects of N, both directly applied as mineral fertilizers and captured by N-fixing plants. A significant increase in soil C was found in the upper mineral soil and in the total soil C pool. A less consistent response was found in a N-fertilization experiment with *Pinus ponderosa* seedlings. The effect of ammonium sulphate on the soil C pool did not differ significantly from the control (Johnson et al., 2000; Johnson and Curtis, 2001).

The effects of N fertilization on the soil C pool vary widely and depend on subsequent soil processes. Often a decrease in the soil C:N ratio is observed, indicating that the N retention effect of the soils is stronger than the C sequestration (Johnson and Curtis, 2001; Jandl et al., 2003). By contrast, a Swedish fertilization experiment to a mature pine forest with very high N applications rates doubled the C pool of the forest floor within 20 years (+5 to 9 t C/ha). This response was interpreted as a consequence of the greatly accelerated growth rate, which in return led to a massive increase in the litter production but also to a decrease in the decomposition rate (Nohrstedt, 1990; Franklin et al., 2003).

Fertilization of forests can lead to the sequestration of larger amounts of soil C than is feasible by afforestation projects. However, the results are site specific and no general recommendation for greater regions can be derived (Canary et al., 2000; Chen et al., 2000).

Nitrogen fertilization stimulates biomass production, but the effect on the soil C pool is more complex. It stimulates the microbial decomposition of SOM, which can lead to a net C loss from the soil and can lead to the formation of nitrogen oxides. The effect of C sequestration in the aboveground biomass is then partly offset by the production of N_2O. This has been shown in agricultural as well as in forest ecosystems (Brumme and Beese, 1992; Mosier et al., 1998). It can be concluded that N fertilization has positive effects on ecosystem level C pools on nutrient-limited sites. However, widespread anthropogenic N deposition has greatly reduced the area of European forests with severe N deficiency. The effects on soil C sequestration are variable.

7.2. Natural aggradation of forests

Many European forests recover from exploitative uses such as litter raking, unregulated fellings, and coppicing (Farrell et al., 2000). Increasing the length of the growing season, N deposition, improved forest management, as well as the enrichment effect of CO_2 has all enhanced the growth rate. In many countries annual increment exceeds the harvest (Spiecker et al., 1996, Fig. 4). Gradually, old forests with a high standing biomass are becoming more common. The current conclusion is that N deposition exerts a fertilization effect on the aboveground biomass, but the effect on soil C is uncertain and at best weak (Nadelhoffer et al., 1999; Davidson and Hirsch, 2001; Oren et al., 2001; Schlesinger and Lichter, 2001; Pussinen et al., 2002).
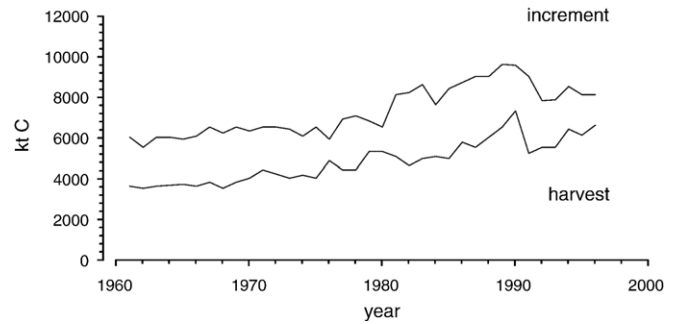

Fig. 4. The persistent difference between increment and harvest leads to C sequestration — example: Austrian forests. Sources: Austrian National Forest Inventory, Austrian Carbon Balance (Weiss et al., 2000).

The interaction between productivity, C sequestration and N availability was confirmed with pan-European data. The C sequestration potential closely follows a deposition gradient: in Northern Europe, where the rate of N deposition is small, C sequestration is also small. A large part of the N is retained in the vegetation and the productivity of the forests is increased. By contrast, both the C sequestration and the N deposition are high in Central and Eastern Europe. The increase in N availability leads to greater productivity and more C sequestration until future constraints to growth are imposed (de Vries et al., 2003). Insufficient water supply may become more common as a result of climate change. The shortage will be aggravated by the increasing water demand of forests, whose productivity will have changed by the increasing length of the growing season and the higher N availability.

7.3. Liming

In Central and Northern Europe many forest soils have been limed in the past in order to regulate soil and surface water chemistry, to protect the ecosystem from irreversible acidification and to mobilize recalcitrant forest floor material (Fiedler et al., 1973; von Wilpert and Schäffer, 2000). However, the target of mobilizing the forest floor is in conflict with the objective of C sequestration. A literature review showed that liming causes a net loss of C in temperate and boreal forests owing to increased microbial activity and DOC leaching (Brumme and Beese, 1992; Jandl et al., 2003; Lundström et al., 2003).

In two fertilizer experiments NPK was applied together with lime. The intention of this 'harmonized amelioration' was the mobilization of nutrients from the forest floor and the provision of readily available nutrients. The overall effect on C is a net loss from the soil (Fig. 5). In the experiment 'Dobrowa' the total soil C content was reduced, whereas in 'Altmanns' C was transferred from the previously inactive mor layer to the mineral soil. In both cases, SOM was mobilized.

7.4. Water management — peatlands

In peat soils, excess water suppresses the rate of decomposition of SOM and leads to C accumulation. It does not influence
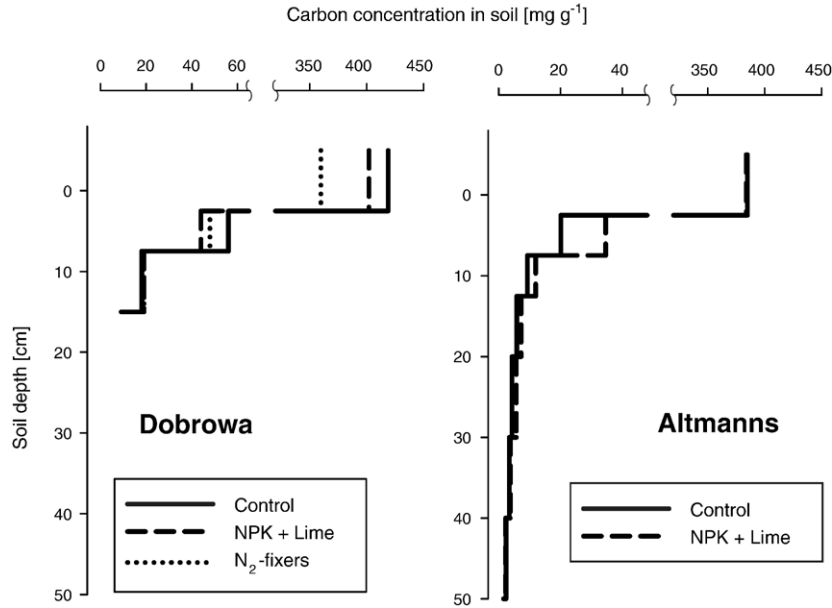
Fig. 5. Effect of NPK fertilization, liming and planting of N_2-fixers (*Lupinus heterophyllus*) on soil C in two Austrian amelioration experiments; Dobrowa (Jandl et al., 2003) and Altmanns (Jandl et al., 2002).

its stabilization. As a result of soil anoxia natural peatlands emit the greenhouse gas methane (CH_4) while nitrous oxide (N_2O) emissions from natural mires are insignificant (Martikainen et al., 1993). In the Nordic countries, approximately 15 million ha peatland have been drained for forestry (Paavilainen and Päivänen, 1995). Drainage stimulates the productivity of forested peatlands and enables the establishment of a forest in otherwise treeless peatlands. Global warming and drainage would result in peatlands becoming drier and the increased microbial activity could turn boreal mires from C sinks to C sources (Moore and Dalva, 1993; Silvola et al., 1996). On the other hand CH_4 emissions would decrease for the same reasons (Nykänen et al., 1998). The increased decomposition of organic matter following drainage is at least partly compensated by the higher inflow of C into the system through increases in plant biomass and primary production and decreases in soil temperature, soil pH and litter decomposability (Minkkinen et al., 1999; Laiho et al., 2003). Leaching of dissolved organic C (DOC) increases immediately after digging the drainage network, but returns to pre-drainage levels later on (Ahtiainen, 1988; Sallantaus, 1994). Direct measurements of soil C balances in peatlands are rare, but both decreases and increases following drainage have been reported (Braekke and Finer, 1991; Sakovets and Germanova, 1992; Minkkinen and Laine, 1998; Minkkinen et al., 1999; Gustafsson, 2001; Hargreaves et al., 2003; Byrne and Farrell, 2005). As C stores in vegetation nearly always increase following forestry drainage, peatlands may remain C sinks despite C losses from the soil (Minkkinen et al., 2002; Hargreaves et al., 2003; Laiho et al., 2003). To conclude, forest drainage decreases CH_4 emissions, increases N_2O and CO_2 emissions from peat, but increases C sequestration in the vegetation. Simulations using data from Finnish peatlands indicated that the radiative forcing of forest drainage may even be negative, *i.e.* drainage may have a "cooling" effect on the global

climate during the first centuries (Laine et al., 1996; Minkkinen et al., 2002).

7.5. Site preparation

Site preparation promotes rapid establishment, early growth and good survival of seedlings. Techniques include manual, mechanical, chemical methods and prescribed burning, most of which include the exposure of the mineral soil by removal or mixing of the organic layer. The soil disturbance changes the microclimate and stimulates the decomposition of SOM, thereby releasing nutrients (Palmgren, 1984; Johansson, 1994). Another effect is improved water infiltration into the soil and better root development. The recent trend towards nature-oriented forest management reduces the importance of site preparation. A review on the effects of site preparation showed a net loss of soil C and an increase in productivity (Johnson, 1992). The effects varied with site and treatment. Several studies that compared different site preparation methods found that the loss of soil C increased with the intensity of the soil disturbance (Johansson, 1994; Örlander et al., 1996; Schmidt et al., 1996; Mallik and Hu, 1997). At scarified sites, organic matter in logging residues and humus, mixed with or buried beneath the mineral soil, is exposed to different conditions for decomposition and mineralization compared with conditions existing on the soil surface of clear-cut areas. The soil moisture status of a site has great importance for the response to soil scarification. The increase in decomposition was more pronounced at poor, coarsely textured dry sites than on richer, moist to wet sites (Johansson, 1994). Sandy soils are particularly sensitive to management practices, which result in significant losses of C and N (Carlyle, 1993). Intensive site preparation methods might result in increased nutrient losses and decreased long-term productivity (Lundmark, 1988). In most of the reviewed studies biomass production was favored by site

preparation and this effect may balance or even outweigh the loss of soil C in the total ecosystem response. In conclusion, there is in general a net loss of soil C with site preparation, which increases with the degree of disturbance. The chosen technique of site preparation is important and will determine if the net C effect of the activity is positive or negative.

8. Discussion

Forest soils are considered to have a considerable potential as C sinks (Frolking et al., 1996; Perruchoud et al., 1999; Halliday et al., 2003). Modeling studies suggest that European forest soils are currently sequestering 26 Tg C yr^{-1}, *i.e.* 30–50% of the estimated C sink in the forest biomass (Liski et al., 2002). However, modeled accumulation rates of soil C have so far not been detected in nature. Field and process-based studies conclude that the rate of soil C accumulation is small, compared with the C accretion in the aboveground biomass, because only a small proportion of plant-derived C becomes stabilized in the mineral soil (Martin and Haider, 1986; Mayer, 1994; Richter et al., 1999; Kaiser and Guggenberger, 2003; Giardina et al., 2005). Either the understanding of the geochemical C fluxes is still incomplete, or the accumulation occurs, but much slower than predicted, or the changes are not detectable owing to the spatial and temporal variability of soil C.

Efforts to increase soil C storage should ideally increase the pool of recalcitrant C. Nevertheless, an increase in less stable pools is also relevant when these pools are sustained by a continuous input of organic matter. The recovery of degraded forest ecosystems and the afforestation of land after agricultural use are cases, that affect mostly the C pool in the forest floor, which is not stabilized by the formation of organo-mineral complexes.

In regions where exploitative historic land-use practices have reduced the soil C pool, one option is to foster the restoration of the previous forest type. This can be achieved by ameliorations, such as underplanting, liming, and fertilizer application, or through a natural aggradation process, which is supported by anthropogenic N deposition and climatic change (Jandl et al., 2002). The response of the aboveground biomass is often an increase in productivity. A temporary soil C sink exists, where intensive litter raking has greatly depleted the soil C pool, and where the previous level can be re-established. At other sites, the nutrient export has created unfavorable conditions for soil microorganisms and biologically inactive mor humus layers have formed. Their mobilization leads to the formation of more favorable humus forms (Jandl et al., 2003). There, site recovery leads to a reduction of the C pool in the forest floor. The C losses may or may not be offset by C gains in the mineral soil and the aboveground biomass. Forest floor C is physically and chemically less stable than C in the mineral soil and can be respired within a few decades under changed site conditions (Covington, 1981; Hamilton et al., 2002). Its mineralization can very quickly turn forest soils from a C sink into a C source.

Afforestation affects the C pool in the forest floor more strongly than in the mineral soil. The accumulation of a forest floor layer in, *e.g.*, a conifer forest is a C sink. The forest floor should not be discounted with regard to C sequestration, al-

though this C pool is more volatile than mineral soil C and can be lost upon changing site conditions. A long-term consequence of afforestation is the gradual incorporation of C in the mineral-associated soil C pool. This effect is by no means intermediate (DeGryze et al., 2004).

