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Use of Remote Sensing to Model Land Use Effects on Carbon Flux in Forests of the Pacific Northwest, USA

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Reducing the uncertainty in the global carbon (C) budget will require better information on regional C budgets. We discuss the use of a simple "metamodel," in conjunction with satellite data, to quantify C flux from a 12,000-km^2 forestland study area in Oregon. The model tracks C storage in living, detrital and forest products pools. Between 1972 and 1991, total C flux from this study area to the atmosphere was estimated to average 1.13 Mg ha^{-1} yr^{-1}, with values ranging from -4.7 to +15.8 Mg ha^{-1} yr^{-1}. This spatial variability was related to site quality, land use and historical factors. These results are used to illustrate the natural and anthropogenic sources of heterogeneity that can influence C budgets at the regional scale and to demonstrate how remotely sensed data can be used to help quantify this heterogeneity.

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Introduction

The potential for substantial climate change resulting from increasing atmospheric concentrations of radiatively active trace gases has motivated efforts to identify natural and anthropogenic sources and sinks for these gases (Houghton and Woodwell 1989, Post et al. 1990). Among these radiatively active trace gases, carbon dioxide (CO_2) has the greatest potential to affect global climate; hence, the global carbon (C) budget has received special attention (IPCC 1990). Recent global-scale estimates (Sundquist 1993, Houghton et al. 1992) suggest that emissions of CO_2 to the atmosphere from the combustion of fossil fuels and land use account for 5.4 ± 0.5 and 1.6 ± 1.0 Pg C yr^{-1}, respectively. Dixon et al. (1994) recently reduced the estimate of land use emissions to 0.9 ± 0.4 Pg C yr^{-1}. The oceans are thought to be a sink for 2.0 ± 0.8 Pg C yr^{-1}, while observed increases in atmospheric CO_2 concentrations account for about 3.2 ± 0.1 Pg C yr^{-1}. This observed increase and the known sources leave a "missing sink" in the biosphere of about 1.1 ± 1.0 Pg C yr^{-1}. The search for this missing sink is at the core of much of the current work on global C budgets.

The certainty in the measurements of various global C pools and fluxes varies widely. There have been economic incentives to carefully track the global consumption of fossil fuels; hence, there is more confidence in the estimates of fossil fuel emissions than for the other sources and sinks. Solomon et al. (1993) pointed out that the ocean and atmosphere systems are reasonably well mixed and that the measurement and modeling of C pools and fluxes between them is a tractable problem in physical chemistry and fluid dynamics. For this reason, the estimates of C flux between the ocean and atmosphere systems have remained within a fairly narrow (although perhaps not accurate) range of values over the past 20 years (Keeling 1973, Tans et al. 1990, Orr 1993). Solomon et al. (1993) went on to point out that the terrestrial system is fundamentally different from those of the ocean and the atmosphere in the sense that it is not at all well mixed with respect to C. In the terrestrial system, heterogeneity in C pools and fluxes exists at all spatial and temporal scales and results from a complex interaction of natural and anthropogenic factors. This heterogeneity has resulted in a wide range of estimates of terrestrial C sources and sinks.

The relative certainty in the other C pools and fluxes points towards a terrestrial C sink. Several lines of evidence suggest that the missing C sink is most likely to be found in northern, mid-latitude forests (Tans et al. 1990, Kauppi et al. 1992, Taylor and Lloyd 1992). Dixon et al. (1994) calculated that, if these forests are to account for the missing C in the global budget, their net accumulation rate must be about 1.5 Mg C yr^{-1}, or about 2–3% of their present standing stock. They pointed out that although this is within the range of values observed at selected sites, it is considerably higher than the most recent estimate for the continental United States (0.4 Mg C ha^{-1} yr^{-1}, Turner et al. 1995) and up to four times higher than the observed globally averaged rate of accumulation for northern, temperate forests (reviewed by Dixon et al. 1994).

