Two Decades of Carbon Flux from Forests of the Pacific Northwest

Estimates from a new modeling strategy

Warren B. Cohen, Mark E. Harmon, David O. Wallin, and Maria Fiorella

Earth's climate is greatly influenced by carbon dioxide concentrations in the atmosphere, which have increased by approximately 25% since the dawning of the Industrial Revolution, when fossil fuel combustion accelerated greatly (IPCC 1990). Another important anthropogenic source of atmospheric carbon is associated with deforestation, especially in primary forests, which can store large amounts of carbon in the form of organic biomass (Detwiler and Hall 1988, Harmon et al. 1990). Attempts to balance the global carbon budget have indicated that 20%–50% (1–3 Pg/yr) of the carbon released to the atmosphere from anthropogenic sources cannot be accounted for or fully explained (Dale et al. 1991, Dixon et al. 1994, Post et al. 1990, Sarmiento 1993). Recent efforts to decrease uncertainty in global carbon cycling processes have focused on improving estimates of land-use change in the tropics (Dale et al. 1991, Skole and Tucker 1993), sequestration in soil humus (Schlesinger 1990), increased biomass accumulation associated with carbon dioxide fertilization (Keeling et al. 1989), and sequestration by temperate forests recovering from past harvest (Kauppi et al. 1992, Sedjo 1992, Tans et al. 1990).

Warren B. Cohen is a research forester and Maria Fiorella is a forester with the Forestry Sciences Laboratory, Pacific Northwest Research Station, USDA Forest Service, 3200 SW Jefferson Way, Corvallis, OR 97331. Mark E. Harmon and David O. Wallin are assistant professors with the Department of Forest Science, Forestry Sciences Laboratory 020, Oregon State University, Corvallis, OR 97331. (Wallin is currently assistant professor at the Center for Environmental Sciences, Huxley College, Western Washington University, Bellingham, WA 98225.)

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Each of the existing studies focusing on terrestrial carbon sources and sinks has potential problems that limit its usefulness in balancing the global carbon budget. Given a lack of regional data, global-scale models are forced to use a limited number of biomes to represent vast areas (e.g., Houghton et al. 1983), ignoring potentially important differences within biomes in terms of production capacity and land-use history (Downing and Cataldo 1992, Lugo and Brown 1991). Another problem is that some of the key carbon budget components (e.g., soils, woody detritus, forest products) are often ignored (Dixon et al. 1994). Because carbon budgets are most often balanced by solving for the value of carbon flux from an unknown component (e.g., terrestrial biomass) using estimated, measured, or modeled fluxes from other components (e.g., fossil fuels, the atmosphere, oceans), each having several unknown terms, large uncertainties in estimates are inevitable (Houghton 1993). Finally, many analyses are not spatially explicit, which leads to several problems. Forest harvest, for example, is not evenly distributed within regions and may be more concentrated in productive areas. Thus, estimates of carbon flux based on clearing rates and average forest characteristics (e.g., Harmon et al. 1990) may be biased. Another factor is that inventories of land use and carbon stores often miss key areas (e.g., Turner et al. 1995 had no data on National Forests) or may contain unknown amounts of overlap (e.g., in Kurz et al. [1992], peatland and forestland inventories overlapped), leading to potentially significant estimation errors.

Terrestrial carbon fluxes cannot realistically be measured directly on a regional or global scale; consequently, large-scale fluxes must be estimated. We are developing a strategy to estimate regional carbon fluxes that is designed to overcome many of the shortcomings of other approaches. Models are being developed to estimate maximal live and detrital biomass accumulation and changing rates of accumulation with time and vegetation development. We are using remotely sensed and spatial biogeoclimatic data to develop estimates that are spatially explicit, complete, and nonoverlapping and that adequately represent variation in land use and productivity in a region. We consider all the major components that store carbon, including all live plant parts, all forms of detritus, mineral soil, and forest products. Although our modeling strategy is evolving and has not yet reached maturity, we recently completed a pilot study that demonstrates its value.

The setting for our pilot study is the Pacific Northwest (PNW) region of the United States. For our purposes, this region is defined as consisting of the roughly 10.4 million ha of forest land between the Cascade Range crest and the Pacific Ocean in Oregon and Washington. Similar vegetation and land-use conditions prevail from northern California to Alaska, and eventual expansion of this study to those areas is desirable. The PNW region is one of the most productive forest regions in the world, with primary old-growth stands having a total of as much as 650 x 10^6 g C/ha in aboveground and belowground pools (Grier and Logan 1977, Harmon et al. 1986). The region also has a long history of intensive clearcut and plantation management for wood products (Waddell et al. 1989). Primary forests of the PNW region contain a large volume of coarse woody debris that generally takes several centuries to reach equilibrium following a disturbance (Harmon and Hua 1991), and amounts of logging debris left behind after clearcut harvest have varied greatly over the past century (Harmon et al. 1996).