Forest management can stimulate the decomposition of the forest floor and can modify its quality by the tree species selection (quantity and chemical quality of litter, rooting depth) and the thinning regime (microclimate). Several studies have stressed the negative impacts of intensive site preparation on the C balance (Johnson, 1992; Schmidt et al., 1996; Mallik and Hu, 1997). Critical situations are after thinning interventions and the end of the rotation period. Frequent thinning of stands throughout the rotation increases their stability. The lightest thinning operation removes at least those trees which would fall victim to natural mortality (Assmann, 1961). Maintaining a high stand density would maximize the C pool, but would also bear a considerable risk of disturbance. A lower stand density increases the stability of individual trees and thus reduces the risk of C losses

Table 2
Summary of the effects of specific forest management actions on ecosystem C stocks ('+'...increases C stock, '−'...decreases C stock; '±' neutral with respect to C stock)

Afforestation
+ Accumulation of aboveground biomass formation of a C-rich litter layer and slow build-up of the C pool in the mineral soil
± Stand stability depends on the mixture of tree species
− Monotone landscape, in the case of even-aged mono-species plantations

Tree species
+ Affects stand stability and resilience against disturbances; effect applies for entire rotation period; positive side-effect on landscape diversity, when mixed species stands are established
− Effect on C storage in stable soil pools controversial and so far insufficiently proven

Stand management
+ Long rotation period ensures less disturbance due to harvesting, many forest operations aim at increased stand stability, every measure that increases ecosystem stability against disturbance
± Different conclusions on the effect of harvesting, depending if harvest residues are counted as a C loss or a C input to the soil
− Forests are already C-rich ecosystems — small increase in C possible; thinning increases stand stability at the expense of the C pool size; harvesting invariably exports C

Disturbance
+ Effects such as pest infestation and fire can be controlled to a certain extent
± Low intensity fires limit the risk of catastrophic events
− Catastrophic (singular) events cannot be controlled; probability of disturbance can rise under changed climatic conditions, when stands are poorly adapted

Site improvement
+ N fertilization affects aboveground biomass; effect on soil C depends on interaction of litter production by trees and carbon use efficiency of soil microbes
± Drainage of peatland enables the establishment of forests (increased C storage in the biomass) and decreases CH_4 emissions from soil, but is linked to the increased release of CO_2 and N_2O from the soil
− Liming and site preparation always stimulate soil microbial activity. The intended effect of activating the nutrient cycle is adverse to C sequestration; N fertilization leads to emission of potent greenhouse gases from soils

due to disturbance. The presence of biomass residues left on site after thinning plays a role in evaluating C pools. Our view is that this pool of thinning residues is not relevant for C sequestration. Nevertheless, we are aware that thinning residues are a C pool that is not clearly represented, because it neither counts as forest floor material nor as wood product.

A trend towards nature-oriented silviculture and continuous-cover forestry will reduce the relevance of site preparations and clear cuts (Pommerening and Murphy, 2004). The effect of continuous-cover forestry is difficult to assess at the present time, because the long-term impacts have not yet been measured. It is characterized by the avoidance of large canopy openings. The forest floor layer will therefore be less exposed to decomposition and will be rather stable in time, but effects on the recalcitrant C pool as a direct result of management specific processes in the mineral soil are not expected. A relevant factor may be the slow formation of organo-mineral complexes in the undisturbed soil (DeGryze et al., 2004).

The relevance of tree species for the objective of C sequestration in Central Europe invariably leads to a weighing of the benefits and peculiarities of Norway spruce versus beech. On most acidic to neutral sites, spruce produces more stem volume. Consequently, many mixed species stands in Central Europe have been converted to "secondary spruce forests". For the objective of C sequestration, the relevant characteristic is total biomass production. The higher C density of beech wood and the higher production of non-stem aboveground biomass mean, that the total aboveground accumulation of C of the two species is not far apart. Moreover, beech develops a deep rooting system which increases the C pool in the mineral soil (Kreutzer et al., 1986), allowing longer rotation periods than spruce, and increasing the stability of mixed stands (Pretzsch, 2005). Mixed species stands are also less susceptible to pest infestations, whereas secondary spruce forests are notorious for extensive bark beetle damage (Baier et al., 2000). Considering these factors we conclude, that mixtures of beech and spruce are a better forest management option than pure spruce stands, when terrestrial C sinks need to be optimized.

Even though single old-growth forests can have impressive rates of C sequestration (Schulze et al., 2000; Knohl et al., 2003), we are skeptical with respect to the role of the elongation of the rotation period of forests. Forests beyond a certain age are susceptible to disturbances. The aboveground productivity declines with age (Ryan et al., 2004). Openings in the canopy are closed more slowly than in younger stands and old stands are therefore more vulnerable to windthrow. Limits in the expectable life span of forests are evident from records of long-term experimental plots. Only a few of these studies can be continued over decades, whereas most stands disintegrate when they reach maturity (Johann, 2000). Recommendations for the elongation of the rotation period need to be based on experimental evidence obtained from a representative set of stands. These trials still await implementation.

This evaluation of forest management activities indicates that few practices are clearly good or bad with respect to C sequestration (Table 2). Productive forests with a high rate of aboveground and belowground litterfall circulate a large amount of C and are a precondition for efficient C sequestration. Their overall impact depends on the degree of soil disturbance in the course of harvesting or thinning operations and the degree of stability against disintegration of the stand structure. Two governing processes are the quantity and quality of the litter (C input) and the decomposition of SOM (C output). Optimized forest management with regard to soil C sequestration should aim to secure a high productivity of the forest on the input side, and avoid soil disturbances as much as possible on the output side. Our review shows that forest management directly influences the C flow into the soil. The pathways are both above- and belowground C fluxes. The subsequent stabilization of SOM in the soil partly depends on soil properties which cannot be influenced by stand management. What is beyond dispute is that the formation of a stable soil C pool requires time. Avoiding soil disturbances is important for the formation of stable organo-mineral complexes which in turn are crucial elements in the process of C soil sequestration.

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Review

Harvest impacts on soil carbon storage in temperate forests

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ABSTRACT

Forest soil carbon (C) storage is a significant component of the global C cycle, and is important for sustaining forest productivity. Although forest management may have substantial impacts on soil C storage, experimental data from forest harvesting studies have not been synthesized recently. To quantify the effects of harvesting on soil C, and to identify sources of variation in soil C responses to harvest, we used meta-analysis to test a database of 432 soil C response ratios drawn from temperate forest harvest studies around the world. Harvesting reduced soil C by an average of $8 \pm 3\%$ (95% CI), although numerous sources of variation mediated this significant, overall effect. In particular, we found that C concentrations and C pool sizes responded differently to harvesting, and forest floors were more likely to lose C than mineral soils. Harvesting caused forest floor C storage to decline by a remarkably consistent $30 \pm 6\%$, but losses were significantly smaller in coniferous/mixed stands (-20%) than hardwoods (-36%). Mineral soils showed no significant, overall change in C storage due to harvest, and variation among mineral soils was best explained by soil taxonomy. Alfisols and Spodosols exhibited no significant changes, and Inceptisols and Ultisols lost mineral soil C (-13% and -7%, respectively). However, these C losses were neither permanent nor unavoidable. Controls on variation within orders were not consistent, but included species composition, time, and sampling depth. Temporal patterns and soil C budgets suggest that forest floor C losses probably have a lesser impact on total soil C storage on Alfisols, Inceptisols, and Ultisols than on Spodosols, which store proportionally large amounts of C in forest floors with long C recovery times (50–70 years). Mineral soil C losses on Inceptisols and Ultisols indicate that these orders are vulnerable to significant harvest-induced changes in total soil C storage, but alternative residue management and site preparation techniques, and the passage of time, may mitigate or negate these losses. Key findings of this analysis, including the dependence of forest floor and mineral soil C storage changes on species composition and soil taxonomic order, suggest that further primary research may make it possible to create predictive maps of forest harvesting effects on soil C storage.

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Contents

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EXHIBIT 15

1. Introduction

Forest soils contain a globally significant amount of carbon (C). Approximately half of Earth's terrestrial C is in forests (1146×10^{15} g), and of this amount, about two-thirds is retained in soil pools (Dixon et al., 1994; Goodale et al., 2002; Johnson and Curtis, 2001). On an annual basis, detrital C inputs (61.4×10^{15} g) slightly exceed respiratory C losses (60×10^{15} g) from soils, suggesting that soil C storage may contribute to the 'missing C sink' implicated in the global C budget (Schimel, 1995). Since the net C balance of forest soils (whether sequestering or losing C) generally is a small difference between two large fluxes, a relatively minor change in either term could have major impacts on the forest C budget.

Soil C storage is important not only because of its role in the global C cycle (Kirschbaum, 1995), but also because it affects forest productivity (Jurgensen et al., 1997; Grigal and Vance, 2000). Since soil C is a principal source of energy for the nutrient-recycling activities of heterotrophic soil organisms, the maintenance of soil C stocks is vital for sustaining forest productivity (Attiwill and Adams, 1993; Vance, 2000). Furthermore, soil C is one of the principal components of soil organic matter (SOM), which also contains significant amounts of water and nitrogen—all of which are exchanged between the biosphere and the atmosphere to affect Earth's atmospheric chemistry, energy and water budgets, and climate (Conrad, 1996; Raich and Schlesinger, 1992). Therefore, improving our understanding of the factors that affect forest soil C storage is fundamentally important for anticipating changes in ecosystem goods and services ranging from forest products, to water resources, to greenhouse gas mitigation.

Forest management, especially the harvesting of biomass for forest products, can significantly affect soil C storage. Forest harvesting may shift the soil C balance by many mechanisms, including altering the quantity and quantity of detrital C inputs, changing soil microbial community composition, and affecting the climatic conditions that drive plant and microbial processes (Chen et al., 1995; Covington, 1981; Gray et al., 2002; Hassett and Zak, 2005; Zogg et al., 1997). However, soil C measurements frequently have high levels of spatial and temporal variability, making it difficult to detect the effects of management on soil C storage within an individual site (Homann et al., 2001, 2008; Magrini et al., 2000). Fortunately, the statistical technique of meta-analysis can be used to find underlying patterns that are broadly consistent across studies, even when such patterns are so obscured by variability as to be rendered undetectable within each individual study. In meta-analysis, the results of many individual experiments are synthesized by compiling a distribution of responses to a treatment applied at multiple locations, or at different times. Analysis then proceeds by testing this distribution for an overall effect of the treatment, and by identifying the sources of variation among responses to that treatment. We collected soil C data from experiments that compared harvested and unharvested temperate forest sites, and used meta-analysis to answer the following questions. First, is there a consistent, overall effect of forest harvesting on soil C storage? Second, what factors control variation in soil C responses to harvest? Third, is it possible to identify soil C pools that exhibit different levels of vulnerability to harvest-induced change? Finally, how much does soil C storage change in response to harvest and site preparation techniques commonly practiced in temperate forests?

2. Methods

We conducted this meta-analysis following the general methods of Curtis (1996) and Johnson and Curtis (2001). We searched the peer-reviewed literature using keyword searches within the online reference databases ISI Web of Science, BIOSIS, Agricola, and CAB Direct. Keyword search strings were combinations of terms such as: forest, timber, logging, harvest, clearcut, thinning, coppice, residue, management, and soil C. In the process of inspecting over 6500 references returned by our literature searches, we found 75 publications that met our inclusion criteria of: (1) reporting control (unharvested) and treatment (harvested) soil C values, and (2) being conducted in a temperate forest (4–8 months of mean air temperature >10 °C; Köppen, 1931). Acceptable controls for harvested forest soils were either pretreatment soil C values, or soil C observations from nearby reference stands that were not harvested. The latter type of control value included both simultaneous measurements of harvested and unharvested soils, and chronosequences, in which case the oldest stand was treated as the control. As a minimum, control stands were those which had not been harvested within the past 30 years, although some publications had control stands that had not been harvested for 1–2 centuries. Therefore, our meta-analysis does not bear specifically on either old-growth conversions or short-rotation plantation forestry, but rather a mix of many different harvest regimes practiced across time scales. Although they did not meet the temperate climate requirement, we included several publications from the southeast United States due to the importance of this region to forestry in the U.S. We accepted soil C concentrations and pool sizes as metrics of soil C, and used meta-analysis to determine whether concentrations and pool sizes significantly differed in their responses to harvest. Among publications that reported both concentrations and pool sizes, we chose pool sizes as the response parameter, and we calculated soil C pool sizes for publications that reported concentrations and bulk densities. The term 'C storage' as used in this study denotes C pool sizes only; we use the more general term 'soil C' when referring to soil C estimates that encompass both types of reporting units.

We extracted meta-data (potentially useful predictor variables) from each publication, including temporal, climatic, soil chemical and physical data, measurement units, and treatment and analytical methods. One pertinent distinction in the soil physical data category was the soil layer sampled. We extracted data for organic and mineral soil layers separately, and coded the data so that we could test for differences between soil layers defined as forest floor (mostly organic horizons), surface mineral soil (5–20 cm deep), deep mineral soil (20–100 cm), and whole mineral soil profile. Inconsistencies among the soil layers reported in primary publications are considered in the Discussion. Regarding our classification of harvest, residue management, and site preparation techniques, we categorized studies as follows, provided meta-data were descriptive enough to ascertain the specific practices used. First, each response ratio was classified according to its harvest type as a clearcut, in which all overstory trees were cut down, or a thinning, if some proportion of the overstory was left intact. If possible, we then categorized each response ratio according to harvest intensity, a categorical variable to distinguish whole-tree and stem-only harvests. Finally, for each response ratio, we noted the residue management and site

Table 1
Categorical factors tested as potential predictor variables in the meta-analysis.

Factor	Levels
Reporting units	Pool size, concentration
Soil layer	Forest floor, surface mineral soil (<20 cm), deep mineral soil (>20 cm), whole mineral soil profile
Species composition	Hardwood, coniferous/mixed
Soil taxonomic order	Alfisol, Andisol, Inceptisol, Mollisol, Spodosol, Ultisol
Geographic group	NE U.S., NW U.S., SE U.S., SW U.S., Europe, Australia, Asia
Harvest type	Thin, clearcut
Harvest intensity	Stem only, whole-tree
Residue management/site preparation methods	None, residue removed, residue spread, broadcast burn, intensive (tillage)
Time since harvest	0–5, 6–20, 21–40, >40 years
Soil texture[a]	Coarse (mostly sand), fine (mostly silt or clay)

The levels listed within each factor define the response ratio groups used for Q_b analysis in Table 2.

[a] Mineral soils only.

preparation methods employed after harvest. We defined residue management as the manipulation of the unused portions of harvested forest biomass, such as tops, limbs, and leaves. We defined site preparation as any process employed with the aim of improving tree regeneration (natural or planted) on the post-harvest landscape. The complete list of factors by which we categorized the response ratios in the database may be found in Table 1.

Meta-analysis estimates the magnitude of change in a parameter (i.e., the 'effect size') in response to an experimental treatment, which may be applied across a wide range of experimental systems and conditions. We used the ln-transformed response ratio R to estimate treatment effect size:

$$\ln(R) = \ln\left(\frac{\bar{X}^E}{\bar{X}^C}\right) \quad (1)$$

where $\bar{X}^E$ is the mean soil C value of treatment (harvested) observations, and $\bar{X}^C$ is the mean soil C value of control observations for a given set of experimental conditions. The number of response ratios (k) from a given publication depends on how many sets of experimental conditions are imposed. For example, one publication with soil C storage data from a control soil and from four different levels of thinning would yield $k = 4$ response ratios, or 'studies'. Because it is unitless, the effect size R is a standardized metric that allows comparison of data between experiments reporting responses in different units (Hedges et al., 1999). After back transformation ($e^{\ln(R)}$), R can be conceptualized as the proportional or per cent change in soil C relative to its control value. When error terms and sample sizes are reported for each $\bar{X}^E$ and $\bar{X}^C$, a parametric, weighted meta-analysis is possible, but many publications we found did not report these data. Therefore, in order to include as many studies as possible, we used an unweighted meta-analysis, in which confidence intervals around mean effect sizes are generated with nonparametric resampling techniques (bootstrapping; Adams et al., 1997). We performed analyses using MetaWin software (Sinauer Associates, Sunderland, MA, USA).