Efforts to reduce the uncertainty in the global terrestrial C budget will require better information about the spatial and temporal heterogeneity of C storage and C flux in various regions throughout the world. In this chapter, we discuss some of the natural and anthropogenic sources of heterogeneity in forested ecosystems and demonstrate how remotely sensed data can be used to quantify this heterogeneity. Our discussion centers on forests because they contain about 60% of the global terrestrial C stocks (Waring and Schlesinger 1985). To provide a framework for discussion, we focus on our efforts to develop a C budget for the Pacific Northwest (PNW) forests of the United States. We describe a new model called LANDCARB that enables us to integrate a simplified stand-level C model with satellite imagery and spatial information on climate and soils. Classified satellite imagery is used to provide initial forest conditions and to track changes in forest structure resulting from succession and both natural and anthropogenic disturbance. Model output includes maps of C storage and C flux. Model results that quantify the C budget from 1972 to 1991 for a 12,000-km^2 study area in the central Oregon Cascade Mountain Range (Cohen et al. in review) are reviewed and used to illustrate our discussion. Our results are not consistent with the most recent reviews (Dixon et al. 1994, Turner et al. 1995), since we conclude that our study area was a large net source of C to the atmosphere between 1972 and 1991.

PNW forests

The forests of the PNW are among the most productive in the world and contain many tree species that attain great ages and substantial stature (Waring and Franklin 1979). Several tree species achieve ages of 500–1000 years in natural forests that may contain individual trees ≥ 100–200 cm in diameter at breast height (dbh) with heights of 60–80 m. These forests have the capacity to store very large quantities of C. An intensively studied, 450-yr-old Douglas-fir (*Pseudotsuga menziesii*) stand on moderately productive (site-class 3) land in the central Oregon Cascade Mountains contained 611 Mg C ha^{-1} in above- and below-ground living and detrital pools and in the mineral soil (Grier and Logan 1977, Harmon et al. 1986). Sixty-yr-old stands on comparable land contain between 259 and 274 Mg C ha^{-1} in these same pools (Harmon et al. 1990b).

Harmon et al. (1990b) showed that harvesting these old-growth forests and replacing them with young plantations would result in a large net release of C to the atmosphere, even when storage of C in forest products is considered. Although young plantations have a higher net annual rate of C uptake than old forests, the total amount of C stored in young plantations is minimal compared with that of old-growth stands. Harvesting an old-growth stand results in a large increase in the amount of dead wood on the site in the form of tops, branches and roots. Even if the decay rate remains constant after harvesting, the amount of C released to the atmosphere from this very large detrital pool is substantial during the first several decades after harvesting. Similarly, much of the C removed from the site by timber harvesting is quickly released to the atmosphere during primary and secondary manufacturing processes and by incineration and decomposition of short-lived forest products. Many decades are required before the net C accumulation rate by regenerating trees in plantation stands exceeds the net C emission rate to the atmosphere by the detrital pools and the forest products sector. It may take 200 years before the total amount of C stored by the stand approaches preharvest levels.

Although these stand-level dynamics are relatively well documented, developing a regional C budget requires the integration of these results in both the time and space domains. Integration in the spatial domain requires information on the distribution of stand ages, species, site productivity and management techniques. Integration in the time domain requires spatially explicit information on changes in stand age in response to timber harvesting, wildfire, succession and changes in management practices and forest product utilization standards. Assembling this information is a challenging interdisciplinary problem in ecology, silviculture, economics, history and social science.

Study area

Our study area is in western Oregon on the western slopes of the Cascade Range. It extends from an elevation of about 300 m on the edge of the Willamette Valley in the west to elevations of over 2000 m at the crest of the Cascade Mountain Range in the east (Plate 1). In general, the climate is characterized by mild, wet winters and warm, dry summers (Waring and Franklin 1979). However, the large elevation range and the steep, highly dissected terrain produces high variability in microclimate and forest productivity. Our study area lies within the Western Cascades and High Cascades physiographic provinces and mostly within the western hemlock (*Tsuga heterophylla*) vegetation zone (Franklin and Dyrness 1973). Dominant tree species include western hemlock, Douglas-fir and western red cedar (*Thuja plicata*). The area includes a wide variety of natural and managed stand types and ages.