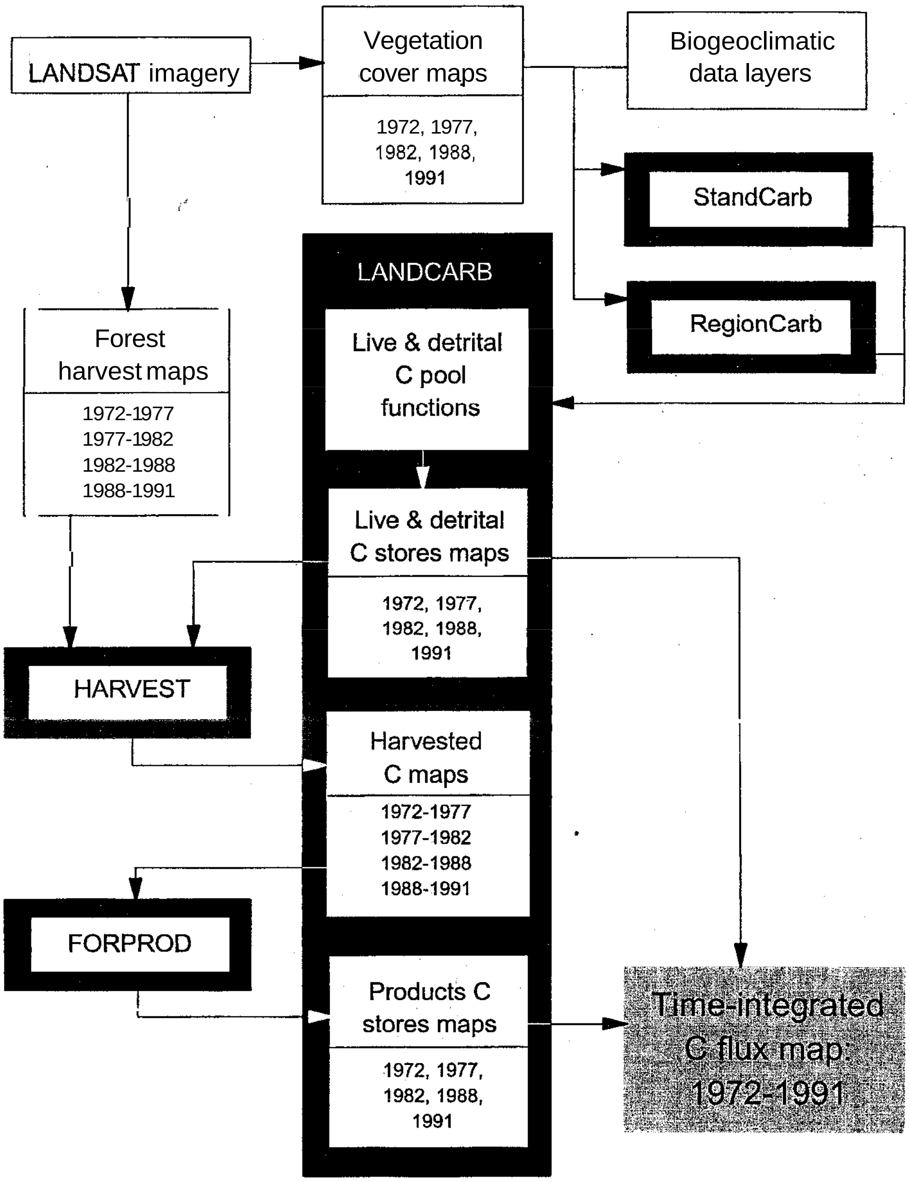

Figure 1. Schematic for the overall modeling strategy **used** to estimate carbon flux.

C flux modeling strategy

Our strategy for estimating carbon fluxes involves a collection of forest ecosystem and related models largely driven by Landsat satellite imagery and other spatial data in raster Geographic Information System (GIS) form (Figure 1). Using the methods of Cohen et al. (1995),[1] Landsat imagery is processed to produce maps of vegetation cover and forest harvest activity for several different dates (e.g., 1972–1991, in roughly five-year intervals). The vegetation cover maps are developed with the aid of extensive ground data and aerial photography interpretation, combined with an in-depth statistical analysis. Harvest maps are derived by direct spatial comparison of multi-date images to detect significant shifts in reflected solar energy by forest stands. These maps, plus spatial data bases on forest site productivity class, climate, and soils are provided as input to two separate models: StandCarb and RegionCarb. StandCarb is a forest stand–level model that integrates the effects of colonization success, species succession, and disturbance severity (e.g., clearcut versus thinning) on rates of carbon production and decomposition over time (Harmon et al. 1995). RegionCarb (which is still in early stages of development) uses regional biogeoclimatic data bases to estimate the effects of radiation, temperature, water balance, physiognomic structure, and species on the maximal potential stores of live and detrital (i.e., dead) carbon. Both RegionCarb and StandCarb are parametrized and corroborated with the aid of ground-based observations.

At the heart of our modeling strategy is the LandCarb model (Wallin

[1]W. B. Cohen, M. Fiorella, E. Helmer, J. Gray, and K. Anderson, 1996, manuscript in review.

Table 1. Percent of forest area (812,000 ha) in carbon flux classes. Percentages are given as a function of harvest status (Figure 5) and 1991 forest cover condition derived from Landsat imagery. Negative flux classes are carbon sinks and positive flux classes are carbon sources to the atmosphere.

