forest ecology

Estimation of Aboveground Forest Carbon Flux in Oregon: Adding Components of Change to Stock-Difference Assessments

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A substantial portion of the carbon (C) emitted by human activity is apparently being stored in forest ecosystems in the Northern Hemisphere, but the magnitude and cause are not precisely understood. Current official estimates of forest C flux are based on a combination of field measurements and other methods. The goal of this study was to improve on existing methods by directly tracking components of change in tree C across a large region using field measurements. We used repeated Forest Inventory and Analysis (FIA) measurements on permanent plots to quantify aboveground live tree C flux over an 11-year period due to land-use change, disturbance, and harvest, on 1,073 plots across 5.96 million ha of nonfederal forestland in Oregon. Land-use change resulted in a 110,000 ha (1.9%) net increase of forestland between 1986 and 1997. However, there was a net loss of 3.4 Tg of live tree C due to land-use change because the forestland lost was more productive than that gained. Live woody C decreased significantly in eastern Oregon (−14.4 Tg), with mortality and harvest exceeding growth, primarily due to severe defoliation by western spruce budworm. However, C stores increased significantly in western Oregon (19.2 Tg) due primarily to large accumulations from growth on nonfederal public lands. We demonstrate that C accounting that uses remeasured probabilistic field sample data can produce detailed estimates of C flux that identify causes and components of change and produce more consistent estimates than alternative approaches.

Keywords: carbon accounting, land-use change, disturbance, management, forest inventory

The increasing concentration of carbon dioxide in the atmosphere due to human activities is implicated as one of the primary causes of recent increases in global mean temperatures and sea levels (Intergovernmental Panel on Climate Change [IPCC] Core Writing Team 2007). A substantial portion of the carbon (C) in the atmosphere is apparently being stored in terrestrial ecosystems in the Northern Hemisphere (1−2 Pg/year). However, neither the magnitude nor the causes of this terrestrial C storage across the many types of ecosystems present are precisely understood (Pacala et al. 2001, Hayes et al. 2012). Because many forest ecosystems can store much larger amounts of C than other terrestrial ecosystems, they are a focus of efforts to understand changes in C storage (i.e., fluxes) over time (McKinley et al. 2011).

The primary drivers of C flux in forests are natural disturbance, forest product harvest, and vegetation growth, which have different effects on the major C pools within forests, namely live trees, dead trees, understory vegetation, and soils. A variety of approaches have been developed to understand and quantify C cycling and flux in forests. Atmospheric inversion modeling uses changes in atmospheric carbon dioxide coupled with atmospheric transport models to infer C fluxes between the surface and atmosphere at subcontinental scales (Gurney et al. 2002). Biogeochemical modeling couples ecosystem models to climate models and remote sensing models to estimate C flux in response to observed changes in land cover for large regions (Turner et al. 2007, Coops et al. 2009). Forest inventory data are often used to calibrate remote sensing and biogeochemical models, but these data can also be used to directly estimate regional C flux from repeated field measurements of C pools (Smith et al. 2004, Woodbury et al. 2007). Inventory measurements usually provide design-unbiased estimates of regional attributes that require fewer assumptions than most models.

The IPCC standards for national and international reporting of C flux from forestlands (IPCC 2006) provide guidance for nations with developed resource information infrastructure to base estimates of C flux on extensive, statistically sound samples of forest C pools across those nations' forests. Parties to the United Nations Framework Convention on Climate Change agreed to those standards and compile national reports of annual C flux for 1990 and each subsequent year. In the United States, official estimates of C flux from forests have relied primarily on the probabilistic sample of the nation's forested lands by the USDA Forest Service's Forest Inventory and Analysis (FIA) program (Woodbury et al. 2007, US Environmental Protection Agency 2011). This inventory has evolved over time from a focus on timber resources on the most productive lands available for harvest to a more comprehensive assessment of all forestlands and from a focus on regional to national information needs (Frayer and Furnival 1999, Gillespie 1999).

Most developed nations derive flux estimates using the "stock-difference" method (IPCC 2006), which takes the difference between C pools estimated for forested lands at different times divided by the number of years (e.g., Baritz and Strich 2000, Smith et al. 2004, Nielsen et al. 2011, Swedish Environmental Protection Agency 2011). However, with this approach, some important components of forest C change must be assessed with other types of information, which may not be fully compatible with inventory data in definitions, timing, and scope. For example, although the net impact of land-use change (i.e., deforestation and afforestation) is included within stock-difference results, estimates of area and type of change are necessary to estimate soil C loss or recovery, which can lag land-use change by several decades (Caspersen et al. 2000). Estimates of the area and severity of forest fires are necessary to estimate emissions of other greenhouse gases (e.g., methane and nitrous oxide). Estimates of C removed for forest products are also needed, since the eventual flux to the atmosphere varies dramatically by product type and use and differs from dead wood in forests (Skog 2008).