Our study area includes 811,788 hectares of forested lands in a mixture of public and private ownership (Plate 1). The USDA Forest Service manages 53.5% of these forested lands including 7% in Congressionally designated wilderness areas. The Bureau of Land Management (BLM) administers 8.7% of the forested lands and 37.1% is in private ownership. Nearly all the private land is held by large corporations and is intensively managed for timber production.

Historical context

In the late nineteenth century, timber harvesting began throughout the PNW on the most productive lands at low elevations near the coast; in the early twentieth century, it began at lower elevations on the western slopes of the Cascade Range (Harris 1984, Robbins 1988). Timber harvesting on public lands throughout the region began in the 1940s. Harvest levels were modest until the 1960s and 1970s. During the period for which satellite data are available (1972 onward), virtually all of the cutting on public lands in our study area was in primary (previously uncut) forest and the cutting on private lands involved a mixture of primary and secondary forest (Bassett and Choate 1974, Gedney 1982, Gedney et al. 1987).

The primary land use for forest lands in this region is timber production, and the PNW has provided a significant share of the global timber supply for many decades (Waddell et al. 1989). Unlike timber harvesting in some parts of the world, this activity in the PNW is not the initial step in a permanent or semipermanent conversion from one cover type to another (i.e., conversion of forest land to agriculture or pasture). Douglas-fir is the dominant tree species in both primary and secondary forests.

Timber harvesting influences the regional C budget by altering the overall frequency distribution of stand age classes and their spatial distribution on the landscape. Under presettlement conditions, wildfire was the primary factor controlling the age-class distribution of these forests (Agee 1991, 1993). Although reconstructing the presettlement forest age-class distribution is technically problematic and politically contentious, it is clear that stands ≥ 200 years old were not uncommon (Spies and Franklin 1988, Booth 1991). In contrast, private lands throughout the region are currently managed on rotation lengths of about 50 years while federal lands have, until recently, been managed for rotation lengths of about 100 years (USDA Forest Service 1990, Spies et al. 1994). In future years, the public forest land will likely be managed on much longer rotations (FEMAT 1993). Since the private lands are located at lower elevations on the most productive land (site classes 1, 2 and 3, Plate 1), they have the largest capacity to store C. In contrast, the public lands are located at higher elevations on less productive land (site classes 3, 4 and 5, Plate 1) and have a lower capacity to store C. Rotation lengths currently in use have the effect of maintaining the private lands much farther below their maximum potential C storage levels than is the case for public lands (Krankina and Harmon 1994).

Changing forest practices over the past several decades must be considered when developing a C budget. Prior to the 1960s, there was greater reliance on natural regeneration after timber harvesting; this resulted in slower rates of C accumulation following a harvest. Currently, on both public and private lands throughout the region, stands are manually replanted with native tree species. On private lands, there tends to be a greater emphasis on replanting the more desirable commercial species and on the use of genetically "improved" varieties. On public lands, replanting includes a wider variety of species from locally collected seed sources. Timber harvesting and site preparation prior to replanting have varied considerably over the past 50 years. One of the most notable trends has been a reduction in the amount of woody debris left on the site after harvesting. During the 1910s, up to 75% of the woody material was left on the site after harvesting, compared with 36–54% during the 1980s (Harmon et al. in press). Additional changes on federal lands since the mid-1980s include reductions both in the use of burning prior to planting and in the use of herbicides to control shrubs and deciduous tree competition for the young conifers.