	Carbon flux class (10^6 g C $\cdot$ ha^{-1} $\cdot$ yr^{-1})								
Vegetation condition	-5.0 to -4.0	-4.0 to -2.0	-2.0 to -0.25	-0.25 to +0.25	+0.25 to +2.0	+2.0 to +4.0	+4.0 to +8.0	+8.0 to +16.0	Total
Harvest status									
Harvest	—	—	—	0.4	0.2	2.1	6.3	6.3	15.3
No harvest	2.7	14.7	15.1	27.2	3.8	13.8	7.4	0.0	84.7
Total	2.7	14.7	15.1	27.6	4.0	15.9	13.7	6.3	100.0
Forest cover condition									
Early-successional	—	—	—	0.4	4.0	15.9	13.7	6.3	40.3
Conifer less than 80 years old	2.7	14.7	2.0	—	—	—	—	—	19.4
Conifer 80–200 years old	—	—	13.1	—	—	—	—	—	13.1
Conifer more than 200 years old	—	—	—	27.2	—	—	—	—	27.2
Total	2.7	14.7	15.1	27.6	4.0	15.9	13.7	6.3	100.0

et al. in press). LandCarb takes the estimated production and decomposition rates from StandCarb and the estimated production potential from RegionCarb (Figure 1), calculated for each resolution cell of the raster vegetation cover maps, to parametrize functions describing live and detrital stores over time. Live carbon stores calculations rely on a Chapman–Richards function (Richards 1959) describing the change in total live carbon stores (including leaves, branches, tree boles, and roots) in relation to time since major disturbance (i.e., forest stand age). To describe changes in detrital carbon stores, a function is used that accepts inputs from the live pools (i.e., newly dead material) and harvest disturbance and calculates losses from decomposition and site preparation after harvest (e.g., burning). Decomposition is a function of site factors (defined by GIS data bases) and type of pool (e.g., leaf matter, coarse woody debris). For our PNW site we have extensive ground data bases with which to parametrize the decomposition functions (e.g., Harmon 1992, Harmon and Hua 1991).

LandCarb uses the live and detrital carbon stores functions to create a pair of live and detrital carbon stores maps for each date examined (Figure 1). Forest harvest maps from each time interval are combined with the live carbon stores maps within the HARVEST model (Harmon et al. 1996), which accounts for historical changes in the mass of the detrital carbon pool left after harvest. This model incorporates data on utilization standards (e.g., stump height, minimum tree and top diameter), tree species, age, and size (which determine amount of decay, defect, and breakage). LandCarb takes the output from HARVEST to create raster-based maps of harvested carbon for each time interval. These maps of harvested carbon are then supplied to a model called FORPROD (Harmon et al. in press). FORPROD accounts for historical changes that have occurred in the distribution of wood through the forest products sector and for decay of the various forest products in use (e.g., buildings, fences, other wood structures, paper, and fuel) or as waste (e.g., landfills, incineration, or recycling). Output from the FORPROD model is used by LandCarb to create maps of forest products carbon stores for each date that a live and detrital carbon stores map is created. In this way, the estimated amounts of carbon in various forest products remain spatially referenced to the geographic locations from which they were derived and temporally linked to the period at which they originated.

The final stage of analysis consists of integration over time of the live, detrital, and products carbon stores maps (Figure 1). This integration is done for each of these three pools independently, by accumulating positive and negative changes in stores that occur through time for each resolution cell of the raster-based carbon stores maps. Accumulation through time in this manner results in three maps of change in carbon stores (ΔC-stores), one for each major pool. These three ΔC-stores maps are then integrated to produce the final, total net carbon flux map.

The mineral soil carbon pool can be influenced by forest management practices, but responses are site specific and can be positive or negative (Johnson 1992). Limited information for our study area about the temporal response of mineral soil carbon to harvesting, site preparation, and succession currently prevents us from including mineral soil carbon in our modeling strategy. Therefore, the mineral soil carbon pool is derived independently from the STATSGO soils data base (NSSC 1994) and soil pedon data (Homann et al. 1995) and is assumed to be constant for areas undergoing timber harvest. Incorporating changes in mineral soil carbon into future analyses could be advanced by rigorous field studies of harvesting effects, such as those carried out in other forest regions (Black and Harden 1995, C. E. Johnson 1995), as well as by identification of vegetation species by remote sensing and related means (because nitrogen-fixing species can enhance mineral soil carbon; Cole et al. 1995, Johnson 1992, D. W. Johnson 1995).

A modeling strategy that combines estimates from several different sources to predict regional carbon fluxes potentially results in errors that can be introduced in several ways, such as by mapping with remote sensing and parametrization of carbon models. (These are imperfect because each is derived from empirical relationships.) We are currently examining each of these errors and assessing how they propagate with increasing scale. This is a key issue because some errors may be insignificant as spatial scale increases, whereas others may become exacerbated. Our vegetation cover maps are assessed for uncertainties using a standard error matrix involving the comparison of reference data with map predictions (Cohen et al. 1995). In general, we obtain cover mapping accuracies in excess of 80%. Because harvest activity can be rather easily detected by spatial comparison of multi-date images, our harvest maps generally have accuracies in excess of 95%.[2] Uncertainties associated with parametrization of models are commonly addressed by sensitivity analysis, and these are ongoing for each major carbon model. Preliminary results suggest that our three most highly developed individual models (StandCarb, HARVEST, and FORPROD) provide reasonable estimates when compared with empirical observations. For the LandCarb model, which integrates all other models and observations, we are currently performing a sensitivity analysis that incorporates a Monte Carlo function to generate uncertainty estimates. In addition to providing assessments of uncertainty and error for remote sensing data and model components at the appropriate spatial and temporal scales, these analyses will provide information on sources of error in the final flux estimates.