One of our principal goals in this analysis was to identify the categorical variables that were the best predictors of variation in soil C responses to harvest. Accomplishing this task with meta-analysis is similar to using analysis of variance to partition the total variance of a group of observations (Q_t, the total heterogeneity) into two components: within- and between-group heterogeneity (Q_w and Q_b, respectively; Hedges and Olkin, 1985). In such a Q_b analysis, a categorical variable that defines a group of response ratios with a large Q_b is a better predictor of variation (or heterogeneity) than a categorical variable associated with small response-group Q_b. In order to determine which categorical variables were the 'best' predictors of variation, we followed the hierarchical approach detailed in Curtis (1996) and Jablonski et al.

(2002). Briefly, we ran meta-analysis on the entire database to determine which categorical variable had the lowest P value, and then divided the database into the categorical groups defined by that variable. Then, within each of these groups, we ran meta-analyses again for each categorical variable, and split the studies into the groups defined by the categorical variable with the lowest P value. We performed this variance-partitioning exercise for a total of three iterations, at which point we felt it prudent to go no further due to limited sample sizes and possible confounding relationships. When, during the course of these Q_b iterations, we found multiple categorical variables with the same P value, we selected the one with the highest Q_b. In Q_b analyses, and all other meta-analyses, we accepted tests with $P < 0.05$ as statistically significant.

While our literature search was not exhaustive, the database we developed for this analysis is quite large, comprising 432 soil C response ratios from 75 papers published between 1979 and 2008. These publications correspond to forest harvests conducted in temperate forests around the world, and the full dataset is available at <http://nrs.fs.fed.us/carbon/data>. Publications included in the analysis are denoted in the References section with a (*), and basic information is provided for each publication in Appendix A.
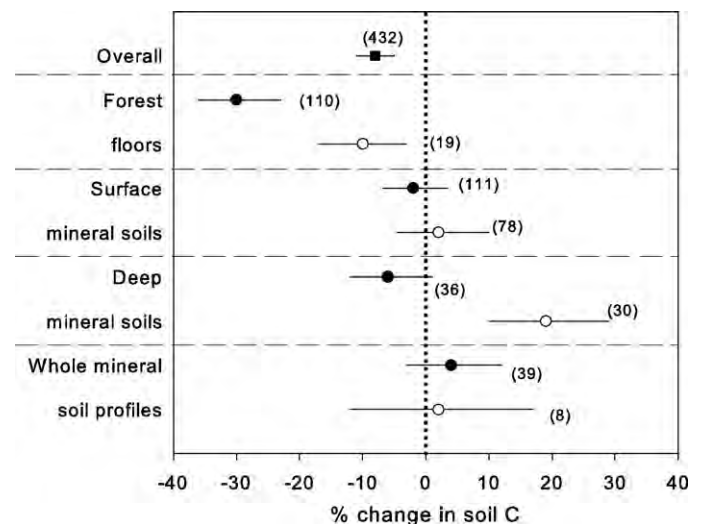

Fig. 1. Soil C changes due to forest harvesting, overall and by soil layer. All points are mean effect sizes ± bootstrapped 95% confidence intervals, with the number of studies (k) in parentheses. Groups with confidence intervals overlapping the dotted reference line (0% change) show no significant change in soil C due to harvest. The filled square at the top shows the overall effect of harvesting on soil C, including C pool sizes and concentrations from forest floors and mineral soils. Within each soil layer, mean effect sizes are shown separately for C pool sizes (C storage; filled circles) and C concentrations (open circles).

Table 2
Between-group heterogeneity (Q_b) among the k studies comprising each response parameter.

Response parameter	k	Reporting units	Soil layer	Species composition	Soil taxonomic order	Geographic group	Harvest type	Harvest intensity	Residue management/ site prep	Time since harvest	Soil texture
Overall soil C	432	2.95**	10.12**	4.38**	3.85**	4.71**	<0.01	0.17	0.33	0.70	–
Forest floor C storage	110	–	–	1.40*	2.27	3.64*	0.37	1.00	0.82	1.52	–
Coniferous/mixed	48	–	–	–	1.38	1.42	<0.01	0.81	0.86	0.40	–
Hardwood	62	–	–	–	0.25	2.32**	<0.01	0.21	0.42	0.65	–
Mineral soil C storage	186	–	0.17	0.56**	1.90**	0.96*	0.09	0.26	0.40	0.29	0.12
Alfisols	32	–	0.03	1.01**	–	0.57	<0.01	0.06	0.01	0.44	0.24
Inceptisols	28	–	0.15	0.31	–	0.14	0.10	0.09	0.91**	0.81**	NA
Spodosols	57	–	0.35*	0.04	–	0.20	0.08	<0.01	0.12	0.14	<0.01
Ultisols	37	–	0.27*	<0.01	–	<0.01	0.06	<0.01	0.21	0.14	NA

The overall soil C response to harvest includes all studies in the database, and is separated into forest floor and mineral soil groups. Forest floor and mineral soil C storage response ratios include only pool sizes, which were significantly different from concentrations in the overall analysis. Within forest floors and mineral soils, Q_b is shown separately for response ratio groups defined by the categorical variable with the lowest P value (species composition for forest floors; soil order for mineral soils). See Table 1 for the levels that comprise each factor (categorical group) included in the Q_b analysis.

* Statistical significance of Q_b is denoted by $P < 0.05$.

** Statistical significance of Q_b is denoted by $P < 0.01$.

3. Results

3.1. Overall effects and primary sources of variation

Averaged across all studies, forest harvesting resulted in a small, but significant reduction in soil C (−8%, Fig. 1). Our meta-analysis revealed several important sources of variation underlying this overall effect, however (Table 2). The two most significant categorical factors accounting for among-study variation in harvest impacts were the soil layer sampled (forest floor vs. mineral soil) and the reporting units (concentration vs. pool size). Specifically, the forest floor was the only soil layer to show an overall, significant change in C storage following harvest (Fig. 1; −30%), an effect which was paralleled by a much smaller impact on forest floor soil C concentration (−10%). Harvesting had no overall effect on surface, deep, or whole mineral soil C storage, but deep mineral soil C concentrations increased by an average of 19%. The significant difference between harvest impact results reported as C concentrations compared to those reported as C pool size, or storage, led us to restrict all further analyses to results reported as C storage.
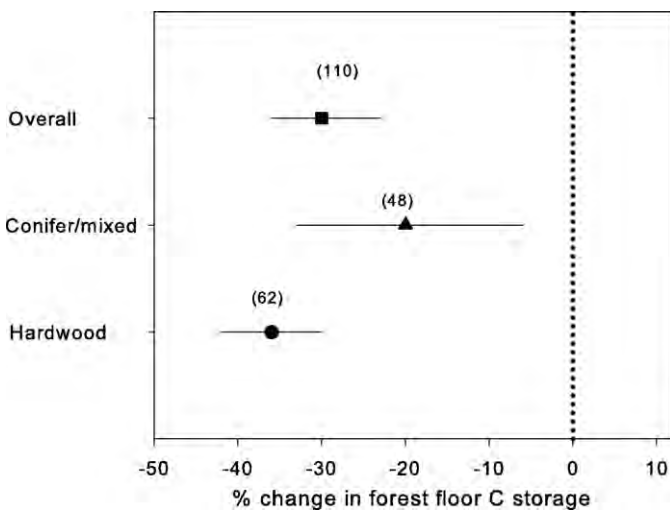

Fig. 2. The effects of harvesting on forest floor C storage, overall and by species composition. Plots show means ± bootstrapped 95% confidence intervals, with number of studies (k) in parentheses.

3.2. Variation within soil layers in harvest impacts

3.2.1. Forest floors

The overall effect of harvest on forest floor C storage was remarkably consistent among studies, with little variation due to differences in soil taxonomic order, time since harvest, or harvest intensity (Table 2). The principal predictor of variation in harvest impacts on C storage was tree species composition, with coniferous/mixed forests losing less forest floor C than do hardwood forests (Fig. 2; −20% and −36%, respectively). Geographic location also accounted for significant between study variation (Table 2), but this was due to two studies from the southeast U.S. that showed a 50% increase in forest floor C storage, both of which were from Mattson and Swank (1989).

In forests growing on Spodosols, forest floor C storage after harvest showed a lengthy, but relatively well-constrained recovery period, on the order of 50–70 years (Fig. 3A). Long-term studies of forest floor C recovery on other soil orders are lacking (Fig. 3B).

3.2.2. Mineral soils

Soil order was the most important predictor of between-study variation in harvest impacts on mineral soil C storage (Table 2). When all layers were analyzed together, mineral soils from Inceptisols and Ultisols had significant declines in C storage following harvest (−13% and −7%, respectively), while Spodosols and Alfisols were not significantly affected (Fig. 4). Among Inceptisol mineral soils, time since harvest was the principal source of between-study variation, with C storage declining by 25% within 5 years of harvest, but recovering to control values within 6–20 years. Both Ultisols and Spodosols showed significant differences in response to harvest between surface and deep mineral soil layers (Fig. 4). Among Ultisols, surface mineral soils lost significant C (−7%, $P = 0.016$), while deep mineral soils were unchanged. Spodosols showed the opposite pattern, with no loss in surface mineral soil C storage but a significant decline in deep mineral soil C (−9%, $P = 0.031$). Species composition was a significant predictor of variation among Alfisols, with hardwoods exhibiting a decline in C storage in response to harvest (−36%, $P = 0.001$) but with no harvest effect seen in coniferous and mixed forests (Fig. 4).

In contrast to forest floors, species composition and geographic factors were of secondary importance in accounting for variation in
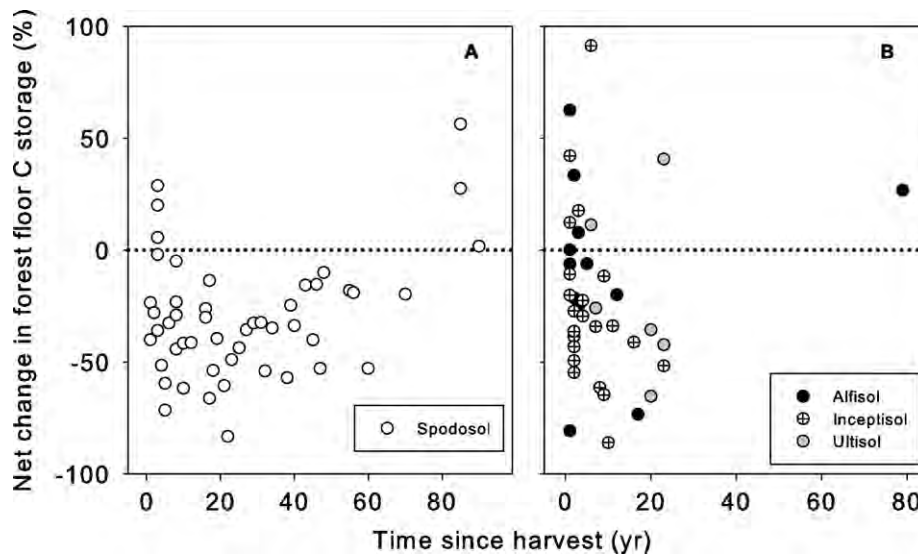
Fig. 3. Temporal patterns in forest floor C storage following harvest for Spodosols (A) and all other soil orders (B; Alfisols, Inceptisols, and Ultisols). Each point represents one response ratio. While the trajectory of post-harvest forest floor C storage is evident for Spodosols, more long-term data are needed to predict forest floor C recovery for other soil orders.

mineral soil response to harvest (Table 2). Overall, coniferous/ mixed forests showed no significant change in mineral soil C storage following harvest (+2%, NS) while hardwoods lost C (−9%). Studies from the southeast U.S. showed a significant reduction in mineral soil C (−7%), while those from other geographic groups exhibited no significant change.

3.3. Soil C budgets

Harvest impacts on forest floor and mineral soil C storage have different consequences for forest soil C budgets because of differences in the magnitude of C pools among the different soil layers (Fig. 5). While forest floors can lose a substantial proportion of their C stocks following harvest, the magnitude

Fig. 4. The effects of forest harvesting on mineral soil C storage, by soil taxonomic order. All points are mean effect sizes ± bootstrapped 95% confidence intervals, with the number of studies (k) in parentheses. The filled square at the top of the figure represents the overall harvest effect on mineral soil C storage, including surface, deep and whole mineral soils from all orders. Within each soil order, the effect of harvest on mineral soil C storage across all layers is represented by a square, while the circle and inverted triangle designate significantly different groups.

of these losses is tempered by the relatively small amount of C stored in the forest floor compared to the mineral soil in most soil orders. Among Alfisols, Inceptisols, and Ultisols, forest floor C storage in unharvested stands generally ranged from 5 to 20 Mg ha^{-1}, while mineral soils held 20–80 Mg C ha^{-1}. Spodosols were an exception to this general pattern, as forest floors and mineral soils contained a similar range of amounts of C (5–50 Mg ha^{-1}).

4. Discussion

4.1. Overall harvest effects and sources of between study variation

Our results show that, across studies, there is a significant effect of forest harvesting on soil C (−8 ± 3% overall, −13 ± 4% for C storage only). This statistically robust conclusion is in spite of the frequently high levels of spatial and temporal variability in forest soil C measurements, which often make it difficult to quantify the effects of management on soil and whole-ecosystem C budgets from a single study (Homann et al., 2001, 2008; Magrini et al., 2000). Fortunately, many factors responsible for variation in soil C responses to harvest, such as species composition and soil order, are typically recorded as meta-data within the experimental design.