Over the last century, there have also been important changes in the ways harvested C is used in the PNW (Harmon et al. in review). These include changes in the disposition of saw and veneer mill waste as well as in the disposal of paper and wood waste. For example, until the 1940s most mill waste was incinerated without energy recovery. Since that time, an increasing share of mill waste (30–40%) has been used for paper production. Perhaps more significant have been the changes in paper waste disposal. Although paper in sanitary landfills decays very slowly, most paper waste was incinerated deliberately or accidentally in open dumps until the mid-1970s. Only since that time have landfills been a significant C sink for forest products (Harmon et al. in review).

The LANDCARB model

The development of C flux models for use at landscape (10^2 km^2) to regional (10^6 km^2) scales has involved several different approaches. One approach for coverage of large areas involves the use of complex, process-based models that operate in large grid cells (Running et al. 1989, Turner and Marks 1993). This approach is attractive in the sense that the process-based foundation makes it practical to evaluate interannual variation and ecosystem responses to climate change scenarios. The use of large grid cells is necessitated by the computational complexity of such models. In some cases, however, critical elements of spatial heterogeneity in the system may not be adequately represented by large grid cells. The use of smaller grid cells can alleviate this problem, but computational limitations will then reduce the spatial extent of the area that can be modeled. Another approach involves the use of a detailed model to generate a series of "look-up tables." Values from these look-up tables can then be used to produce maps of C storage for very large areas (Burke et al. 1990, 1991). We have chosen a "metamodel" approach: the linkage of a detailed stand-level model to a less detailed model that mimics its behavior (Law and Kelton 1991).

In our case, a detailed, stand-level C model (Harmon et al. 1990b) was used to generate "data" to parameterize simple functions that described the mass of C in living and dead pools of a forest. The original model, called Disturbed Forest Carbon (DFC) (Harmon et al. 1990b), simulates the accumulation of C in living pools (leaves, branches, boles, fine roots and coarse roots) following a disturbance and tracks this C as it cascades through various detrital pools (forest floor, fine woody debris, coarse woody debris and dead roots). The live portion of DFC was parameterized to mimic commonly used yield tables for Douglas-fir (Curtis et al. 1982). The input of detritus and its decay was parameterized from published (Harmon et al. 1986, Harmon and Chen 1991, Harmon et al. 1990a) and unpublished data (unpublished data, Oregon State University Forest Science Data Bank, Corvallis, OR). The functions that were parameterized by means of the DFC model were used to develop a less detailed and less computationally intensive metamodel that could be applied at landscape to regional scales using small grid cells. This model, called LANDCARB, was specifically designed to examine the effects of wildfire and land use on C budgets. The model was not designed to evaluate the direct (CO_2 fertilization) or indirect (climatic) effects of changes in atmospheric CO_2 concentrations on C budgets. Hence, the model evaluates the effects of land use under constant climate conditions with no CO_2 fertilization. LANDCARB was designed to be spatially explicit and to utilize input from satellite imagery and other spatial databases. The model is described in detail by Wallin et al. (unpublished manuscript); key features are briefly discussed in the following section.

Living carbon pools

A Chapman-Richards function was used in LANDCARB to describe the change in total live C as a function of time since disturbance:

$$\mathrm{LIVE} = \mathrm{LIVEMAX} * (1-e^{(-B_1*\mathrm{AGE})})^{B_2}$$

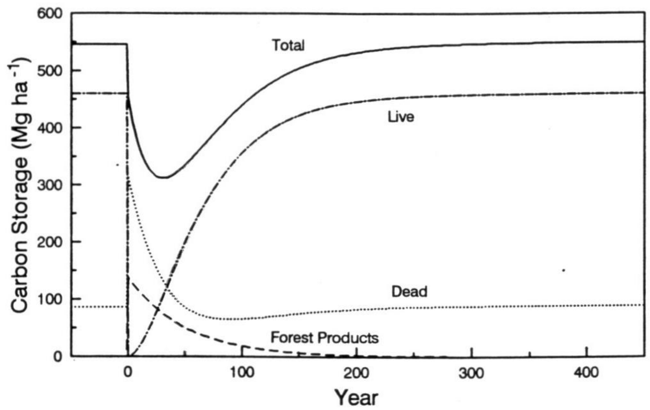

Figure 1. **Change in C storage at the stand level on site-class 3 lands. Curves illustrate response following the clearcut harvesting of an old-growth stand at time zero. Output is from the LANDCARB model. See text for details.**

where LIVE is the total live C stores in Mg C ha^{-1}, LIVEMAX is the maximum live C stores, B_1 is the rate that determines how quickly live C approaches the maximum and B_2 determines how long plant production lags behind the maximum rate (Fig. 1). AGE is the number of years since the last disturbance. Parameter values are presented in Table 1.