Use of inventory data for detailed assessments of past C pools and fluxes is challenging because of past regional variation in survey designs, design changes over time, and the complexity of compiling C change across different periods and regions. As a result, online FIA databases tend to be simplified snapshot versions of inventories (Woudenberg et al. 2010, Gray et al. 2012). Improvements to the FIA program have been advocated to allow more explicit tracking of C over time (Wofsy and Harriss 2002). In many cases, information on components of change for trees needed for more detailed assessments of C gain and loss (i.e., growth, removals, mortality, and disturbance events) already exist in archive databases. Many of the plot locations have been revisited for decades with full tree remeasurement, and records of land-use change and disturbance have been collected. From these data, estimates of mortality, harvest, and growth have been reported as periodic change (e.g., Conner and O'Brien 1993, Azuma et al. 2004b, Hartsell and Brown 2002, Hansen and Brand 2006).

In this article, we present a method that uses plot disturbance data combined with individual tree growth and mortality information to make annual estimates of C stocks for live trees, harvested trees, trees that died from natural causes, and newly regenerating trees between surveys. We apply this method to data from FIA periodic surveys made on nonfederal forests in Oregon in the1980s and 1990s. These results provide greater temporal resolution than

previous studies and more detail about components of change in the live tree C stocks.

Methods
Study Area and Sampling Design
The study area consisted of 17.8 million ha of primarily nonfederal land in Oregon, USA, between 41.9 and 46.3°N latitude and 116.5 and 124.6°W longitude (Figure 1). Forests in Oregon grow in a great variety of conditions, with annual precipitation ranging from 25 to >350 cm, mean annual temperature from −1 to 12° C, and elevations from 0 to 2,500 m above sea level (Franklin and Dyrness 1973). On average, forests west of the Cascade Mountains ("western Oregon") are more productive and have a greater abundance of hardwood species than those east of the Cascades ("eastern Oregon").

The data used in this study were collected by the USDA Forest Service's FIA program. The objectives of the FIA program are to characterize status and trends of multiple resource attributes for lands that meet the criteria for "forestland" (i.e., land areas ≥0.4 ha that support or previously supported ≥10% canopy cover of trees and were not primarily managed for a nonforestland use). Changes in forest land use based on this definition were used to estimate afforestation and deforestation, even though in some cases trees may still be present on a nonforest land use. Stands that regenerate after logging or fire are considered to be continuous forestland use. These resources are measured with a probability-based sample design (Olsen et al. 1999). Plots were measured in the 1980s (1985–1988) and remeasured in the 1990s (1995–1999). This decadal "periodic" inventory approach was common across the nation before 2000. Our study does not include lands managed by the USDA Forest Service or the US Bureau of Land Management (BLM) in western Oregon, because a probabilistic sample of those lands did not begin until the mid-1990s (Max et al. 1996), and comparable data on C flux for the period including 1990 were not available. However, other federal land managers (e.g., National Park Service) and BLM lands in eastern Oregon were in the FIA population and made up 0.3 and 28% of the forested land area in western and eastern Oregon, respectively. Because the unsurveyed USDA Forest Service lands and western Oregon BLM lands comprise the large majority of federal forestland in Oregon, for simplicity, we refer to the study area as "nonfederal."

The periodic inventory of the 1980s and 1990s was based on a randomized systematic grid of 1 plot/3,000 ha for most of the state, and a stratified sample in portions of eastern Oregon dominated by rangeland and low-productivity western juniper (*Juniperus occidentalis* Hook.) forests (Azuma et al. 2004a, 2004b, 2005). We used a systematic subset of the periodic inventory sample that were part of the later "annualized" inventory (Bechtold and Patterson 2005) and that had gone through a careful effort to compile and reconcile long-term disturbance and land-use change data across multiple inventories. There were 1,664 plots in our sample, of which 1,073 were forested at one or both inventory dates.