Demonstration of modeling strategy

For our pilot study we estimated carbon flux over the 19-year period from 1972 to 1991 for a 1.2-million ha landscape of the PNW region in the central Oregon Cascades Range (Figure 2). This landscape was chosen to include all major site productivity classes and forest ownership and management categories. The derivation of vegetation cover maps from Landsat imagery is described by Cohen et al. (1994, 1995) and Wallin et al. (in press), and the derivation of harvest maps is documented in Cohen et al.[3] Mapped forest vegetation includes three early-successional mixed (hardwood and conifer) species cover classes that occur after severe disturbance (including open forest, with less than 30% cover; semi-closed forest, with 30%–85% cover; and closed forest, with more than 85% cover), and three classes of closed (more than 85% cover) conifer forest based on forest age (less than 80 years, 80–200 years, and more than 200 years). These six classes of forest succession are closely related to important time intervals associated with rates of change in live and detrital carbon stores predicted by the LandCarb model. Although some insect and fire disturbances were noted, the overwhelming type of disturbance during the time period under consideration was clearcut forest harvesting. Biogeoclimatic data bases have been obtained from several different sources to predict decomposition and production potentials, but for this pilot study we relied mainly on a digitized site productivity class map (Isaac 1949).

Of the total 1.2-million ha pilot study area, 812,000 ha were mapped as forest land in 1988 (see Cohen et al. 1995).[4] From the forests of this area, total net carbon flux during the 19-year period was $+17.5 \times 10^{12}$ g, a net source to the atmosphere. Across the study area, net carbon flux ranged between a sink of -4.7×10^6 g·ha^{-1}·yr^{-1} and a source of +15.8

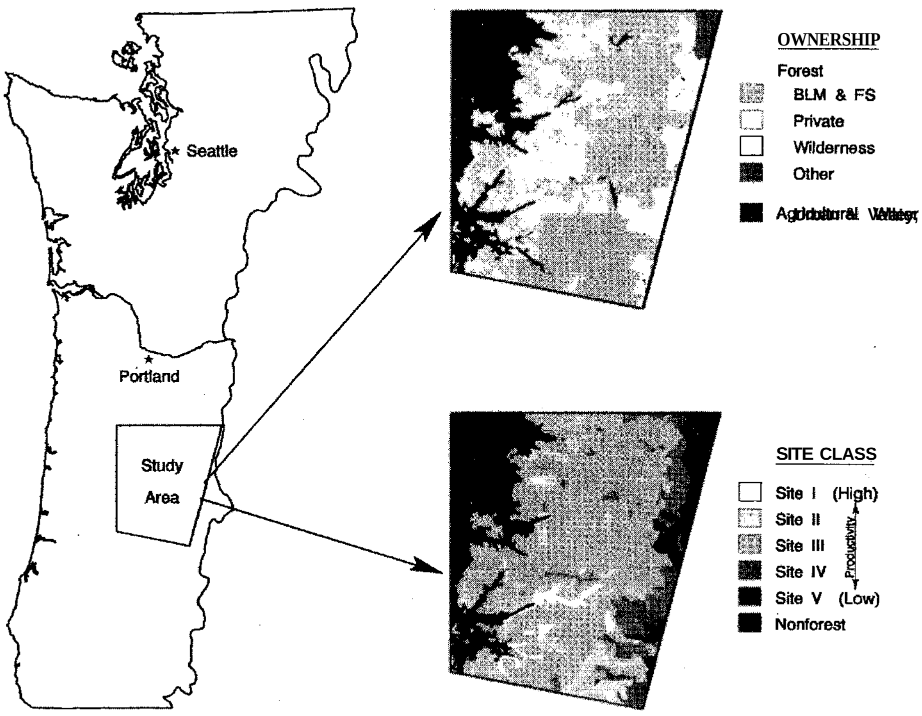

Figure 2. The Pacific Northwest (PNW) region of western Oregon and Washington, and the subset area examined in this study. The study area encompasses the full spectrum of land use (i.e., congressionally designated wilderness lands, federal lands managed under a multiple-use plan by the US Department of Agriculture's Forest Service [FS] and the US Department of the Interior's Bureau of Land Management [BLM], private industrial and nonindustrial forest land, and other lands managed for a variety of objectives) and site productivity conditions found in the region.

[2]See footnote 1.

[3]See footnote 1

[4]This is somewhat smaller than the actual amount reported by Cohen et al. (1995), as the site productivity data used here imposed additional restrictions on forest land definition.