Soil layer was the strongest predictor of soil C storage shifts due to harvest in the overall meta-analysis, despite variable sampling depths among studies. For example, forest floor material from some studies (e.g., Yanai et al., 2000, and references therein) included mineral soil, and the depth of the surface and deep mineral soil categories varied substantially across studies (5–20 and 25–100 cm, respectively). Nonetheless, forest floor C storage was significantly more vulnerable to decline following harvest than was mineral soil C storage. There may be several reasons for this difference in sensitivity to disturbance. First, there are significant differences in pool sizes, turnover times, and molecular characteristics of C stored at different depths in forest soils, which may cause the forest floor to be more responsive to disturbance or management than the mineral soil (Currie, 1999; Cromack et al., 1999; Dai et al., 2001; Trumbore, 2000). For example, the smaller C pool size of the forest floor means that even a modest C loss in absolute terms
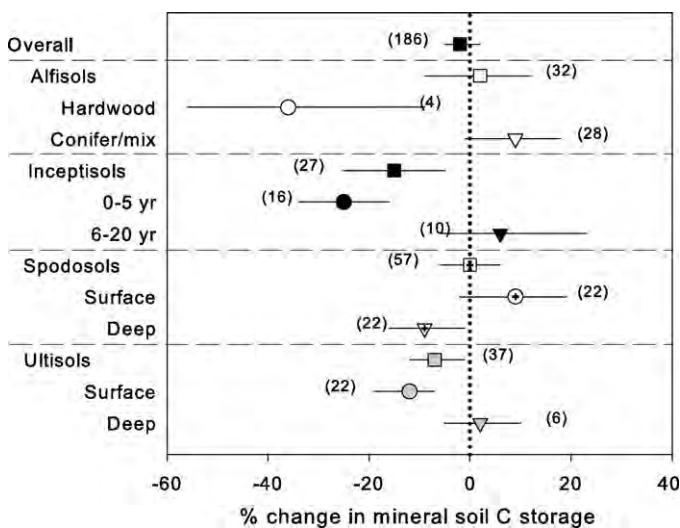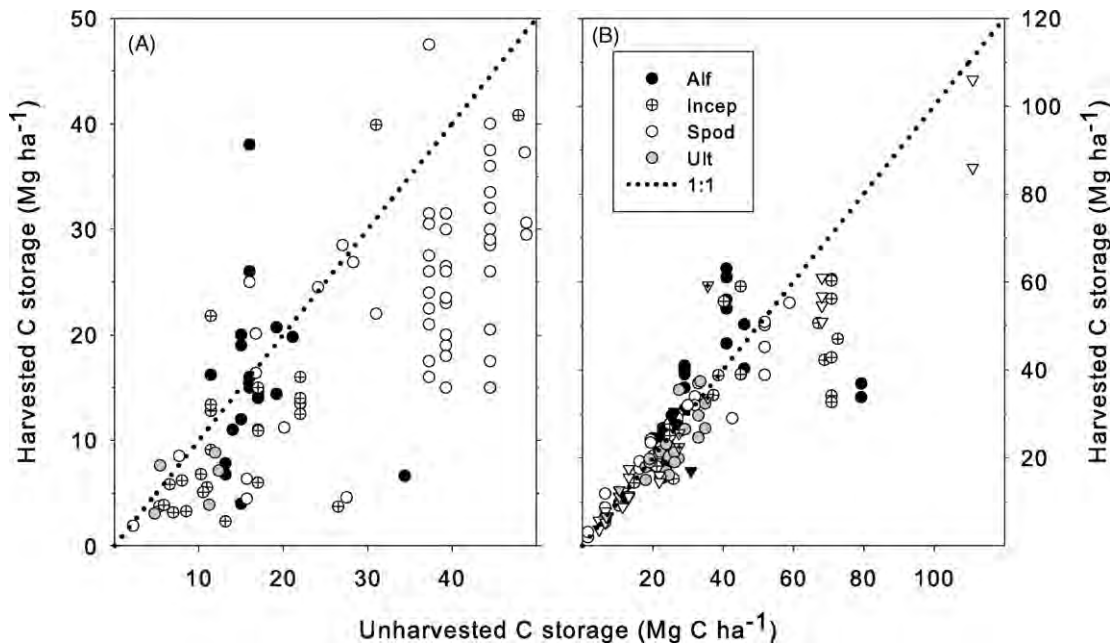
Fig. 5. Absolute changes in C storage due to harvest, for forest floors (A) and mineral soils (B), by soil taxonomic order. In panel B, surface mineral soils are represented by circles; deep mineral soils are triangles. Forest floor and soil C storage values for some points were estimated from loss on ignition data (C = 0.5 × LOI). The 1:1 reference line in each panel denotes no difference in C storage between unharvested and harvested stands; points below represent decreases, while points above are C storage increases. Points from unidentified or under-represented soil orders (Andisols, Entisols, Mollisols) are not plotted.

can cause a large proportional reduction, compared to the mineral soil. Forest soil C generally has longer turnover times and increasing molecular complexity with depth in the profile, and the abundance of labile organic matter in the forest floor may promote a more rapid microbial response following disturbance. The physical effects of harvesting on the forest floor, where machinery can directly disturb organic matter through mixing and fragmentation, also are different from those on the mineral soil, which is generally protected from the direct physical effects of harvesting. Forest floor C losses during harvest may be due to mixing and incorporation of surface organic matter into the upper mineral soil as suggested by several studies of whole-tree harvesting in northern hardwoods (Mroz et al., 1985; Ryan et al., 1992). However, our results indicate that this is generally not the case, since, in our meta-analysis, surface mineral soil C storage decreased significantly (−8%) in the time category immediately following harvesting (0–5 years).

4.2. Variation within soil layers

Although forest floors lose C after harvest regardless of species composition, the smaller C storage declines in forest floors from coniferous/mixed forests compared to hardwood forests may reflect the greater recalcitrance of coniferous residue. Generally, coniferous detritus and forest floor materials have higher C/N and lignin/N, slower decomposition and N-mineralization rates, and longer organic matter residence times than hardwood detritus/forest floor materials (Currie, 1999; Finzi et al., 1998; Silver and Miya, 2001). Although forest floor C losses were substantial, temporal trends suggest that these losses were not permanent: on Spodosols forest floors appear to recover after 50–70 years. This estimate supports the seminal study by Covington (1981), even when his data are removed from Fig. 3A (13 of 59 response ratios). It may be that forest floor C recovers more slowly in Spodosols than in other soil orders since Spodosols tend to have larger forest floor C pools, such that a similar proportional reduction in forest floor C

corresponds to a greater absolute amount of C in Spodosols than in other soil orders. In addition, productivity of forests growing on Spodosols generally is less than on Alfisols and Ultisols, and approximately equal to Inceptisols (Vogt et al., 1995). Therefore, Spodosols might also require a longer recovery period than Alfisols or Ultisols due to lower rates of litter inputs. To fill the knowledge gap that exists for most temperate forest soil taxonomic orders, there clearly is a need for additional long-term forest harvest–forest floor C studies. Chronosequences, such as those surveyed in Yanai et al. (2000) and Covington (1981), yield large amounts of data, but are a weaker experimental design than long-term monitoring of control and treatment stands. Namely, it may be impossible to distinguish whether forest floor C loss was due to changes in treatment over time or time since treatment based on chronosequence studies.

Harvest impacts on mineral soil C varied among soil orders, suggesting that order-specific properties or soil-forming factors mediate management effects on soil C storage. Within each soil order, a dominant soil forming process mediates the physical and chemical properties of that soil's horizons, including accumulation and distribution of soil C (Shaw et al., 2008). For example, Spodosols form through the process of podzolization, which occurs as soluble organic compounds are eluviated from forest floors and surface mineral soils, and illuviated at deeper depths in the mineral soil. Results from our meta-analysis suggest that this process may be responsible for the impacts of forest harvesting on Spodosol mineral soil C storage. While Spodosol surface mineral soils showed no changes following harvest, a significant increase in C *concentration* accompanied a significant decrease in C *storage* in the deep mineral soil. This suggests that a downward redistribution of soil C, perhaps due to accelerated podzolization, changed the organic matter chemistry of the deep mineral soil. Ussiri et al. (2007) reached a similar conclusion in their study of 15-year changes in soil organic matter in a paired-watershed clearcut experiment at Hubbard Brook. They used nuclear magnetic resonance to show that changes in organic matter composition accompanied the downward redistribution of soil C

after harvest, such that biogeochemically stable forms of organic matter were lost and replaced with less stable compounds. If shifting the balance of soil organic matter towards less stable compounds results in faster overall decomposition, this change in organic matter chemistry may explain why Spodosols lost significant amounts of deep mineral soil C stocks in our meta-analysis.

An additional factor accounting for differences among soil orders in their sensitivity to harvesting could be the specific management techniques most commonly practiced on them. For example, intensive site preparation (tillage) following harvest caused a significant decline in surface mineral soil C storage (−20%) but was practiced almost exclusively on Inceptisols and Ultisols (e.g., Carter et al., 2002; Merino and Edeso, 1999). Other post-harvest residue management and site preparation methods, such as broadcast burning or complete residue retention, did not reduce Inceptisol/Ultisol surface mineral C storage (e.g., Mattson and Smith, 1993; Shelburne et al., 2004). This suggests that C losses on these two soil orders may be mitigated, or even prevented, through the use of management practices that minimize physical disturbance to the soil profile.

Significant sources of variation in sensitivity to mineral soil C loss in Alfisols and Inceptisols also were good predictors of harvest impacts on mineral soil C storage of all orders. This is in contrast to Spodosols and Ultisols, which had controls on mineral soil C variation that were not present in the overall mineral soil analysis. Across all soil orders, and specifically within Alfisols, hardwoods lost mineral soil C and coniferous/mixed stands showed no change. This may reflect a general effect of differential residue quality, as hypothesized for forest floor C responses to harvest. Time since harvest affected the magnitude of mineral soil C losses on Inceptisols (and mineral soils in general, although it was not a significant predictor), which declined significantly 0–5 years post-harvest, but recovered after 6–20 years.

Although mineral soils across orders showed no significant harvest impact on surface or deep mineral soil C storage, Spodosols and Ultisols lost C from deep and surface layers, respectively. Thus, at what layer soil C was lost following harvest varied with soil order, perhaps due to differences in the dominant soil-forming processes among orders. Despite our large overall sample size, the relatively small number of studies conducted on specific soil orders precludes a full understanding of mechanisms responsible for variation within and between soil orders. Our meta-analysis has helped underscore the need to better describe relationships between soil taxonomy and variation in mineral soil C responses to harvest.

4.3. Soil C budgets

We found that forest floor C storage was more sensitive to harvest impacts than was mineral soil C storage, but the long-term implications of this differential sensitivity on forest productivity are difficult to predict. On one hand, the smaller amount of C stored in forest floors compared to mineral soils, and the shorter residence times of forest floor C pools suggests that C lost from the forest floor will be more readily replaced than C lost from the mineral soil. Forest floor C losses therefore may have only modest effects on total soil C storage, especially over long rotations or C accounting intervals. Alternatively, forest floor C reductions may have large impacts on forest productivity because forest floor organic matter plays important roles in nutrient cycling and water retention (Attiwill and Adams, 1993; Currie, 1999; Schaap et al., 1997; Tietema et al., 1992). Forest floor C losses could have a great impact on forest productivity when recovery times are multidecadal, as is the case on Spodosols. Due to their greater C pool sizes, changes in mineral

soil C are capable of causing greater changes to soil C budgets than losses from forest floors. However, since mineral soils showed no overall response to harvest, forest floors probably have a greater general effect on the soil C budgets of harvested forests. It is also worth noting that, among most studies we analyzed, residues such as coarse woody debris were not sampled as a component of the forest floor. Therefore, while forest floor C stocks did decline significantly, harvesting presumably increased the amount of C stored in woody debris pools, which promote nutrient and water retention and also have a significant impact on whole-ecosystem C budgets (Eisenbies et al., 2009; Janisch and Harmon, 2002).

An additional finding of this analysis related to soil C budgets pertains to the choice of units used for measuring and reporting soil C values. We found that soil C concentration and soil C storage responded differently to harvest in the overall analysis, or when examining forest floor and deep mineral soil layers individually. Measurements of soil C concentrations and soil C pool sizes are appropriate for different situations. For example, if microbial processes are the topic of study, then soil C concentrations may be relevant. However, if soil or whole-ecosystem C budgets are to be assessed, then soil C pool sizes are necessary. At the very least, bulk density links C concentration with C storage, and should be more widely reported in primary research articles focusing on all aspects of soil C. The significant difference between reporting units indicates that measurements of soil C concentrations are not adequate for soil C accounting purposes.

4.4. Conclusions

We analyzed 432 studies of soil C responses to harvest drawn from temperate forests around the world. We found a significant overall impact of harvesting on soil C storage, and determined that variation among harvest impacts was best explained by variation in species composition, soil taxonomic order, and time since harvest. One of the most important overall findings of this analysis was that C stored in forest floors is more vulnerable to harvest-induced loss (−30% on average) than mineral soil C (no significant change). Species composition (hardwood vs. coniferous/mixed) had a significant effect on forest floor C storage responses to harvest, with hardwoods generally losing more forest floor C than coniferous/mixed stands. Reductions in forest floor C storage probably have a greater impact on the soil C budgets of Spodosols than on other soil orders, since Spodosols store large amounts of C in forest floors relative to mineral soils, and require 50–70 years to recover lost forest floor C. Harvesting caused significant mineral soil C losses on Inceptisols and Ultisols, but not on Alfisols or Spodosols. Mineral soil C losses on Inceptisols were temporary, with C stocks recovering within 6–20 years after harvest. Ultisol mineral soil C losses were restricted to the surface mineral layer. The effects of species composition and soil taxonomic order on harvest-induced changes in forest floor and mineral soil C storage suggest that further research may allow development of predictive maps of forest management effects on soil C storage.

Appendix A

References providing response ratios for the present analysis. The full citation for each is denoted with a (*) in the references section.