Field Data
Outside the eastern Oregon juniper zone, five points were installed in the 1980s at each sample grid point in productive forest using a fixed design (four points located 66 m from a central point in each cardinal direction), although points were systematically moved off the intended design if necessary to keep the sample within a single stand or land-use type. Small trees (>2.5 cm and <12.5 cm dbh, 1.37 m) were selected with a 2.35-m radius plot at each point,
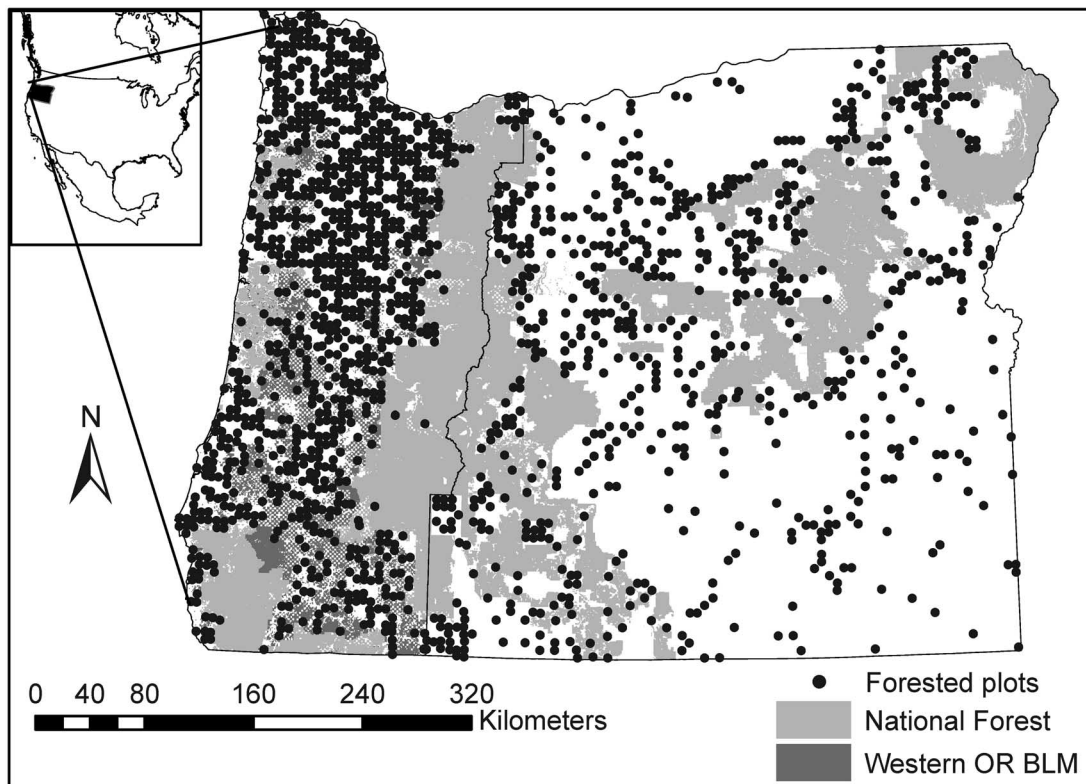
Figure 1. Oregon, USA, with approximate locations of the forested plots used in this study. Shaded areas indicate lands managed by National Forests (statewide) and the Bureau of Land Management (western Oregon), which were not included in this study.

and larger-diameter trees were selected with a variable radius, 7-m basal area factor prism up to a maximum distance of 17 m from point center. Diameter, height, species, crown ratio, crown class, and damage were measured for all selected trees. An estimate of cull was made for each tree bole to account for breakage and rot. For low-productivity forests, defined as those estimated to produce less than 1.4 m³/ha/year of wood at culmination of mean annual increment (Hanson et al. 2002), only areal cover by species was recorded. In the 1990s plots were fully remeasured on productive forestland using the same procedures, with individual trees relocated and measured. New individual-tree measurements were taken on low-productivity forest and on grid locations that changed from federal to nonfederal ownership. Field crews increment-cored trees that grew into the plot (i.e., passed the diameter threshold for measurement) to measure growth, assigned causes of tree mortality, determined changes in land use, and estimated types and dates of natural and human disturbances that had occurred on the plot since the previous measurement. Dates of distinct events (e.g., logging and fire) were obtained via contact with the landowner or estimated based on the amount of vegetation regrowth. For prolonged mortality events, crews either assigned the peak year (e.g., for insect outbreak) or identified it as chronic (e.g., for root disease). Selected trees were measured and increment-cored for age to calculate potential site productivity (Hanson et al. 2002) unless previous measurements were available and appropriate.

In the juniper zone, most of the data came from 497 new plots sampled in 1999, because only 60 plots had been sampled in the 1980s and remeasured in 1999. We used measurements from the smaller sample to build empirical equations of juniper mortality and growth. The new juniper plots were sampled with the new nation-ally standardized annual plot design, which consisted of four points in a fixed design (three points located 36.6 m from a central point), with trees >2.5 and <12.5 cm dbh measured on 2.1-m radius plots and larger trees measured on 7.3-m radius plots at each point. A subset of trees were increment-cored for growth. All other measurements were the same as in the nonjuniper sample.

Periodic Change Analysis

For the periodic (decadal) change analysis, we followed standard FIA compilation procedures for periodic inventories (e.g., Azuma et al. 2004a) to estimate the aboveground C of live trees at each inventory period and the components of change, namely growth, removals (e.g., from logging), and mortality. Growth of individual trees was calculated from the change in tree size between measurement dates or between the first date and the year of death for removals and mortality. We assigned year of death from recorded disturbance dates for distinct events and to the midpoint between measurements for tree death not associated with a distinct event. The biomass and C lost in removals and mortality were based on the estimated size of the tree at the time of death. For trees that grew into the plot, estimates of heights (and dbh for trees not able to be increment-cored) at the time of the first survey were calculated using equations of annual height and annual squared diameter growth. These empirical regression equations were developed by FIA from remeasured trees by regressing growth rates on species, diameter, crown ratio, tree age, and site index. Six of the plots that had not been measured previously were disturbed between survey dates. We used crew notes on previous stand age and photo interpretation of previous stand density to select subsets of similar, fully remeasured plots, and average biomass, growth, and mortality rates were used to estimate

biomass on the new plots before disturbance. Of the 23,599 trees in the analysis, 60% were measured twice, 20% had been increment-cored at time 2 to estimate growth and previous dbh, and growth for the remaining 20% was modeled (8% were <2.5 cm dbh at time 1).