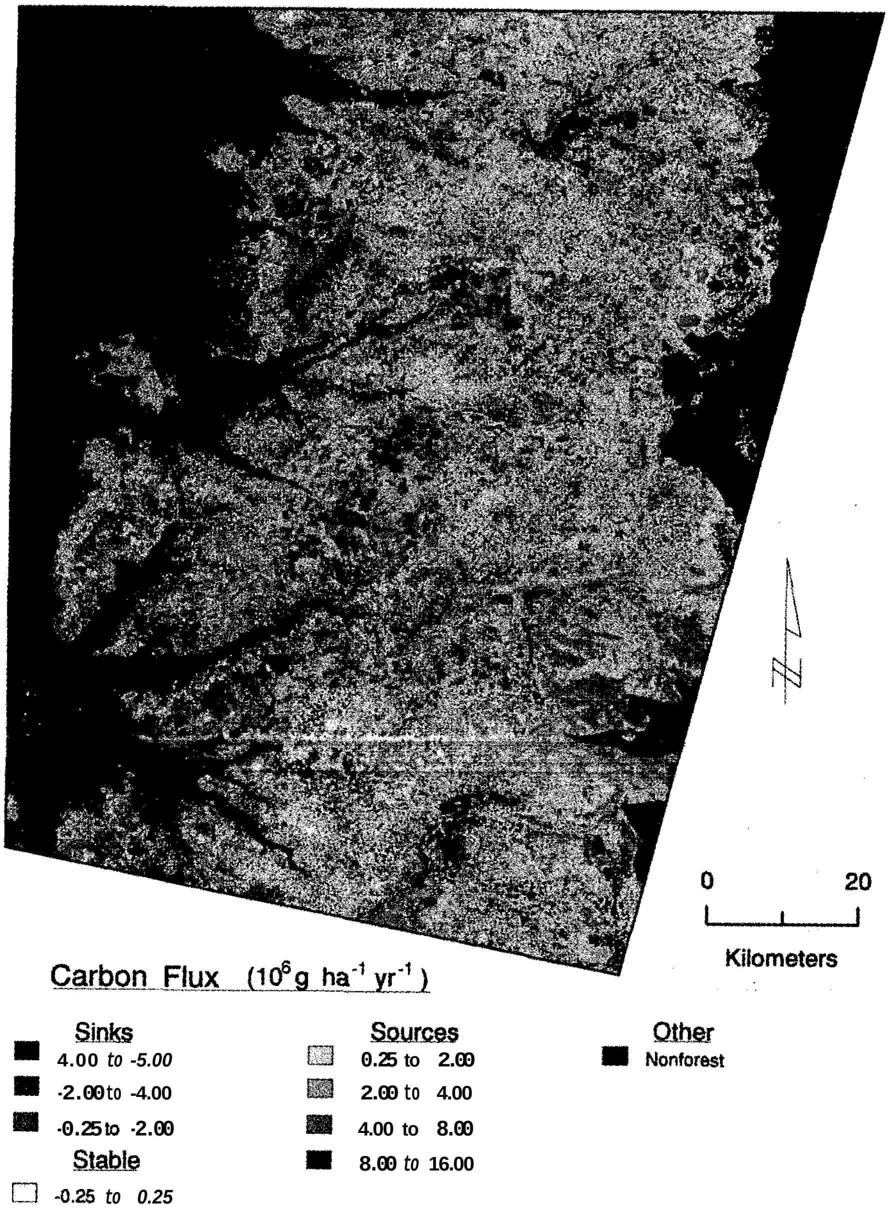

Figure 3. Net carbon flux over the subset study area between 1972 and 1991.

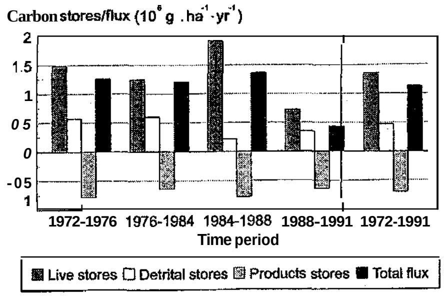

Figure 4. Carbon budget, by time period evaluated, for the subset study area. Negative values for individual pools represent increased carbon storage, and positive values represent decreases in storage. A positive total net carbon flux from the forest to the atmosphere was observed.

x 10^6 g . ha^{-1} . yr^{-1} (Figure 3). The average total net carbon flux was +1.13 x 10^6 g . ha^{-1} $\cdot$ yr^{-1}, with both the live and detrital pools contributing positively to the total net flux and forest products contributing negatively (Figure 4). The live pool was a positive contributor because more carbon was harvested than was sequestered via growth. Harvested carbon became detritus or forest products. The contribution of harvested carbon to the detrital pool was outweighed by decomposition and release to the atmosphere of this pool. Forest products were accumulating at a greater rate than they were decomposing. Actual contributions of these three major pools varied by time intervals from 1972 to 1991, but a given pool was either always a positive or negative contributor to the total net flux among time intervals.