Reference	Soil layers sampled	Dominant canopy genera	Locations
Alban and Perala (1992)	WM	*Populus*	MN, USA
Bauhus et al. (2004)	FF, SM, DM	*Fagus*	Germany
Black and Harden (1995)	FF, SM	*Abies, Calocedrus*	CA, USA
Boerner et al. (2006)	SM	*Pinus*	SC, USA
Borchers and Perry (1992)	SM	*Abies, Pseudotsuga*	OR, USA
Cade-Menun et al. (2000)	FF, SM	*Thuja, Tsuga*	BC, Canada
Carter et al. (2002)	SM	*Pinus*	LA, TX, USA
Cromack et al. (1999)	FF	*Pseudotsuga, Tsuga*	OR, USA
Dai et al. (2001)	FF, SM	*Fagus, Acer*	NH, USA
DeByle (1980)	SM	*Pinus*	WY, USA
DeLuca and Zouhar (2000)	SM	*Pinus*	MT, USA
Edmonds and McColl (1989)	SM	*Pinus*	Australia
Edwards and Ross-Todd (1983)	SM, DM	*Quercus, Liriodendron*	TN, USA
Elliott and Knoepp (2005)	SM	*Quercus*	NC, USA
Ellis and Graley (1983)	SM	*Eucalyptus*	Tasmania
Ellis et al. (1982)	SM	*Eucalyptus*	Tasmania
Esquilin et al. (2008)	SM	*Pinus*	CO, USA
Fraterrigo et al. (2005)	SM	*Liriodendron, Acer*	NC, USA
Frazer et al. (1990)	SM	*Pinus, Abies*	CA, USA
Gillon et al. (1999)	FF	*Pinus*	France
Goh and Phillips (1991)	FF	*Nothofagus*	New Zealand
Goodale and Aber (2001)	FF, SM	*Fagus, Acer*	NH, USA
Gough et al. (2007)	SM, DM	*Populus*	MI, USA
Grady and Hart (2006)	SM	*Pinus*	AZ, USA
Gresham (2002)	WM	*Pinus*	SC, USA
Griffiths and Swanson (2001)	SM	*Pseudotsuga*	OR, USA
Gundale et al. (2005)	FF, SM	*Pinus*	MT, USA
Hart et al. (2006)	FF, SM	*Pinus*	AZ, USA
Hendrickson et al. (1989)	FF, SM	*Pinus, Populus*	ON, Canada
Herman et al. (2003)	SM	*Quercus*	CA, USA
Holscher et al. (2001)	FF, SM	*Fagus, Betula*	Germany
Hwang and Son (2006)	WM	*Pinus, Larix*	Korea
Johnson (1995)	FF, SM, DM, WM	*Fagus, Acer*	NH, USA
Johnson and Todd (1998)	SM, DM	*Quercus, Liriodendron*	TN, USA
Johnson et al. (1991)	FF, SM	*Fagus, Acer*	NH, USA
Johnson et al. (1997)	FF, SM, DM	*Fagus, Acer*	NH, USA
Kaye and Hart (1998)	FF, SM	*Pinus*	AZ, USA
Keenan et al. (1994)	SM	*Thuja, Tsuga*	BC, Canada
Kelliher et al. (2004)	FF, SM, DM	*Pinus*	OR, USA
Klopatek (2002)	FF, SM	*Pseudotsuga, Tsuga*	WA, USA
Knoepp and Swank (1997)	SM, DM	*Quercus, Acer*	NC, USA
Korb et al. (2004)	SM	*Pinus*	AZ, USA
Kraemer and Hermann (1979)	SM	*Pseudotsuga*	WA, USA
Laiho et al. (2003)	SM, WM	*Pinus*	NC, LA, USA
Latty et al. (2004)	FF, SM	*Fagus, Acer*	NY, USA
Law et al. (2001)	SM, DM	*Pinus*	OR, USA
Law et al. (2003)	WM	*Pinus*	OR, USA
Leduc and Rothstein (2007)	FF + SM	*Pinus*	MI, USA
Maassen and Wirth (2004)	FF, SM	*Pinus*	Germany
Mattson and Smith (1993)	FF, SM	*Quercus, Acer*	WV, USA
Mattson and Swank (1989)	FF, SM, DM	*Quercus, Carya*	NC, USA
May and Attiwill (2003)	SM	*Eucalyptus*	Australia
McLaughlin and Phillips (2006)	FF, WM	*Picea, Abies*	ME, USA
McLaughlin et al. (1996)	FF, SM, DM	*Picea*	MI, USA
Merino and Edeso (1999)	SM	*Pinus*	Spain
Murphy et al. (2006)	FF, SM, DM	*Pinus, Abies*	CA, USA
Neher et al. (2003)	SM	*Pinus*	NC, USA
O'Brien et al. (2003)	WM	*Eucalyptus, Pinus*	Australia
Prietzel et al. (2004)	FF	*Pseudotsuga*	WA, USA
Rab (1996)	WM	*Eucalyptus*	Australia
Riley and Jones (2003)	SM	*Pinus*	SC, USA
Sanchez et al. (2007)	SM, DM	*Pinus*	SC, USA
Sanscrainte et al. (2003)	FF, WM	*Abies, Tsuga*	WA, USA
Selig et al. (2008)	SM, DM	*Pinus*	VA, USA
Shelburne et al. (2004)	FF, SM	*Pinus*	SC, USA
Skovsgaard et al. (2006)	FF, SM, DM	*Picea*	Denmark
Small and McCarthy (2005)	SM	*Quercus*	OH, USA
Stone and Eliof (1998)	FF, SM	*Populus*	MN, USA
Stone et al. (1999)	SM	*Pinus*	AZ, USA
Strong (1997)	SM, DM	*Acer, Tsuga*	MN, USA
Ussiri and Johnson (2007)	FF, SM, DM	*Fagus, Acer*	NH, USA
Vesterdal et al. (1995)	FF	*Picea*	Denmark
Waldrop et al. (2003)	FF	*Pinus, Libocedrus*	CA, USA
Yanai et al. (2000)	FF	*Fagus, Betula*	NH, USA
Zhong and Makeschin (2003)	FF, SM	*Picea*	Germany

Soil layer abbreviations: FF, forest floor; SM, surface mineral; DM, deep mineral; WM, whole mineral.

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Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems

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Abstract. Two forest management objectives being debated in the context of federally managed landscapes in the U.S. Pacific Northwest involve a perceived trade-off between fire restoration and carbon sequestration. The former strategy would reduce fuel (and therefore C) that has accumulated through a century of fire suppression and exclusion which has led to extreme fire risk in some areas. The latter strategy would manage forests for enhanced C sequestration as a method of reducing atmospheric CO_2 and associated threats from global climate change. We explored the trade-off between these two strategies by employing a forest ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on fire severity and the resulting long-term C dynamics among three Pacific Northwest ecosystems: the east Cascades ponderosa pine forests, the west Cascades western hemlock–Douglas-fir forests, and the Coast Range western hemlock–Sitka spruce forests. Our simulations indicate that fuel reduction treatments in these ecosystems consistently reduced fire severity. However, reducing the fraction by which C is lost in a wildfire requires the removal of a much greater amount of C, since most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, all of the fuel reduction treatments simulated for the west Cascades and Coast Range ecosystems as well as most of the treatments simulated for the east Cascades resulted in a reduced mean stand C storage. One suggested method of compensating for such losses in C storage is to utilize C harvested in fuel reduction treatments as biofuels. Our analysis indicates that this will not be an effective strategy in the west Cascades and Coast Range over the next 100 years. We suggest that forest management plans aimed solely at ameliorating increases in atmospheric CO_2 should forgo fuel reduction treatments in these ecosystems, with the possible exception of some east Cascades ponderosa pine stands with uncharacteristic levels of understory fuel accumulation. Balancing a demand for maximal landscape C storage with the demand for reduced wildfire severity will likely require treatments to be applied strategically throughout the landscape rather than indiscriminately treating all stands.

Key words: biofuels; carbon sequestration; fire ecology; fuel reduction treatment; Pacific Northwest, USA; Picea sitchensis; Pinus ponderosa; Pseudotsuga menziesii.

INTRODUCTION

Forests of the U.S. Pacific Northwest capture and store large amounts of atmospheric CO_2, and thus help mitigate the continuing climatic changes that result from extensive combustion of fossil fuels. However, wildfire is an integral component to these ecosystems and releases a substantial amount of CO_2 back to the atmosphere via biomass combustion. Some ecosystems have experienced an increase in the amount of CO_2 released due to a century-long policy of fire suppression that has led to increased levels of fuel buildup, resulting in wildfires of uncharacteristic severity. Fuel reduction treatments have been proposed to reduce wildfire severity, but like wildfire, these treatments also reduce the C stored in forests. Our work examines the effects of fuel reduction

on wildfire severity and long-term C storage to gauge the strength of the potential trade-off between managing forests for increased C storage and reduced wildfire severity.

Forests have long been referenced as a potential sink for atmospheric CO_2 (Vitousek 1991, Turner et al. 1995, Harmon et al. 1996, Harmon 2001, Smithwick et al. 2002, Pacala and Socolow 2004), and are credited with contributing to much of the current C sink in the coterminous United States (Pacala et al. 2001, Hurtt et al. 2002). This U.S. carbon sink has been estimated to be between 0.30 and 0.58 Pg C/yr for the 1980s, of which between 0.17 Pg C/yr and 0.37 Pg C/yr has been attributed to accumulation by forest ecosystems (Pacala et al. 2001). While the presence of such a large sink has been valuable in mitigating global climate change, a substantial portion of it is due to the development of understory vegetation as a result of a national policy of fire suppression (Pacala et al. 2001, Donovan and Brown 2007). Fire suppression, while capable of incurring

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short-term climate change mitigation benefits by promoting the capture and storage of atmospheric CO_2 by understory vegetation and dead fuels (Houghton et al. 2000, Tilman et al. 2000), has, in part, led to increased and often extreme fire risk in some forests, notably *Pinus ponderosa* forests (Moeur et al. 2005, Donovan and Brown 2007).

Increased C storage usually results in an increased amount of C lost in a wildfire (Fahnestock and Agee 1983, Agee 1993). Many ecosystems show the effects of fire suppression (Schimel et al. 2001, Goodale et al. 2002, Taylor and Skinner 2003), and the potential effects of additional C storage on the severity of future wildfires is substantial. In the *Pinus ponderosa* forests of the east Cascades, for example, understory fuel development is thought to have propagated crown fires that have killed old-growth stands not normally subject to fires of high intensity (Moeur et al. 2005). Various fuel reduction treatments have been recommended for risk-prone forests, particularly a reduction in understory vegetation density, which can reduce the ladder fuels that promote such severe fires (Agee 2002, Brown et al. 2004, Agee and Skinner 2005). While a properly executed reduction in fuels could be successful in reducing forest fire severity and extent, such a treatment may be counterproductive to attempts at utilizing forests for the purpose of long-term C sequestration.

Pacific Northwest forests, particularly those that are on the west side of the Cascade mountain range, are adept at storing large amounts of C. Native long-lived conifers are able to maintain production during the rainy fall and winter months, thereby out-competing shorter-lived deciduous angiosperms with a lower biomass storage capacity (Waring and Franklin 1979). Total C storage potential, or upper bounds, of these ecosystems is estimated to be as high as 829.4 Mg C/ha and 1127.0 Mg C/ha for the western Cascades and Coast Range of Oregon, respectively (Smithwick et al. 2002). Of this high storage capacity for west Cascades and Coast Range forests, 432.8 Mg C/ha and 466.3 Mg C/ha, respectively, are stored in aboveground biomass (Smithwick et al. 2002), a substantial amount of fuel for wildfires.

High amounts of wildfire-caused C loss often reflect high amounts of forest fuel availability prior to the onset of fire. Given the magnitude of such losses, it is clear that the effect of wildfire severity on long-term C dynamics is central to our understanding of the global C cycle. What is not clear is the extent to which repeated fuel removals that are intended to reduce wildfire severity will likewise reduce long-term total ecosystem C storage (TECμ). Fuel reduction treatments require the removal of woody and detrital materials to reduce future wildfire severity. Such treatments can be effective in reducing future wildfire severity, but they likewise involve a reduction in stand-level C storage. If repeated fuel reduction treatments decrease the mean total ecosystem C storage by a quantity that is greater than

the difference between the wildfire-caused C loss in an untreated stand and the wildfire-caused C loss in a treated stand, the ecosystem will not have been effectively managed for maximal long-term C storage.

Our goal was to test the extent to which a reduction in forest fuels will affect fire severity and long-term C storage by employing a test of such dynamics at multi-century time scales. Our questions were as follows: (1) To what degree will reductions in fuel load result in decreases in C stores at the stand level? (2) How much C must be removed to make a significant reduction in the amount of C lost in a wildfire? (3) Can forests be managed for both a reduction in fire severity and increased C sequestration, or are these goals mutually exclusive?

METHODS

Model description

We conducted our study using an ecosystem simulation model, STANDCARB (Appendix A), that allows for the integration of many forest management practices as well as the ensuing gap dynamics that may result from such practices. STANDCARB is a forest ecosystem simulation model that acts as a hybrid between traditional single-life-form ecosystem models and multi-life-form gap models (Harmon and Marks 2002). The model integrates climate-driven growth and decomposition processes with species-specific rates of senescence and stochastic mortality while incorporating the dynamics of inter- and intraspecific competition that characterize forest gap dynamics. Inter- and intraspecific competition dynamics are accounted for by modeling species-specific responses to solar radiation as a function of each species' light compensation point as well as the amount of solar radiation delineated through the forest canopy to each individual. By incorporating these processes the model can simulate successional changes in population structure and community composition without neglecting the associated changes in ecosystem processes that result from species-specific rates of growth, senescence, mortality, and decomposition.

STANDCARB performs calculations on a monthly time step and can operate at a range of spatial scales by allowing a multi-cell grid to capture multiple spatial extents, as both the size of an individual cell and the number of cells in a given grid can be designated by the user. We used a 20×20 cell matrix for all simulations (400 cells total), with 15×15 m cells for forests of the west Cascades and Coast Range and 12×12 m cells for forests of the east Cascades. Each cell allows for interactions of four distinct vegetation layers, represented as upper canopy trees, lower canopy trees, a species-nonspecific shrub layer, and a species-nonspecific herb layer. Each respective vegetation layer can have up to seven live pools, eight detrital pools, and three stable C pools. For example, the upper and lower tree layers comprise seven live pools: foliage, fine roots, branches, sapwood, heartwood, coarse roots, and heart-rot, all of

which are transferred to a detrital pool following mortality. Dead wood is separated into snags and logs to capture the effects of spatial position on microclimate. After detrital materials have undergone significant decomposition, they can contribute material to three increasingly decay-resistant, stable C pools: stable foliage, stable wood, and stable soil. Charcoal is created in both prescribed fires and wildfires and is thereafter placed in a separate pool with high decay resistance. Additional details on the STANDCARB model can be found in Appendix A.

Fire processes

We generated exponential random variables to assign the years of fire occurrence (sensu Van Wagner 1978) based on the literature estimates (see experimental design for citations) of mean fire return intervals (MFRI) for different regions in the U.S. Pacific Northwest. The cumulative distribution for our negative exponential function is given in Eq. 1 where X is a continuous random variable defined for all possible numbers x in the probability function P, and λ represents the inverse of the expected time $E[X]$ for a fire return interval given in Eq. 2:

$$P\{X \leq x\} = \int_0^x \lambda e^{-\lambda x} dx \qquad (1)$$

where

$$E[X] = \frac{1}{\lambda}. \qquad (2)$$

Fire severities in each year generated by this function are cell specific, as each cell is assigned a weighted fuel index calculated from fuel accumulation within that cell and the respective flammability of each fuel component, the latter of which is derived from estimates of wildfire-caused biomass consumption (see Fahnestock and Agee 1983, Covington and Sackett 1984, Agee 1993). Fires can increase (or decrease) in severity depending on how much the weighted fuel index of a given cell exceeds (or falls short of) the fuel level thresholds for each fire severity class (T_{light}, T_{medium}, T_{high}, and T_{max}), and the probability values for the increase or decrease in fire severity (P_i and P_d). For example, while the natural fire severity of many stands of the west Cascades can be described as high severity, other stands of the west Cascades have a natural fire severity that can be best described as being of medium severity (~60–80% overstory tree mortality) (Cissel et al. 1999). For these stands, medium-severity fires are scheduled to occur throughout the simulated stand and can increase to a high-severity fire depending on the extent to which the weighted fuel index in a cell exceeds the threshold for a high-severity fire, as greater differences between the fuel index and the fire severity threshold will increase the chance of a change in fire severity. Conversely, medium-severity fires may decrease to a low-severity fire if the fuel index is sufficiently below the threshold for a medium-severity fire. High-severity fires are likely to become medium-severity fires if the weighted fuel index within a given cell falls sufficiently short of the threshold for a high-severity fire, and low-severity fires are likely to become medium severity if the weighted fuel index in a given cell is sufficiently greater than the threshold for a medium-severity fire. Fuel level thresholds were set by monitoring fuel levels in a large series of simulation runs where fires were set at very short intervals to see how low fuel levels needed to be to create a significant decrease in expected fire severity. We note that, like fuel accumulation, the role of regional climate exerts significant influence on fire frequency and severity, and that our model does not attempt to directly model these effects. We suspect that an attempt to model the highly complex role of regional climate data on fine-scale fuel moisture, lightning-based fuel ignition, and wind-driven fire spread adds uncertainties into our model that might undermine the precision and applicability of our modeling exercise. For that reason we incorporated data from extensive fire history studies to approximate the dynamics of fire frequency and severity.