Estimates of aboveground live-tree woody C were based on regional FIA equations of the sum of bole, bark, and branch biomass for each tree measurement multiplied by 0.5. Bole biomass (ground to tip) was calculated from regional species-specific volume equations and wood densities documented in Waddell and Hiserote (2005; see also Zhou and Hemstrom 2010) after accounting for the proportion of the bole estimated to be cull. Bark and branch biomass were calculated from regional species-specific equations selected from Means et al. (1994) and documented in Zhou and Hemstrom (2010), except for branch Equation 16 (from Snell and Little 1983) and bark Equations 8 and 20 (from Means et al. 1994, Equations 5 and 275, respectively). Most equations use both dbh and height data, whereas a few bark and branch equations use dbh only. An expansion factor derived from the fixed-area plot size or the tree's dbh and prism factor (depending on tree selection method) was used to convert individual tree C to an area basis (Mg/ha).

Annual Tree Size and Biomass Change

Annual estimates of aboveground live tree woody biomass and C were calculated for each plot, by determining each tree's status (live, dead, and cut) and size (height and dbh) each year between surveys. For trees that were alive in both surveys, height and dbh were interpolated between survey year values using annual height growth and annual squared diameter growth rates. Annual sizes of trees that died from distinct harvest or mortality events were obtained by interpolating between the size at the first survey and their estimated size at the year of death. For other natural mortality trees (lacking specific mortality years), we reduced each tree's live biomass (from the first survey) proportionally between surveys, which is effectively the same at the population level as assigning a random mortality year. For small trees in the second survey, we back-calculated using the annual height growth and annual squared diameter growth rates to determine the year they became at least 2.5 cm dbh to be included in the sample population.

Statistical Analysis

Statistical calculation of survey estimates differs from standard approaches commonly applied to designed studies (e.g., analysis of variance). We calculated variance estimates for all values using double-sampling for stratification (Cochran 1977, Scott et al. 2005), as follows. The primary sample consisted of 154,195 points (~1 point/187 ha), which included all field grid locations. These points had been photo-interpreted and classified to land class, tree density, and tree height for standard FIA statistical estimation. Photo points were grouped into strata based on land class, stand size, and ownership. The ratios of the number of points within each strata to the known area of sampled nonfederal lands determined the plot weight and were used to calculate population means and variances (e.g., MacLean 1972). Ratio estimates (e.g., Mg/ha) and their variances were calculated using the ratio of means estimator (Scott et al. 2005). We compared means by calculating the mean and variance of the difference of two estimates (e.g., C in mortality versus removals) and determining the type I error rate for the Z-statistic of the mean being different from 0 (Zar 1984). Results were considered signifi-

Table 1. Change matrix table for land area and live-tree above-ground carbon on nonfederal lands in Oregon.

	Eastern Oregon	Western Oregon	Total
Area (10³ ha)			
Forestland 1986	2,771 (76)	3,011 (46)	5,783 (89)
Conversion to nonforest	−21 (11)	−55 (17)	−75 (20)
Reversion to forest	150 (46)	32 (10)	183 (47)
Forestland 1997	2,901 (80)	2,989 (47)	5,890 (93)
Nonforest 1997	10,435	1,494	11,929
Total	13,336	4,483	17,819
Carbon (10¹² g)			
Forestland 1986	76.6 (4.5)	212.6 (7.9)	289.2 (9.1)
Conversion to nonforest	−0.4 (0.2)	−4.6 (1.9)	−5.0 (1.9)
Reversion to forest	0.9 (0.5)	0.7 (0.4)	1.6 (0.6)
Net land-use change	0.5 (0.6)	−3.9 (1.9)	−3.4 (2)
Growth	18.1 (0.7)	106.1 (3.2)	124.2 (3.2)
Harvest	−22.0 (3.9)	−70.8 (6.4)	−92.9 (7.5)
Mortality	−10.4 (1.7)	−16.1 (2)	−26.4 (2.6)
Net tree change	−14.4 (4.2)	19.2 (6.4)	4.9 (7.6)
Forestland 1997	62.8 (3.1)	227.9 (8)	290.7 (8.5)

Data are totals (SE).

cant if the probability of a type I error was <0.05. For simplicity of reporting, the 1985–1988 and 1995–1999 sample periods are labeled as 1986 and 1997, respectively. Our analyses investigate the primary variables associated with forest C density and flux, namely region, ownership, forest type (Eyre 1980), site productivity, and stand age.

Results
Periodic Change

There were an estimated 5.78 million ha of nonfederal forested land in Oregon in 1986, which increased to 5.89 million ha by 1997 (Table 1). Most of the net increase in forestland area occurred on low-productivity juniper sites in eastern Oregon, whereas in more productive western Oregon there was a net loss of 23,000 ha. Despite the net gain in forestland area, there was a significant net loss of 3.4 Tg of aboveground live woody C due to land-use change ($Z = −1.71$, $P = 0.044$). C losses due to land-use change were attributed to constructed development, conversion to agricultural use, and river erosion (67, 29, and 4% of total loss, respectively). C gains due to land-use change were primarily attributed to juniper encroachment on rangeland in eastern Oregon and reversion to forest from agricultural use in western Oregon (57 and 43% of the land-use change total, respectively).