Harvest activity was distributed widely across the study area (Figure 5) and occurred over 15.3% of the total forest area (Table 1). Of the total forest area, 32.5% was a net carbon sink, all represented by nonharvested stands. Most of the area with no change in carbon stores (a total of 27.6% of the forest area) was also in nonharvested forest. Except for the 0.4% of forest area that was harvested but stable, all harvested areas were a net source of carbon to the atmosphere. Curiously, although only 15.3% of the forest area was harvested during the 19-year period, a full 39.936 of the forest area was nonetheless a net carbon source to the atmosphere. Because we mapped harvest activity over'the 19-year study period, the minimal possible age for nonharvested stands is 20 years. This finding raises an important question: How can forests 20 years of age and greater be a source of carbon to the atmosphere?

The answer can be found in the complex relationship between forest succession and decomposition processes, harvest and other disturbance activity, and forest products manufacturing. Except in riparian zones, a closed canopy of mixed coniferous trees is the general cover condition of primary forest stands within the PNW region (Franklin and Dyrness 1988). The most recent severe, widespread fire or other natural disturbance activity in the study area was more than 450 years ago, with more recent localized, partial burning occurring at intermittent intervals (Agee 1991). Thus, if an area was never harvested, it most likely consists of a mature (80–200 years) or old-growth (more than 200 years) closed-canopy conifer (Franklin and Spies 1991). These mature and old-growth forests are generally slow growing, such that they are at most only a moderate carbon sink or stable (Grier and Logan 1977). This situa-

Table 2. Proportional distribution of forest conditions harvested. Harvested early-successional forest was mostly contained as small patches within harvested conifer forest.

Forest condition	Percent of total harvest
Early-successional	16.4
Young conifer	14.5
Mature conifer	30.9
Old-growth conifer	38.2

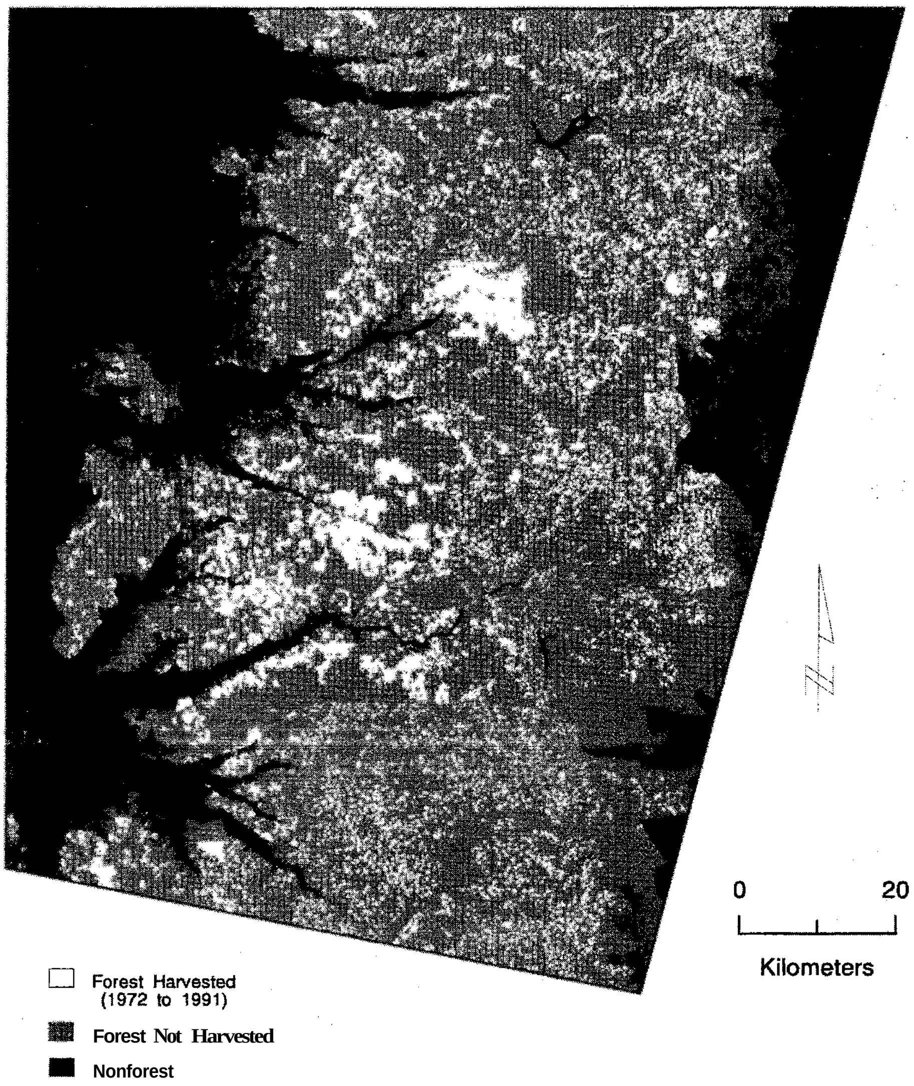

Figure 5. Clearcut harvest activity between 1972 and 1991 over the subset study area.

tion is revealed in Table 1, in which net carbon flux over the 19-year period is summarized by 1991 forest cover condition. After severe disturbance, such as clearcut logging, early-successional brush-dominated and other nonconifer cover conditions occur. But under ideal conditions, especially with plantation forestry, return to a closed-canopy conifer state is generally expected within 20 years, according to forestry professionals of the PNW region. By the time this young conifer state is reached, accumulation of biomass by live trees is expected to result in high rates of carbon sequestration (Long and Turner 1975, Turner and Long 1975). This latter point also is revealed in Table 1, in which our estimates show that young conifer stands were the largest net carbon sinks over the study period.