Final calculations for the expected stand fire severity $E[F_s]$ at each fire are performed as follows:

$$E[F_s] = \frac{100}{C} \sum_{i=1}^{n} c_{i(L)} m_{i(L)} + c_{i(M)} m_{i(M)} + c_{i(H)} m_{i(H)} \qquad (3)$$

where C is the number of cells in the stand matrix and $c_{i(L)}$, $c_{i(M)}$, and $c_{i(H)}$ are the number of cells with light, medium, and high-severity fires, and $m_{i(L)}$, $m_{i(M)}$, and $m_{i(H)}$ represent fixed mortality percentages for canopy tree species for light, medium, and high-severity fires, respectively. This calculation provides an approximation of the number of upper-canopy trees killed in the fire. The resulting expected fire severity calculation $E[F_s]$ is represented on a scale from 0 to 100, where a severity index of 100 indicates that all trees in the simulated stand were killed.

Our approach at modeling the effectiveness of fuel reduction treatments underscores an important trade-off between fuel reduction and long-term ecosystem C storage by incorporating the dynamics of snag creation and decomposition. Repeated fuel reduction treatments may result in a reduction in long-term C storage, but it is possible that if such treatments are effective in reducing tree mortality, they may also offset some of the C losses that would be incurred from the decomposition of snags that would be created in a wildfire of higher severity. STANDCARB accounts for these dynamics by directly linking expected fire severity with a fuel accumulation index that can be altered by fuel reduction treatments while also incorporating the decomposition of snags as well as the time required for each snag to fall following mortality.

Total ecosystem C storage (TEC) is calculated by summing all components of C (live, dead, and stable). For each replicate ($i = 1, 2, \ldots 5$) and for each period
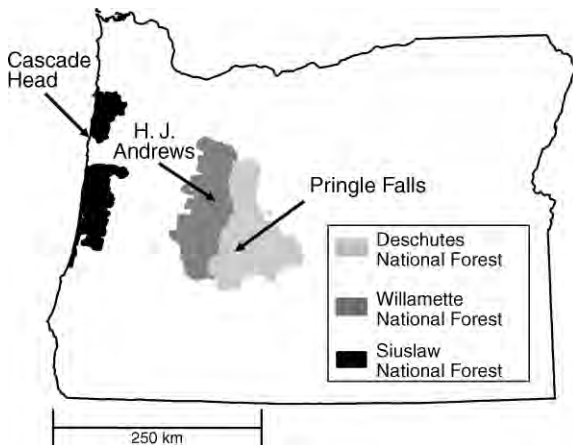
FIG. 1. Site locations in Oregon. Pringle Falls is our representative site for the east Cascades, H. J. Andrews is our representative site for the west Cascades, and Cascade Head is our representative site for the Coast Range.

between fires ($x = 1, 2, \ldots P_i$), the mean total ecosystem C storage (TEC_μ) is calculated by averaging the yearly TEC values ($k = 1, 2, \ldots R_x$).

$$\text{TEC}_{\mu(i,x)} = \frac{1}{R} \sum_{k=1}^{R} \text{TEC}_{(i,x,k)}.$$

Aggregating TEC_μ values in this manner permits the number of TEC_μ values to be the same as the number of $E[F_s]$ values, permitting a PerMANOVA analysis to be performed on $E[F_s]$ and TEC_μ.

Fuel reduction processes

STANDCARB's fire module allows for scheduled prescribed fires of a given severity (light, medium, high) to be simulated in addition to the nonscheduled wildfires generated from the aforementioned exponential random variable function. In addition to simulating the prescribed fire method of fuel reduction, STANDCARB has a harvest module that permits cell-by-cell harvest of trees in either the upper or lower canopy. This module allows the user to simulate understory removal or overstory thinning treatments on a cell-by-cell basis. Harvested materials can be left in the cell as detritus following cutting or can be removed from the forest, allowing the user to incorporate the residual biomass that results from harvesting practices. STANDCARB can also simulate the harvest of dead salvageable materials such as logs or snags that have not decomposed beyond the point of being salvageable.

Site descriptions

We chose the *Pinus ponderosa* stands of the Pringle Falls Experimental Forest as our representative for east Cascades forests (Youngblood et al. 2004). Topography in the east Cascades consists of gentle slopes, with soils derived from aerially deposited dacite pumice. The *Tsuga heterophylla–Pseudotsuga menziesii* stands of the

H. J. Andrews Experimental Forest were chosen as our representative of west Cascades forests (Greenland 1994). Topography in the west Cascades consists of slope gradients that range from 20% to 60% with soils that are deep, well-drained dystrochrepts. The *Tsuga heterophylla–Picea sitchensis* stands of the Cascade Head Experimental Forest were chosen as our representative of Coast Range forests. We note that most of the Oregon Coast Range is actually composed of *Tsuga heterophylla–Pseudotsuga menziesii* community types, similar to much of the west Cascades. *Tsuga heterophylla–Picea sitchensis* communities occupy a narrow strip near the coast, due to their higher tolerance for salt spray, higher soil moisture optimum, and lower tolerance for drought compared to forests dominated by *Pseudotsuga menziesii* (Minore 1979), and we incorporate this region in order to gain insight into this highly productive ecosystem. Topography in the Cascade Head Experimental Forest consists of slope gradients of ~10% with soils that are silt loams to silt clay loams derived from marine siltstones. Site locations are shown in Fig. 1 and are located within three of the physiographic regions of Oregon and Washington as designated by Franklin and Dyrness (1988). Additional site data are shown in Table 1.

Experimental design

The effectiveness of forest fuel reduction treatments is often, if not always, inversely related to the time since their implementation. For this reason, our experiment incorporated a factorial blocking design where each ecosystem was subjected to four different frequencies of each fuel reduction treatment. We also recognize the fact that fire return intervals can exhibit substantial variation within a single watershed, particularly those with a high degree of topographic complexity (Agee 1993, Cissel et al. 1999), so we examined two likely fire regimes for each ecosystem. Historic fire return intervals may become unreliable predictors of future fire intervals (Westerling et al. 2006); thus ascertaining the differences in TEC_μ that result from two fire regimes might be a useful metric in gauging C dynamics resulting from fire regimes that may be further altered as a result of continued global climate change.

We based the expected fire return time in Eqs. 1 and 2 on historical fire data for our forests based on the following studies. Bork (1985) estimated a mean fire return interval of 16 years for the east Cascades *Pinus ponderosa* forests, and we also considered a mean fire return interval of 8 years for this system. Cissel et al. (1999) reported mean fire return intervals of 143 and 231 years for forests of medium- and high-severity (stand-replacing) fire regimes, respectively, among the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades. Less is known about the fire history of the Coast Range, which consists of *Tsuga heterophylla–Pseudotsuga menziesii* communities in the interior and *Tsuga heterophylla–Picea sitchensis* communities occu-

TABLE 1. Site characteristics (from Smithwick et al. 2002).

Site characteristic	Pringle Falls	H. J. Andrews	Cascade Head
Vegetation	PIPO	TSHE–PSME	TSHE–PISI
Elevation (m)	1359	785	287
Mean annual temperature (°C)	5.5	8.4	8.6
Mean annual precipitation (mm)	544	2001	2536
Soil porosity	sandy loam	loam	loam
Mean C storage potential (Mg C/ha)	183	829	1127

Note: Species codes: PIPO, *Pinus ponderosa*; TSHE, *Tsuga heterophylla*; PSME, *Pseudotsuga menziesii*; PISI, *Picea sitchensis*.

pying a narrow edge of land along the Oregon Coast. Work by Impara (1997) in the interior region of the Coast Range suggested a natural fire return interval (expected fire return time) of 271 years in the *Tsuga heterophylla–Pseudotsuga menziesii* zone, and Long et al. (1998) reported lake-derived charcoal sediment-based estimates of mean fire return interval for the Coast Range forests to be fairly similar, at 230 years. However, the *Tsuga heterophylla–Picea sitchensis* community type dominant in our study area of the Cascade Head Experimental Forest has little resistance to fire, and thus rarely provides a dendrochronological record. We estimated a mean fire return interval of 250 years as one fire return interval for a high-severity fire, derived from interior Coast Range natural fire return interval estimates, and also included another high-severity fire regime with a 500-year mean fire return interval in our analysis.

It is important to note that while the forests of the east Cascades exhibit a significant and visible legacy of effects from a policy of fire suppression, many of the mean fire return intervals for the forests of the west Cascades and Coast Range exceed the period of fire suppression (~100 years), and these forests in the west Cascades and Coast Range will not necessarily exhibit uncharacteristic levels of fuel accumulation (Brown et al. 2004). However, the potential lack of an uncharacteristic amount of fuel accumulation does not necessarily preclude these forests from future fuel reduction treatments or harvesting; thus we have included these possibilities in our analysis. The frequencies at which fuel reduction treatments are applied were designed to be reflective of literature-derived estimates of each ecosystem's mean fire return intervals, since forest management agencies are urged to perform fuel reduction treatments at a frequency reflective of the fire regimes and ecosystem-specific fuel levels (Franklin and Agee 2003, Dellasala et al. 2004). Treatment frequencies for the Coast Range and west Cascades were 100, 50, 25 years, plus an untreated control group, while treatment frequencies in the east Cascades were 25, 10, and 5 years, and an untreated control group.

We incorporated six different types of fuel reduction treatments largely based on those outlined in Agee (2002), Hessburg and Agee (2003), and Agee and Skinner (2005). Treatments 2–5 were taken directly from the authors' recommendations in these publications, treatment 1 was derived from the same principles

used to formulate those recommendations, and treatment 6, clear-cutting, was not recommended in these publications but was incorporated into our analysis because it is a common practice in many Pacific Northwest forests. Treatments 1–4 were applied to all ecosystems, while treatments 5 and 6 were applied only to the west Cascades and Coast Range forests, as such treatments would be unrealistic at the treatment intervals necessary to reduce fire severity in the high-frequency fire regimes of the east Cascades *Pinus ponderosa* forests. Note that these treatments and combinations thereof are not necessarily utilized in each and every ecosystem. Managers of forests on the Oregon Coast, for example, would be unlikely to use prescribed fire as a fuel reduction technique. Our experimental design simply represents the range of all possible treatments that can be utilized for fuel reduction and is applied to all ecosystems purely for the sake of consistency.

1. *Salvage logging (SL).*—The removal of large woody surface fuels limits the flame length of a wildfire that might enter the stand. Our method of ground fuel reduction entailed a removal of 75% of salvageable large woody materials in the stand. Our definition of salvage logging includes both standing and downed salvageable materials (sensu Lindenmayer and Noss 2006).

2. *Understory removal (UR).*—Increasing the distance from surface fuels to flammable crown fuels will reduce the probability of canopy ignition. This objective can be accomplished through pruning, prescribed fire, or the removal of small trees. We simulated this treatment in STANDCARB by removing lower canopy trees in all cells.

3. *Prescribed fire (PF).*—The reduction of surface fuels limits the flame length of a wildfire that might enter the stand. In the field, this is done by removing fuel through prescribed fire or pile burning, both of which reduce the potential magnitude of a wildfire by making it more difficult for a surface fire to ignite the canopy (Scott and Reinhardt 2001). We implemented this treatment in STANDCARB by simulating a prescribed fire at low severity for all cells.

4. *Understory removal and prescribed fire (UR + PF).*—This treatment is a combination of treatments 2 and 3, where lower canopy trees were removed (treatment 2) before a prescribed fire (treatment 3) the following year for all cells.

5. *Understory removal, overstory thinning, and prescribed fire (UR + OT + PF).*—A reduction in crown density by thinning overstory trees can make crown fire spread less probable (Agee and Skinner 2005) and can reduce potential fuels by decreasing the amount of biomass available for accumulation on the forest floor. Some have suggested that such a treatment will be effective only if used in conjunction with UR and PF (Perry et al. 2004). We simulated this treatment in STANDCARB by removing all lower canopy trees (treatment 2), removing upper canopy trees in 50% of the cells, and then setting a prescribed fire (treatment 3) the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

6. *Understory removal, overstory removal, and prescribed fire (clear-cutting) (UR + OR + PF).*—Clear-cutting is a common silvicultural practice in the forests of the Pacific Northwest, notably on private lands in the Oregon Coast Range (Hobbs et al. 2002), and we included it in our analysis for two ecosystems (west Cascades and Coast Range) simply to gain insight into the effects of this practice on long-term C storage and wildfire severity. We simulated clear-cutting in STANDCARB by removing all upper and lower canopy trees, followed by a prescribed burn the following year. This treatment was excluded from the east Cascades forests because it would be unrealistic to apply it at intervals commensurate with the high-frequency fires endemic to that ecosystem.

7. *Control group.*—Control groups had no treatments performed on them. The only disturbances in these simulations were the same wildfires that occurred in every other simulation with the same MFRI.

In sum, our east Cascades analysis tested the effects of four fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 8 years, MFRI = 16 years). Our analysis of west Cascades and Coast Range forests tested the effects of six fuel reduction treatment types, four treatment frequencies, including one control group, and two site mean fire return intervals (MFRI = 143 years, MFRI = 230 years for the west Cascades, MFRI = 250 years, MFRI = 500 years for the Coast Range) on expected fire severity and long-term C dynamics. This design resulted in 32 combinations of treatment types for the east Cascades and 48 combinations of treatment types and frequencies for each fire regime in the west Cascades and Coast Range, with each treatment combination in each ecosystem replicated five times.