The net change of aboveground live woody C on nonfederal forestland was characterized by a substantial decrease in eastern Oregon ($Z = −3.45$, $P = 0.0003$) and a substantial increase in western Oregon ($Z = 3.00$, $P = 0.0013$), with a statewide net change indistinguishable from 0 ($Z = 0.64$, $P = 0.52$) (Table 1). Growth as a proportion of 1986 C was twice as high in western Oregon as in eastern Oregon (0.50 versus 0.24), whereas proportional mortality was lower (0.08 versus 0.14) and proportional harvest was somewhat higher (0.33 versus 0.29).

The causes of C change due to natural mortality and harvest over the 11-year period varied across the state. Most of the aboveground live-tree C converted to mortality in eastern Oregon was caused by bark beetles or defoliating insects, which tend to be important in dry forest types (Figure 2). The major causes of natural mortality in western Oregon were weather (primarily wind-throw), suppression (from shading by faster-growing trees), or root disease. The majority
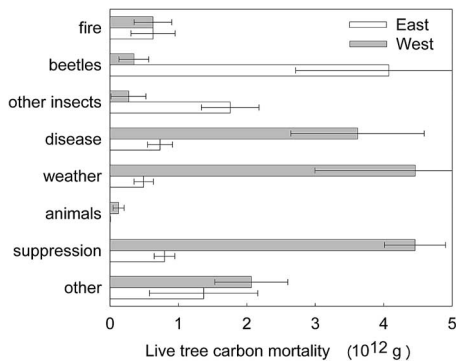
Figure 2. Total C of trees that left the live-tree pool between 1986–1997 in eastern and western Oregon nonfederal forestlands, by cause of mortality. Animals refers to noninsect damage, mostly grazing, browsing, or beaver damage; suppression refers to mortality caused by shading and crowding. Bars indicate the SE of the estimate.
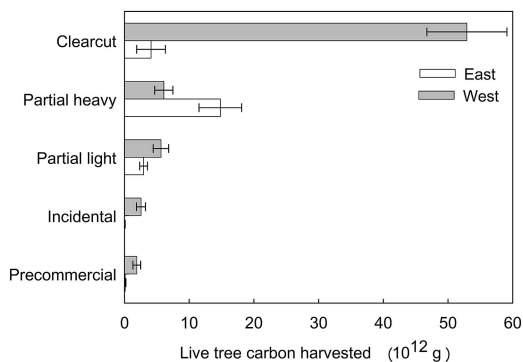

Figure 3. Total C of trees that left the live-tree pool between 1986 and 1997 in eastern and western Oregon nonfederal forestlands, by harvest type. Precommercial refers to thinning of noncommercial trees, usually left on site. Bars indicate the SE of the estimate.

of the C converted from the live tree pool by harvesting was done with partial-cut harvesting in eastern Oregon, primarily in the form of heavy thinning (Figure 3). Although substantial amounts of C were removed with partial-cut harvesting in western Oregon as well, 77% of harvest removal was by clearcutting. A large proportion of the harvest in eastern Oregon consisted of trees that were already killed or heavily damaged from repeated defoliation by an outbreak of western spruce budworm (*Choristoneura occidentalis* Freeman) that occurred in the 1980s and 1990s (Azuma and Overhulser 2008). Other instances of multiple disturbances were evident; for example, some of the harvested plots had first experienced forest fire and were subsequently salvage-logged.

The rate and type of C flux differed significantly among regions, ownerships, and forest types and were related to inherent site productivity, stand age, and management objectives. In eastern Oregon, forest industry lands had the highest C density at the beginning of the period but also had the highest harvest rate, with a 35% reduction in C density and a final value similar to that for the other two ownership groups (Figure 4). Part of this change may be due to the impacts of western spruce budworm; although the forest industry owned 25% of the nonfederal forestland, they owned 45% of the area in the highly budworm-susceptible forest types (Grand fir [*Abies grandis* (Dougl. ex D. Don) Lindl.], white fir [*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.], Douglas fir [*Pseudotsuga men-*

ziesii (Mirbel) Franco], and Engelmann spruce [*Picea engelmanii* Parry ex Engelm.]). In contrast, eastern Oregon public lands (outside National Forests), dominated by protected areas and low-productivity juniper forest, had very little harvest. Unharvested mortality levels were similar among owners. In western Oregon, harvest rates were also highest on industry lands (Figure 4) but with substantial growth rates, C density at the end of the period was slightly higher than that at the beginning of the period. On western Oregon nonfederal public lands, lower harvest rates and high growth rates resulted in a 27% increase in C density over the measurement period.