If all closed-canopy conifer stands were either a net carbon sink or carbon stable during the period of this study, then early-successional forest conditions must have been responsible for the full net carbon source observed. Table 1 clearly demonstrates this scenario. By superimposing the harvest map on top of the vegetation cover map of 1991 we discovered that much of the remaining early-successional forest was apparently clearcut before 1972. That these areas were clearcut is evident from the size, shape, and location of early-successional nonconifer forest stands not harvested during the study period (Figure 6).

There are several probable causes for early-successional, nonconifer stands remaining a net carbon source for an extended period. The most likely reason is that the predominant preharvest condition was mature or old-growth forest (Table 2). These forests commonly contain a large amount of fine and coarse woody debris that is not removed during or after harvest, but is left on site to decay slowly, at a rate of approximately 3% per year (Harmon et al. 1996). Furthermore, during harvest approximately 50% of the living biomass is converted to additional woody debris that decays on site (Harmon et al. in press). Of the carbon removed from the site and distributed throughout the forest products sector, approximately 40% quickly returns to the atmosphere due to losses during primary and secondary manufacturing, and to incineration and decomposition of short-lived forest products (Harmon et al. in press). The remaining forest products decompose slowly, at a rate of approximately 2% per year (Harmon et al. in press). Thus, even though early-successional nonconifer forests are sequestering carbon by accumulating living biomass, decay of carbon pools from the previous forest outweighs production by the new forest.

Significance to regional and global carbon cycle

The results from this pilot study will be relevant to the PNW region as a whole only if the study area is representative of the entire region. A comparison of a variety of data (both tabular and spatial) from several sources indicates that this is the case. Because many of the data sets used are county-level statistics, we com-

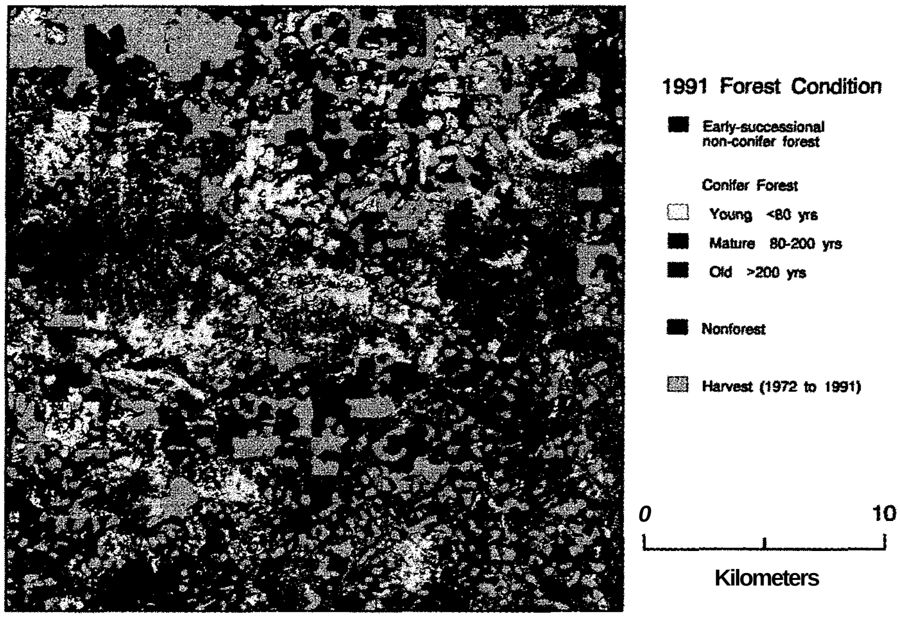

Figure **6.** Harvest activity between 1972 and 1991 (see Figure *5)* superimposed on the 1991 land cover map for a portion of the subset study area. The three early-successional classes mapped by Cohen et al. (1995) were all color-coded the same for display here.

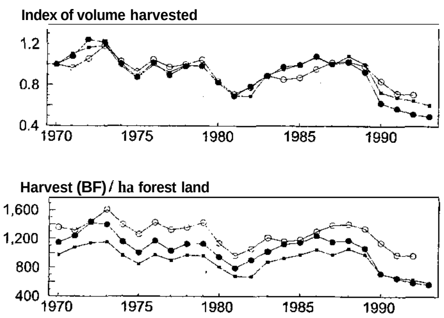

Figure **7.** Normalized forest harvest statistics from 1970 to 1993 for the three-county area represented by our subset study area (filled circles), and for western Oregon (squares) and western Washington (open circles). (Top) Index of volume harvested; (bottom) board feet of harvest per hectare of total forest land area. See text for further information.

pared data from the three counties (Lane, Linn, and Marion) of western Oregon in which our study area was largely contained to similar summary statistics from all counties of western Oregon and western Washington. Of the total land area within the three-county area, 82.7% is forest land. Comparable figures for western Oregon and western Washington are 86.2% and 63.0%, respectively. Normalized harvest statistics (i.e., all harvest volumes relative to 1970 levels) reveal that not only were trends in harvest activity similar among the three subregions, but relative harvest volumes were also comparable (Figure 7, top). A second comparison involved dividing actual harvest volume for each year (reported in terms of board feet [BF]) by total forest land area of the subregion to account for several differences in public versus private ownership. In terms of amount harvested per hectare of total forest land area, the three-county subregion had values between the western Oregon and western Washington provinces over most of the study period from 1972 to 1991 (Figure 7, bottom).