Biofuel considerations

Future increases in the efficiency of producing biofuels from woody materials may reduce potential trade-offs between managing forests for increased C storage and reduced wildfire severity. Much research is currently underway in the area of lignocellulase-based (as opposed to sugar- or corn-based) biofuels (Schubert 2006). If this area of research yields efficient methods of utilizing woody materials directly as an energy source or indirectly by converting them into biofuels such as ethanol, fuels removed from the forest could be utilized as an energy source and thus act as a substitute for fossil fuels by adding only atmosphere-derived CO_2 back to the atmosphere. However, the conversion of removed forest biomass into biofuels will only be a useful method of offsetting fossil fuel emissions if the amount of C stored in an unmanaged forest is less than the sum of managed stand TEC_μ, and the amount of fossil fuel emissions averted by converting removed forest biomass from a stand of identical size into biofuels over the time period considered. We performed an analysis on the extent to which fossil fuel CO_2 emissions can be avoided if we were to use harvested biomass directly for fuel or indirectly for ethanol production. We recognize that many variables need to be considered when calculating the conversion efficiencies of biomass to biofuels, such as the amount of energy required to harvest the materials, inefficiencies in the industrial conversion process, and the differences in efficiencies of various energy sources that exist even after differences in potential energy are accounted for. Rather than attempt to predict the energy expended to harvest the materials, the future of the efficiency of the industrial conversion process, and differences in energy efficiencies, we simply estimated the maximum possible conversion efficiency that can be achieved, given the energy content of these materials. The following procedure was used to estimate the extent to which fossil fuel CO_2 emissions can be avoided by substituting harvested biofuels as an energy source:

1) Estimate the mean annual biomass removal that results from intensive fuel reduction treatments.

2) Calculate the ratio of the amount of potential energy per unit C emissions for biofuels (both woody and ethanol) to the amount of energy per unit C emissions for fossil fuels.

3) Multiply the potential energy ratios by the mean annual quantity of biomass harvested to calculate the mean annual C offset by each biofuel type for each forest.

4) Calculate the number of years necessary for biofuels production to result in an offset of fossil fuel C emissions. This procedure was performed for two land-use histories: managed second-growth forests, and old-growth forests converted to managed second-growth forests.

Calculations for each ecosystem are shown in Appendix B.

Simulation spin-up

STANDCARB was calibrated to standardized silvicultural volume tables for Pacific Northwest stands. We then calibrated it to permanent study plot data from three experimental forests in the region (Fig. 1) to

TABLE 2. Treatment abbreviations.

Treatment abbreviation	Treatment
SL	salvage logging
UR	understory tree removal
PF	prescribed fire
UR + PF	understory tree removal + prescribed fire
UR + PF + OT	understory removal + prescribed fire + overstory thinning
UR + PF + OR	understory removal + prescribed fire + overstory removal

incorporate fuel legacies, which were taken from a 600-year spin-up simulation with fire occurrences generated from the exponential distribution in Eq. 1, where λ was based on each ecosystem's mean fire return interval. Spin-up simulations were run prior to the initiation of each series of fuel reduction treatments, and simulations were run for a total of 800 years for forests of the east Cascades, and a total of 1500 years for simulations of the west Cascades and Coast Range.

Data analysis

We employed a nonparametric multivariate analysis of variance, PerMANOVA (Anderson 2001), to test group-level differences in the effects of fuel reduction frequency and type on mean total ecosystem C storage and expected fire severity. PerMANOVA employs a test statistic for the F ratio that is similar to that of an ANOVA calculated using sum of squares, but unlike an ANOVA, PerMANOVA calculates sums of squares from distances among data points rather than from differences from the mean. PerMANOVA was used instead of a standard MANOVA because it was highly unlikely that our data would meet the assumptions of a parametric MANOVA. PerMANOVA analysis treated fuel reduction treatment type and treatment frequency as fixed factors within each respective fire regime for each ecosystem simulated. The null hypothesis of no treatment effect for different combinations of these factors on TEC_μ and $E[F_s]$ was tested by permuting the data into randomly assigned sample units for each combination of factors so that the number of replicates within each factor combination were fixed. Each of our 12 PerMANOVA tests incorporated 10 000 permutations using a Euclidian distance metric, and multiple pairwise comparison testing for differences among treatment types and treatment frequencies was performed when significant differences were detected (i.e., $P < 0.05$).

Results

Results of the PerMANOVA tests indicate that mean expected fire severity ($E[F_s]$) and mean total ecosystem C storage (TEC_μ) were significantly affected by fuel reduction type ($P < 0.0001$), frequency ($P < 0.0001$), and interactions between type and frequency ($P < 0.0001$) in all three ecosystems. These results were significant for type, frequency, and interaction effects even when clear-cutting was excluded from the analysis for the west Cascades and Coast Range simulations, just

as it was a priori for simulations of the east Cascades. When the PerMANOVA was performed on only one of our response variables ($E[F_s]$ or TEC_μ), groupwise comparisons of effects of treatment type showed that the most significant effects of treatment and frequency were related to TEC_μ. TEC_μ was strongly affected by treatment frequency for each fire regime in each ecosystem ($P < 0.0001$) and consistently showed an inverse relationship to the quantity of C removed in a given fuel reduction treatment, and was thus highly related to treatment type. $E[F_s]$, similar to TEC_μ, showed significant relationships with treatment frequency for all three ecosystems ($P < 0.0001$), with statistically significant differences among most treatment types. Boxplots of TEC_μ and $E[F_s]$ for each treatment type in each fire regime for each ecosystem are shown in Appendix C.

Fuel reduction treatments in east Cascades simulations reduced TEC_μ with the exception of one treatment type; UR treatments (see Table 2 for acronym descriptions) in these systems occasionally resulted in additional C storage compared to the control group. These differences were very small (0.6–1.2% increase in TEC_μ) but statistically significant (Student's paired t test, $P < 0.05$) for the treatment return interval of 10 years in the light fire severity regime No. 1 (MFRI = 8 years) and for all treatment return intervals in light fire severity regime No. 2 (MFRI = 16 years). The fuel reduction treatment that reduced TEC_μ the least was SL, which, depending on treatment frequency and fire regime, stored between 93% and 98% of the control group, indicating that there was little salvageable material. UR + PF, depending on treatment frequency and fire regime, resulted in the largest reduction of TEC_μ in east Cascades forests, storing between 69% and 93% of the control group.

Simulations of west Cascades and Coast Range forests showed a decrease in C storage for all treatment types and frequencies. Fuel reduction treatments with the smallest effect on TEC_μ were either SL or UR, which were nearly the same in effect. The treatment that most reduced TEC_μ was UR + OT + PF. Depending on treatment frequency and fire regime, this treatment resulted in C storage of between 50% and 82% of the control group for the west Cascades, and between 65% and 88% of the control group for the Coast Range. Simulations with clear-cutting (UR + OR + PF), depending on application frequency and fire regime, resulted in C storage that was between 22% and 58% of
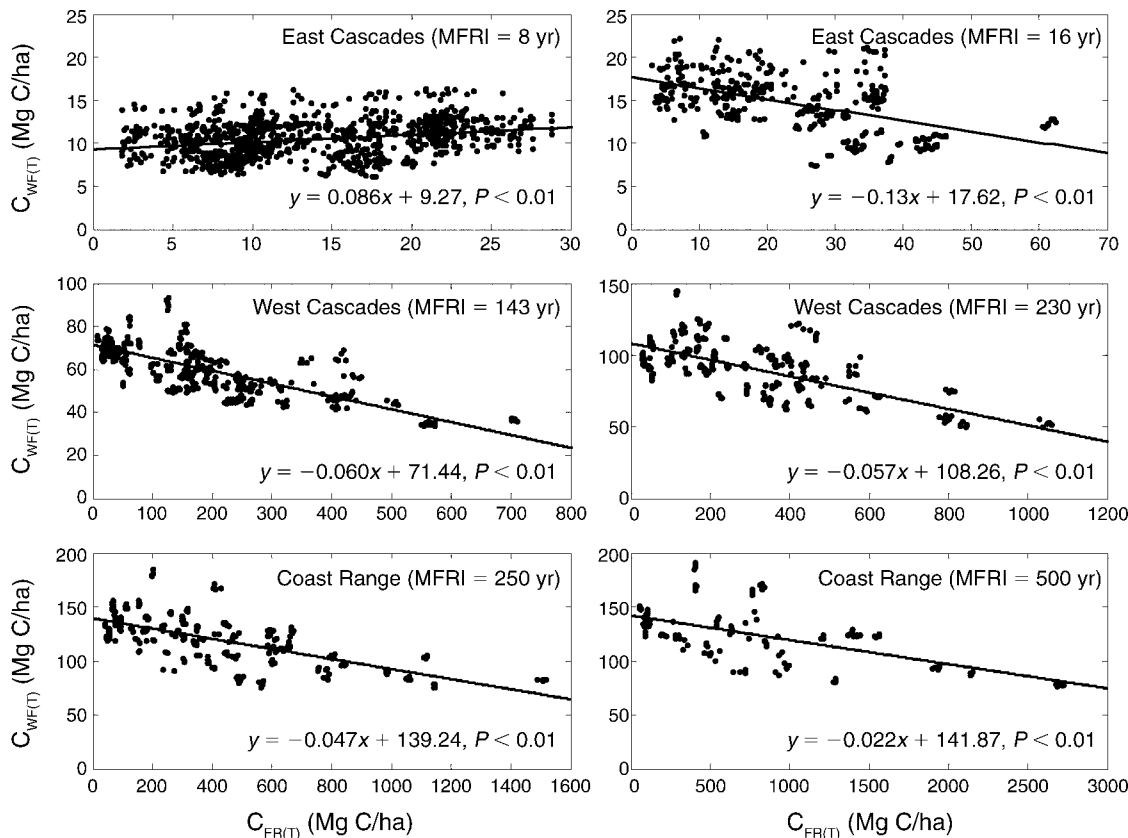
FIG. 2. Scatterplots of C removed in fuel reduction treatments between wildfires $C_{FR(T)}$ (representing fuel reduction [treatment]) and C lost in wildfires $C_{WF(T)}$ for the east Cascades, west Cascades, and Coast Range. Notice the differences in the axes scales. Also note the downward sloping trend for all ecosystems except for the east Cascades where MFRI = 8 years.

the control group for the west Cascades and between 44% and 87% of the control group for the Coast Range.

Similar to TEC_μ, $E[F_s]$ was significantly affected by fuel reduction treatments. Fuel reduction treatments were effective in reducing $E[F_s]$ for all simulations. UR treatments had the smallest effect on $E[F_s]$ in the east Cascades simulations and $E[F_s]$ in the east Cascades simulations was most affected by combined UR + PF treatments applied every five years, which reduced $E[F_s]$ by an average of 6.01 units (units range from 0 to 100, see Eq. 3) for stands with an MFRI = 8 years and by 11.08 units for stands with an MFRI = 16 years. In the west Cascades and Coast Range, $E[F_s]$ was least affected by UR treatments, similar to the east Cascades simulations. The most substantial reductions in $E[F_s]$ were exhibited by treatments that removed overstory as well as understory trees, as in treatments UR + OT + PF and UR + OR + PF. In the west Cascades simulations, depending on treatment frequency, $E[F_s]$ was reduced by an average of 11.72–15.68 units where the MFRI = 143 years and by an average of 3.92–26.42 units where the MFRI = 230 years when UR + OT + PF was applied. When UR + OT + PF was applied to the Coast Range, $E[F_s]$ was reduced by an average of 7.06–23.72 units where the MFRI = 250 years and by an

average of 1.95–20.62 units where the MFRI = 500 years, depending on treatment frequency. Some UR + OR + PF treatments, when applied at a frequency of 25 years, resulted in $E[F_s]$ that was higher than that seen in UR + OT + PF in spite of lower TEC_μ in UR + OT + PF. A result such as this is most likely due to an increased presence of lower canopy tree fuels as a consequence of the increased lower stratum light availability that follows a clear-cut, as lower canopy tree fuels are among the highest weighted fuels in our simulated stands.

Modeled estimates of $E[F_s]$ were reflective of the mean amounts of C lost in a wildfire ($\bar{C}_{WF}$). $\bar{C}_{WF}$ was lower in the stands simulated with fuel reduction treatments compared to the control groups, with the exception of the east Cascades stands subjected to understory removal. Reductions in the amount of C lost in a wildfire, depending on treatment type and frequency, were as much as 50% in the east Cascades, 57% in the west Cascades, and 50% in the Coast Range. In the east Cascades simulations, amounts lost in wildfires were inversely related to the amounts of C removed in an average fire return interval for each ecosystem (Fig. 2), except for the Light Fire Regime No. 1 (MFRI = 8 years). Simulations in this fire regime revealed a slightly

increasing amount of C lost in wildfires with increasing amounts removed, though amounts removed were nonetheless larger than the amounts lost in a typical wildfire.

Biofuels

Biofuels cannot offset the reductions in TEC_μ resulting from fuel reduction, at least not over the next 100 years. For example, our simulation results suggest that an undisturbed Coast Range *Tsuga heterophylla–Picea sitchensis* stand (where MFRI = 500 years) has a TEC_μ of 1089 Mg C/ha. By contrast, a Coast Range stand that is subjected to UR + OT + PF every 25 years has a TEC_μ of 757.30 Mg C/ha. Over a typical fire return interval of 450 years (estimated MFRI was 500 years, MFRI generated from the model was 450 years) this stand has 1107 Mg C/ha removed, a forest fuel/biomass production of 2.46 Mg C·ha^{-1}·yr^{-1}, which amounts to emissions of 1.92 Mg C·ha^{-1}·yr^{-1} and 0.96 Mg C·ha^{-1}·yr^{-1} that can be avoided by substituting biomass and ethanol, respectively, for fossil fuels (see calculations in Appendix B). This means that it would take 169 years for C offsets via solid woody biofuels and 339 years for C offsets via ethanol production before ecosystem processes result in net C storage offsets (see Fig. 3). Converting Coast Range old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 34 years for biomass and 201 years for ethanol, and like all other biofuel calculations in our analysis, these are assuming a perfect conversion of potential energies. West Cascades *Tsuga heterophylla–Pseudotsuga menziesii* ecosystems (where MFRI = 230 years) that are subjected to UR + OT + PF every 25 years would require 228 years for C offsets using biomass as an offset of fossil-fuel-derived C and 459 years using ethanol. Converting west Cascades old-growth forest to second-growth forest reduces the amount of time required for atmospheric C offsets to 107 years for biomass fuels and 338 years for ethanol. Simulations of east Cascades *Pinus ponderosa* ecosystems had cases where stands treated with UR stored more C than control stands, implying that there is little or no trade-off in managing stands of the east Cascades for both fuel reduction and long-term C storage.