Total C density consistently increased with stand age, but differences among stand ages varied substantially by site productivity (Figure 5). Forests in higher productivity classes generally had greater C density and more rapid increases in C density with stand age than forests in lower productivity classes. Growth rates increased with productivity class and increased with age at least up to 40 years, followed by a decline in older stands. Note that although individual tree growth rates in Oregon continue to increase with age (e.g., Curtis and Marshall 1993), stand-level growth show different patterns because the proportion of stands that were harvested increased greatly with stand age; for example, 37% of the stands that were 40–50 years old at the first survey had undisturbed growth for only a few years before being harvested before the second period. Harvest rates increased with stand age up to 60–80 years, but then dropped off as the few remaining older stands were left uncut. There were no apparent patterns of natural mortality C density related to site productivity and stand age.

Annual trends in C density also varied among ownership groups and forest types, revealing the effects of disturbances, timing of harvest (driven by market conditions and disturbances), inherent growth rates, and differences in management objectives. For example, C density in white fir forests in eastern Oregon declined steadily on nonfederal public lands throughout our study period but did not begin to decline until 1993 on other private lands (Figure 6). In contrast, C density in red alder forests in western Oregon increased steadily on public lands but oscillated around a lower mean for the other ownership classes because mortality and harvest were roughly balanced by growth.

Discussion
Spatial and Temporal Patterns of C Flux in Oregon

We have illustrated how ground-measured forest inventory data for vegetation density, components of biomass change, site potential, and the nature and timing of land-use changes, disturbance, and harvest events can be used to provide detailed insight into the components and timing of forest C flux. These measurements and attributes enhance our understanding of potential causes and mechanisms of changes in forest C beyond the relatively simple accounting of net changes between inventory periods. For example, we were able to show that land-use change was a relatively minor component of aboveground live-tree C flux in Oregon during this period, possibly due to statewide land-use planning policies (Kline et al. 2004) and in likely contrast with those of other western states with greater rates of deforestation (e.g., Gray et al. 2005). Despite a net increase of 1.8% in nonfederal forestland area, land-use change caused a net decrease of 1.2% in live tree C due primarily to losses occurring in more productive forest types. In addition to human-caused land-use
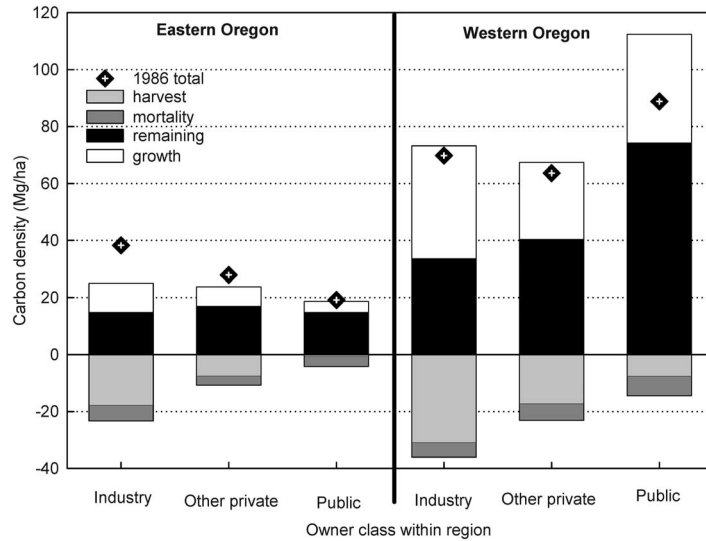
Figure 4. C density (Mg/ha) and components of change by owner class for eastern and western Oregon. Diamond indicates C density in 1986, which are reduced by the negative values (harvest and mortality) to densities labeled "remaining," which is augmented by growth to the C density in 1997, the level at the top of each bar.
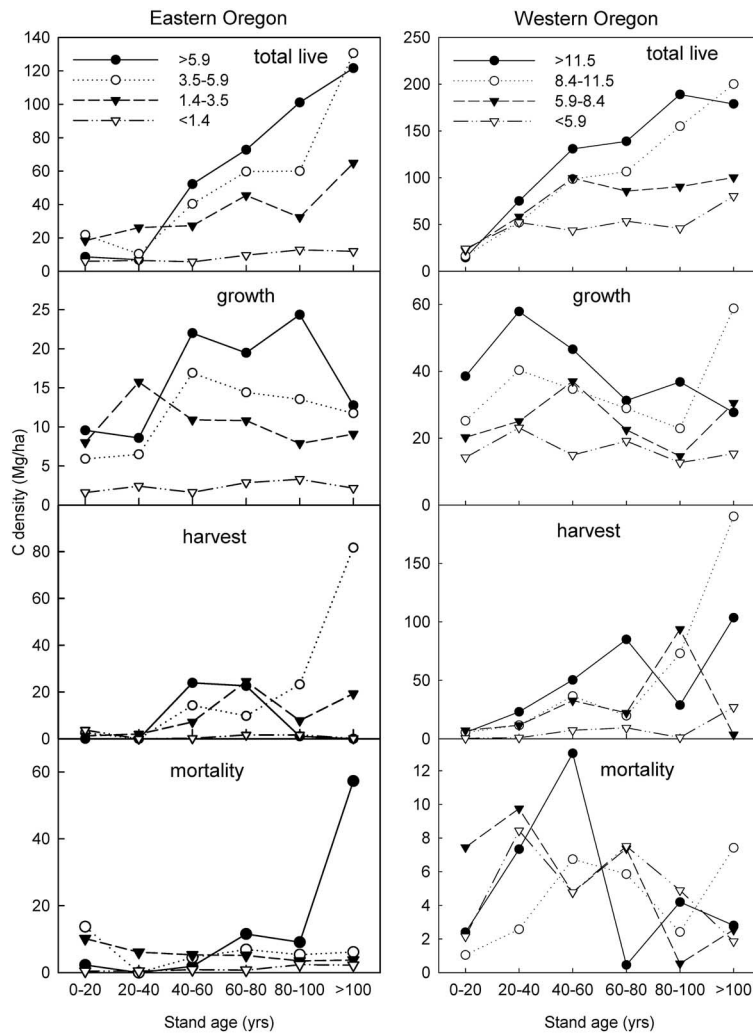