On balance, between 1972 and 1991 the forests of the pilot study area have been a net source to the atmosphere of 1.13×10^6 g C $\cdot$ ha^{-1} $\cdot$ yr^{-1} (Figure 4). A simple areal extrapolation of this average net flux to the 10.4 million forested ha of the full PNW region yields a total net flux from the region to the atmosphere of 11.8×10^{12} g C/yr. This regional flux estimate is lower than that of Harmon et al. (1990), who found that the amount ranged between $+15.3 \times 10^{12}$ g C/yr and $+18.5 \times 10^{12}$ g C/yr over the past 100 years. Our lower estimate for the more recent time period probably reflects the fact that during the last two decades some harvest activity occurred in second-growth forest on private lands and in less productive primary forests on public lands. Our extrapolated regional estimate indicates that the PNW region, although it represents only 0.25% of the total 4.1 billion ha of forest on Earth, was the source of 1.31% of the total recent land-use related carbon flux of $+0.9 \pm 0.4 \times 10^{15}$ g/yr on a global basis (Dixon et al. 1994). The most probable explanation for this disproportionate contribution is that forests of the PNW region generally store significantly more carbon than most other forest systems. Although replacing older forests with more vigorous young forest can increase sequestration by live carbon pools, decomposition of the large detrital pools after harvest greatly offsets gains in biomass by living pools for an extended period of time.

Our findings differ from those of Tans et al. (1990), who concluded that on a global basis, northern temperate forests are a net carbon sink. Their conclusion was supported by the studies by Kauppi et al. (1992) and Sedjo (1992), which found that in several regional forest systems around the globe, forests recovering from earlier harvest activity are now a carbon sink. However, as Houghton (1993) pointed out, these latter two studies failed to account for the fate of all major carbon pools; had they done so, they would likely have observed a net sink near zero.

Resource management implications

Managing global forest resources to enhance carbon sequestration is an important new emphasis in the management of regional and global carbon cycles (King 1993, Turner et al. 1993, 1995). In the PNW region, existing forest management plans for private holdings call for cutting on short rotation lengths that will not permit living and detrital biomass to accumulate after harvest to anywhere near the levels of mature and old-growth forests. Thus, privately owned forests are likely to continue to be net sources of carbon to the atmosphere as remaining primary and secondary forests are converted to new forest plantations. This expected net flux may be offset by carbon sinks on the region's federally owned land, which generally have greater rotation lengths on managed stands. Furthermore, recent political developments have resulted in much of the federal land base being excluded from harvest activity for the foreseeable future (FEMAT 1993). Because of the uncoordinated regional forest management plan that now exists for the PNW, it is difficult to predict the future role of the region's forest in the global carbon cycle.

Whatever the desired carbon management strategy for any given ownership category, management goals will best be realized only if a complete understanding of crucial carbon production and decomposition processes in the region is available. Several studies have addressed decomposition processes in the PNW region (Harmon and Hua 1991, Harmon et al. 1986, 1987), and a major long-term inter-regional decomposition study is currently underway (LIDET 1995). Although forest succession processes in the region are beginning to be understood (Acker et al. in press, Franklin and Dyrness 1988, McComb et al. 1994), the mechanisms are complex and interactive. Under the natural regeneration regime that was common before the 1970s, closed-canopy conifer forests were expected to emerge approximately 30–40 years after harvest. The now-common intensive forest planting regime, which involves immediate planting of improved genetic stock and timely hardwood and brush control, has narrowed the estimated time to closed-canopy conifer condition to as little as 20 years. Thus, barring regeneration failure, most forests currently in an early-successional condition due to harvest activity are expected to return to closed-canopy conifer condition within the next two decades. However, many harvested forest stands in the region appear to have remained in early-successional brush fields for extended periods of time, even under intensive reforestation regimes (Haynes 1986). In an ongoing study, we found that factors such as site condition, solar radiation regime, elevation, competition from nonconifer species, site preparation, and reforestation regimes play important roles in successional rates and pathways. Interactions among these factors alter successional pathways in ways that we are just beginning to understand. Developing a better understanding of these controls on forest succession is crucial for predicting how forest management will influence the region's carbon budget in the future.

Understanding and managing the global carbon budget is an important new focus of cooperative international science and politics. Many strategies exist for future carbon sequestration on a global scale, and these are largely dependent on forest management practices (Dixon et al. 1994, Joyce et al. 1990, Turner et al. 1993). Because regional productivity and land-use patterns are highly variable, sound scientific studies need to be conducted to better understand and model controls on the carbon cycles of other important forest regions. This article demonstrates the potential in such analyses of linking carbon models to remote sensing in a geographic information system and of accounting for all major forest carbon pools. Additional rigorous regional studies should help refine the global carbon budget.

Acknowledgments

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