Discussion

We employed an ecosystem simulation model, STANDCARB, to examine the effects of fuel reduction on expected fire severity and long-term C dynamics in three Pacific Northwest ecosystems: the *Pinus ponderosa* forests of the east Cascades, the *Tsuga heterophylla–Pseudotsuga menziesii* forests of the west Cascades, and the *Tsuga heterophylla–Picea sitchensis* forests of the Coast Range. Our fuel reduction treatments for east Cascades forests included salvage logging, understory removal, prescribed fire, and a combination of understory removal and prescribed fire. West Cascades and Coast Range simulations included these treatments as

.... Mean C storage (UR + OT + PF applied every 25 years)
▪▪▪▪ C storage (UR + OT + PF applied every 25 years)
----- Mean C storage (Control group)
━ ━ ━ C storage (Control group)
——— Cumulative C offsets from ethanol production
       (from UR + OT + PF applied every 25 years)
━━━ Cumulative C offsets from biomass production
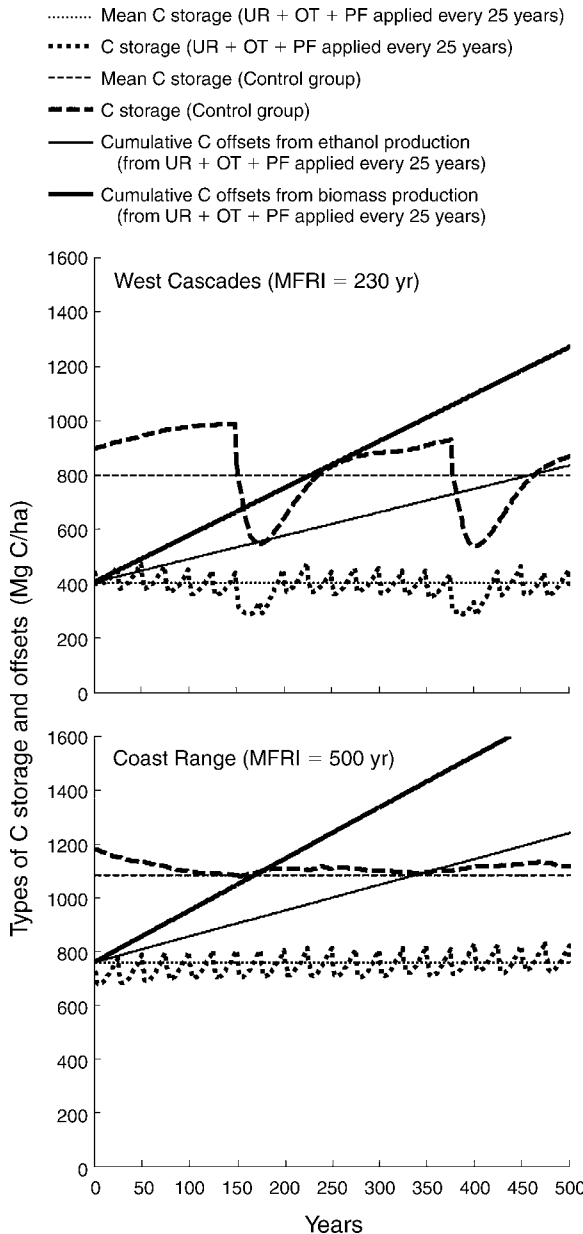   (from UR + OT + PF applied every 25 years)

Fig. 3. Time series plots of C storage, mean C storage, and biofuels offsets for control groups and fuel reduction treatment UR + OT + PF (understory removal + overstory thinning + prescribed fire) applied to a second-growth forest every 25 years for the west Cascades and Coast Range. East Cascades simulations were excluded from this plot because there was little or no trade-off incurred in managing these forests for both fuel reduction and C sequestration.

well as a combination of understory removal, overstory thinning, and prescribed fire. We also examined the effects of clear-cutting followed by prescribed fire on expected fire severity and long-term C storage in the west Cascades and Coast Range.

Our results suggest that fuel reduction treatments can be effective in reducing fire severity, a conclusion that is shared by some field studies (Stephens 1998, Pollet and

Omi 2002, Stephens and Moghaddas 2005) and modeling studies (Fulé et al. 2001). However, fuel removal almost always reduces C storage more than the additional C that a stand is able to store when made more resistant to wildfire. Leaves and leaf litter can and do have the majority of their biomass consumed in a high-severity wildfire, but most of the C stored in forest biomass (stem wood, branches, coarse woody debris) remains unconsumed even by high-severity wildfires. For this reason, it is inefficient to remove large amounts of biomass to reduce the fraction by which other biomass components are consumed via combustion. Fuel reduction treatments that involve a removal of overstory biomass are, perhaps unsurprisingly, the most inefficient methods of reducing wildfire-related C losses because they remove large amounts of C for only a marginal reduction in expected fire severity. For example, total biomass removal from fuel reduction treatments over the course of a high-severity fire return interval (MFRI = 230 years) in the west Cascades could exceed 500 Mg C/ha while reducing wildfire-related forest biomass losses by only ~70 Mg C/ha in a given fire (Fig. 2). Coast Range forests could have as much as 2000 Mg C/ha removed over the course of an average fire return interval (MFRI = 500 years), only to reduce wildfire-related biomass combustion by ~80 Mg C/ha (Fig. 2).

East Cascades simulations also showed a trend of decreasing $E[F_s]$ with increasing biomass removal, though a higher TEC_μ was seen in some understory removal treatments compared to control groups. We believe that the removal of highly flammable understory vegetation led to a reduction in overall fire severity that consequently lowered overall biomass combustion, thereby allowing increased overall C storage. Such a result may be indicative of actual behavior under field conditions, but the very low magnitude of the differences between the treated groups and the control group (0.6%–1.2%) suggests caution in assuming that understory removal in this or any ecosystem can be effective in actually increasing long-term C storage. Furthermore, we recognize that the statistically significant differences between the treated and control groups are likely to overestimate the significance of the differences between groups that would occur in the field, as the differences we are detecting are modeled differences rather than differences in field-based estimates. Field-based estimates are more likely to exhibit higher inter- and intrasite variation than modeled estimates, even when modeled estimates incorporate stochastic processes, such as those in STANDCARB. Our general findings, however, are nonetheless consistent with many of the trends revealed by prior field-based research on the effects of fuel reduction on C storage (Tilman et al. 2000), though differences between modeled and field-based estimates are also undoubtedly apparent throughout other comparisons of treated and control stands in our study.

We note an additional difference that may exist between our modeled data and field conditions. Our study was meant to ascertain the long-term average C storage (TEC_μ) and expected fire severities ($E[F_s]$) for different fuel reduction treatment types and application frequencies, a goal not be confused with an assessment of exactly what treatments should be applied at the landscape level in the near future. Such a goal would require site-specific data on the patterns of fuel accumulation that have occurred in lieu of the policies and patterns of fire suppression that have been enacted in the forests of the Coast Range, west Cascades, and east Cascades for over a century. We did not incorporate the highly variable effects of a century-long policy of fire suppression on these ecosystems, as we know of no way to account for such effects in a way that can be usefully extrapolated for all stands in the landscape. *Pinus ponderosa* forests may exhibit the greatest amount of variability in this respect, as they are among the ecosystems that have been most significantly altered as a result of fire suppression (Veblen et al. 2000, Schoennagel et al. 2004, Moeur et al. 2005). Furthermore, additional differences may be present in our estimates of soil C storage for the east Cascades. Our estimates of soil C storage match up very closely with current estimates from the Pringle Falls Experimental Forest, but it is unclear how much our estimates would differ under different fuel reduction treatment types and frequencies. Many understory community types exist in east Cascades *Pinus ponderosa* forests (i.e., *Festuca idahoensis*, *Purshia tridentata*, *Agropyron spicatum*, *Stipa comata*, *Physocarpus malvaceus*, and *Symphoricarpos albus* communities) (Franklin and Dyrness 1988). An alteration of these communities may result from fuel reduction treatments such as understory removal or prescribed fire, leading to a change in the amount and composition of decomposing materials, which can influence long-term belowground C storage (Wardle 2002). Furthermore, there may be an increase in soil C storage resulting from the addition of charcoal to the soil C pool, whether from prescribed fire or wildfire (DeLuca and Aplet 2008).

By contrast, ecosystems with lengthy fire return intervals, such as those of the west Cascades and Coast Range, may not be strongly altered by such a policy, as many stands would not have accumulated uncharacteristic levels of fuel during a time of fire suppression that is substantially less than the mean fire return intervals for these systems. Forests such as these may actually have little or no need for fuel reduction due to their lengthy fire return intervals. Furthermore, fire severity in many forests may be more a function of severe weather events rather than fuel accumulation (Bessie and Johnson 1995, Brown et al. 2004, Schoennagel et al. 2004). Thus, the application of fuel reduction treatments such as understory removal is thought to be unnecessary in such forests and may provide only limited effectiveness (Agee and Huff 1986, Brown et al. 2004). Our results

provide additional support for this notion, as they show a minimal effect of understory removal on expected fire severity in these forests, and if in fact climate has far stronger control over fire severity in these forests than fuel abundance, then the small reductions in expected fire severity that we have modeled for these fuel reduction treatments may be even smaller in reality.

We also note that the extent to which fuel reductions in these forests can result in a reduction in fire severity during the extreme climate conditions that lead to broad-scale catastrophic wildfires may be different from the effects shown by our modeling results, and are likely to be an area of significant uncertainty. Fuel reductions, especially overstory thinning treatments, can increase air temperatures near the ground and wind speeds throughout the forest canopy (van Wagtendonk 1996, Agee and Skinner 2005), potentially leading to an increase in fire severity that cannot be accounted for within our particular fire model. In addition to the microclimatic changes that may follow an overstory thinning, logging residues may be present on site following such a procedure, and may potentially nullify the effects of the fuel reduction treatment or may even lead to an increase in fire severity (Stephens 1998). Field-based increases in fire severity that occur in stands subjected to overstory thinning may in fact be an interaction between the fine fuels created by the thinning treatment and the accompanying changes in forest microclimate. These microclimate changes may lead to drier fuels and allow higher wind speeds throughout the stand (Raymond and Peterson 2005). While our model does incorporate the creation of logging residue that follows silvicultural thinning, increases in fire spread and intensity due to interactions between fine fuels and increased wind speed are neglected. However, we note that even if our model is failing to capture these dynamics, our general conclusion that fuel reduction results in a decrease in long-term C storage would then have even stronger support, since the fuel reduction would have caused C loss from the removal of biomass while also *increasing* the amount that is lost in a wildfire.

The amounts of C lost in fuel reduction treatments, whether nearly equal to or greater than our estimates, can be utilized in the production of biofuels. It is clear, however, that an attempt to substitute forest biomass for fossil fuels is not likely to be an effective forest management strategy for the next 100 years. Coast Range *Tsuga heterophylla–Picea sitchensis* ecosystems have some of the highest known amounts of biomass production and storage capacity, yet under the UR + OT + PF treatment a 169-year period is necessary to reach the point at which biomass production will offset C emitted from fossil fuels, and 338 years for ethanol production. Likewise, managed forests in the west Cascades require time scales that are too vast for biofuel alternatives to make a difference over the next 100 years. Even converting old-growth forests in these ecosystems would require at least 33 and 107 years for woody

biomass utilization in the Coast Range and west Cascades, respectively, and these figures assume that all possible energy in these fuels can be utilized. Likewise, our ethanol calculations assumed that the maximum theoretical ethanol yield of biomass is realized, which has yet to be done (Schubert 2006); a 70% realization of our maximum yield is a more realistic approximation of contemporary capacities (Galbe and Zacchi 2002).

In addition to these lags, management constraints could preclude any attempt to fully utilize Pacific Northwest forests for their full biofuels production potential. Currently in the Pacific Northwest there are $\sim 3.6 \times 10^6$ ha of forests in need of fuel reduction treatments (Stephens and Ruth 2005), and in 2004 the annual treatment goal for this area was 52 000 ha (1.44%). Unless a significantly larger fuel reduction treatment workforce is employed, it would take 69 years to treat this area once, a period that approximates the effective duration of fire suppression (Stephens and Ruth 2005). The use of SPLATs (strategically placed area treatments) may be necessary to reduce the extent and effects of landscape-level fire (Finney 2001). SPLATs are a system of overlapping area fuel treatments designed to minimize the area burned by high-intensity head fires in diverse terrain. These treatments are costly, and estimates of such treatment costs may be underestimating the expense of fuel reduction in areas with high-density understory tree cohorts that are time consuming to extract and have little monetary value to aid in offsetting removal expenses (Stephens and Ruth 2005). Nevertheless, it is clear that not all of the Pacific Northwest forests that are in need of fuel reduction treatments can be reached, and the use of strategically placed fuel reduction treatments such as SPLATs may represent the best option for a cost-effective reduction in wildfire severity, particularly in areas near the wildland–urban interface. However, the application of strategically placed fuel reduction treatments is unlikely to be a sufficient means in itself toward ecosystem restoration in the forests of the east Cascades. Stand-level ecosystem restoration efforts such as understory removal and prescribed fire may need to be commenced once landscape-level reductions in fire spread risk have been implemented.

CONCLUSIONS

Managing forests for the future is a complex issue that necessitates the consideration of multiple spatial and temporal scales and multiple management goals. We explored the trade-offs for managing forests for fuel reduction vs. C storage using an ecosystem simulation model capable of simulating many types of forest management practices. With the possible exception of some xeric ecosystems in the east Cascades, our work suggests that fuel reduction treatments should be forgone if forest ecosystems are to provide maximal amelioration of atmospheric CO_2 over the next 100

years. Much remains to be learned about the effects of forest fuel reduction treatments on fire severity, but our results demonstrate that if fuel reduction treatments are effective in reducing fire severities in the western hemlock–Douglas-fir forests of the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range, it will come at the cost of long-term C storage, even if harvested materials are utilized as biofuels. We agree with the policy recommendations of Stephens and Ruth (2005) that the application of fuel reduction treatments may be essential for ecosystem restoration in forests with uncharacteristic levels of fuel buildup, as is often the case in the xeric forest ecosystems of the east Cascades. However, this is often impractical and may even be counterproductive in ecosystems that do not exhibit uncharacteristic or undesirable levels of fuel accumulation. Ecosystems such as the western hemlock–Douglas-fir forests in the west Cascades and the western hemlock–Sitka spruce forests of the Coast Range may in fact have little sensitivity to forest fuel reduction treatments and may be best utilized for their high C sequestration capacities.

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APPENDIX A

STANDCARB model description (*Ecological Archives* A019-028-A1).

APPENDIX B

Biofuels analysis calculations (*Ecological Archives* A019-028-A2).

APPENDIX C

Carbon storage and fire severity results for each treatment type and frequency (*Ecological Archives* A019-028-A3).