Figure 5. C density and components of change by site productivity (maximum predicted wood volume growth [m3/ha/year]) and stand age class at the first survey date. Site productivity classes differ between eastern and western Oregon, and the y-axis scale varies among graphs.
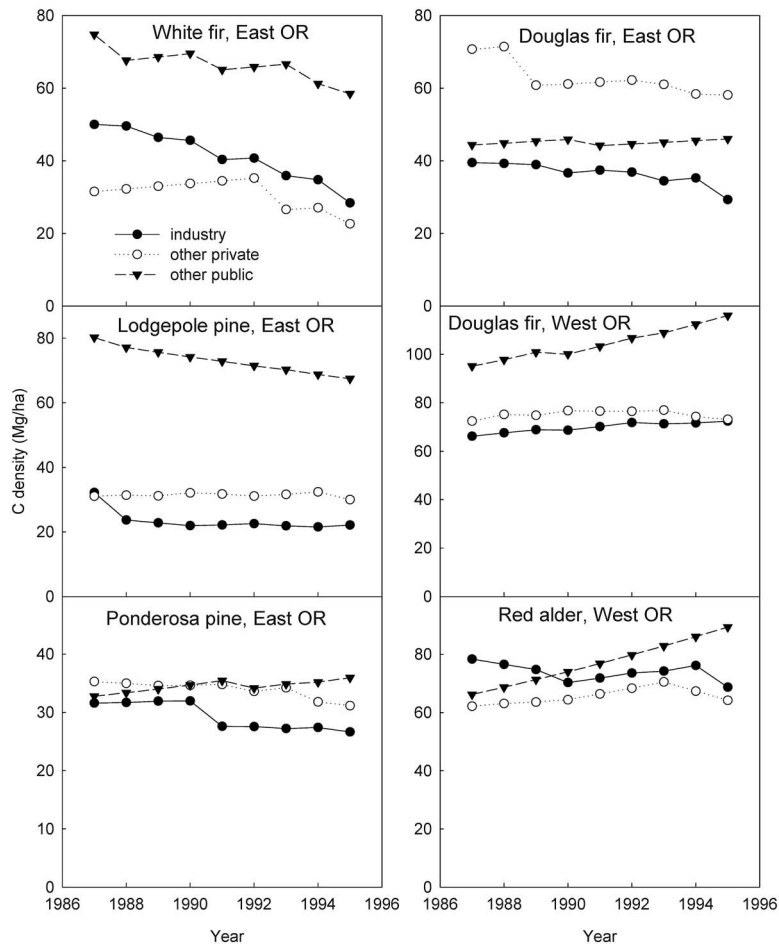
Figure 6. Annual estimates of C density between inventories, by ownership class for selected forest types (based on dominant species) in eastern and western Oregon. Changes reflect timing of cutting, mortality, and growth on mean C density. The y-axis scale varies among graphs.

change, the inventory detected natural land type changes, for example, from river erosion. River transport of C to oceans is an important flux estimated in C accounting with substantial uncertainty (Pacala et al. 2001, Lal 2003).

Information on site potential, ownership, and management practices developed in this assessment provided insights into the nature and rate of C flux. The net change in aboveground live-tree C varied greatly among regions within Oregon and by owner class. Mortality agents differed, which affects the eventual fate of wood after it dies (e.g., standing versus down) (Vanderwel et al. 2006). Owner response to mortality events differed as well, with private (and to some extent state) owners more likely to respond to mortality from insects and fire with salvage-logging than federal land managers whose management goals include providing habitat for dead wood-dependent wildlife (Russell et al. 2007). Potential productivity classes are rarely addressed directly in many C analyses but can be key to understanding differences in C accumulation across a region. The most important factor contributing to differences in aboveground live-tree C flux over this 11-year period in Oregon was the difference in growth rates between eastern and western Oregon. Because the proportional allocation of tree biomass to roots declines with increasing productivity (Litton et al. 2007), differences in total tree C storage were probably lower than those we report for aboveground C. Nevertheless, we found that live-tree C accumulation declined more quickly with stand age on nonfederal land than it

does on idealized curves based on productivity of unmanaged stands (e.g., Curtis and Marshall 1993), in part because harvests reduced tree density and subsequent short-term stand-level growth.

We found an increase in C density on nonfederal public lands, which was probably due to a combination of recovery of many of the state forests from severe wildfires in the 1930s, recent policies of providing some late-successional habitat, and the goal of maximizing wood production on lands that are harvested. In contrast, most industrial private lands are managed to maximize net present value, which results in stands being harvested at earlier stand ages and lower C densities, resulting in less total C being stored or harvested, than could be accomplished with longer rotations (Duerr 1960, Curtis and Marshall 1993). Although there seems to be a North American terrestrial sink for C (e.g., Pacala et al. 2001), Oregon nonfederal forests were not contributing to it between 1986–1997 in terms of aboveground live-tree C. Including the fate of C in wood products and killed trees might change that conclusion, however, because the C removed from the live tree pool is not immediately respired to the atmosphere (Malmsheimer et al. 2011).

Methods and Models of C Flux Estimation

Because C is ubiquitous in terrestrial ecosystems, quantifying its abundance and change over time requires data and models that are large in geographic scope and have temporal and spatial consistency.

The stock-change method of C accounting is simpler than one including components of change in that it only requires forest inventory assessments of C stocks over time, but more complex in that it is necessary to integrate other information to quantify the types and amounts of C fluxes. In the United States, for example, the official analysis of C flux from forest due to land-use change is based on USDA National Resources Inventory photo interpretation, using different sample points with slightly different definitions than those used by FIA (US Environmental Protection Agency 2011). Similarly, harvest data are commonly based on comprehensive surveys of wood product manufacturing facilities (e.g., Johnson 2001) and may be a more accurate assessment of the amount of wood going to wood products in a given year in a state than the FIA sample of trees cut on plots. However, manufacturers measure wood volume directly, whereas inventories estimate volume from dbh and height using regional volume equations, which could result in substantial differences in estimates.

We believe there are advantages to deriving as many estimates as possible from comprehensive, internally consistent sources like forest inventories. Records of land-use and disturbance histories from permanent plots could also address the significant temporal lags in C flux, as dead wood decays over time, as soils lose or accumulate C after deforestation or afforestation, and as vegetation grows through typical successional patterns (Caspersen et al. 2000, McKinley et al. 2011). The different types of disturbance detected from field remeasurement, particularly partial mortality events with modest effects on canopy cover, can be difficult to detect with satellite imagery; although recent developments in image processing are promising in this regard (e.g., Cohen et al. 2010, Kennedy et al. 2010).

Comparing estimates of C flux among a variety of approaches should improve overall estimates and resolve uncertainties. One current approach to estimating C flux is to apply vegetation succession models to stand age and vegetation change information derived from satellite classifications (Schroeder et al. 2008). Other approaches to modeling regional C flux use physiologically based ecosystem models driven by climate data and initialized by remote-sensing classifications of stand age and disturbance (e.g., Masek and Collatz 2006, Turner et al. 2011). Forest inventory data are often used to develop the remote-sensing classifications and to refine the modeling parameters, but otherwise the models run independently of regional inventory assessments. Physiological parameters for these models are typically available for only a few of the species in a region and stands are usually assumed to be even-aged. These models are particularly useful for assessing potential responses to future changes in climate or disturbance. Other multiscale efforts are working to improve estimation of C flux by integrating intensive physiological measurements with extensive climate, inventory, and satellite information (Wofsy and Harriss 2002). More accurate and inclusive inventory measurements would improve our understanding of the differences in estimates among methods and help quantify aspects that are not well inventoried currently, like woody encroachment and afforestation, wetland sinks, and C export in rivers (Hayes et al. 2012). One important limitation to all of these forest C accounting methods is that tree C is estimated using allometric equations of tree biomass components and wood density factors, some of which are based on highly localized original data. The accuracy of estimates based on these combined equations with respect to the true mean of forestlands or subdomains within forestlands is largely unknown (e.g., Melson et al. 2011).

Future Enhancements

The method of C accounting we have demonstrated suggests several potential enhancements. Current classifications of satellite imagery are based on static plot measurements, but it should be possible to develop models linking annual changes in spectral response to annual estimates of changes in plot-measured C stores. This dynamic approach might be particularly useful in countries currently unable to measure actual change on forest plots in a probabilistic design. Our approach can also be expanded to all forested lands in a region and larger sets of C pool measurements. Although newer data have been collected on all lands in our region using a different, nationally standardized plot design (Bechtold and Patterson 2005), the measured growth and mortality rates from our study can be combined with information on disturbance and harvest in the intervening period to project plot C estimates between survey years. In addition, repeated measurements of standing dead trees (snags), down wood, and forest floor thickness can be used to estimate those C pools directly and develop improved regionally specific models of change for plots where remeasurements are not available. Although this type of detailed accounting can be more complicated than other approaches, our expectation is that it will result in more precise estimates of regional C flux.

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