Table 1. **Parameter values for the Chapman-Richards live C function in LANDCARB.**

Site index	LIVEMAX Mg C ha^{-1}	B_1	B_2
1	650	0.021	1.98
2	570	0.021	1.97
3	460	0.021	1.96
4	310	0.020	1.92
5	230	0.020	1.88

Labile dead carbon pools

We did not include mineral soil C in our dead pool, because this pool is thought to change little following disturbance (Johnson 1992). Since we are primarily interested in C pools that are likely to change in response to wildfire or land use, we model only the labile dead C (forest floor, fine woody debris, coarse woody debris and dead roots). "Total dead C" could be obtained by adding an estimate of mineral soil C derived from the STATSGO database (Soil Conservation Service 1991).

The rate of change for the dead C pool is a function of the inputs from the living pool and harvesting disturbance minus the losses from decomposition and site preparation (i.e., residue burning). Both decomposition and input rates represent aggregated values for all components (leaves, small branches, boles, roots) of the dead or living pools, respectively (Table 2). At any given time step, the decay rate is the proportion of the dead labile pool that is lost to decomposition. Similarly, the litterfall rate is the proportion of the living pool that is added to the dead labile pool. With increasing site index (decreasing site productivity), leaf area index (LAI) changes very little, but bole and large root volume — and, therefore, total C storage — declines markedly (Table 1). This means that the more recalcitrant components of the dead labile pool (boles and large roots) make up a smaller proportion of the total dead labile pool. Although decay rates for each component of this pool generally decrease with increasing site index, the smaller volume of boles and large roots in the dead labile pool means that the aggregate decay rate actually increases slightly. With declining site productivity, boles and large roots also account for a declining proportion of the living C pool. Since the turnover of leaves, small branches and fine roots changes very little with site quality, the aggregate litterfall rate also increases with increasing site index. We assume that harvesting removes 50% of the live C and adds the remainder to the dead pool (Harmon et al. in press). Finally, for these simulations we assume that fire was not used for site preparation, so no reductions were made to the dead pool after harvesting.

Table 2. **Parameter values of the asymptotic decay function used in modeling labile dead C stores in LANDCARB.**

Site index	Decay rate yr^{-1}	Litterfall rate yr^{-1}
1	0.030	0.005
2	0.030	0.005
3	0.030	0.006
4	0.031	0.007
5	0.031	0.008

We initialized the dead labile C pool with estimates for stands with a range of ages, site classes and management histories (Table 3). Values used for this initialization were developed using the preceding equations and by assuming that stands ≥ 140 years old (in 1972) originated after a wildfire in an old-growth forest and stands ≤ 60 years old (in 1972) originated from clearcut logging of old-growth stands. Rates of C removal during harvesting are consistent with historical data (Harmon et al. in press). Values in Table 3 were used only to assign the initial dead labile C pool sizes for 1972. For all subsequent dates, dead labile C storage was calculated based on the model framework described previously and the observed changes in stand age.

Table 3. **Harvest removal rates and initial C stores in the labile dead C pool for various forest ages and site classes used in parameterizing LANDCARB.**

		Initial stores (Mg C ha^{-1})				
		Site class				
Stand age (yr)	Harvest removal (%)	1	2	3	4	5
5	50	349	308	252	178	137
15	50	258	228	187	140	112
25	50	194	172	141	116	97
60	40	104	92	79	78	75
140	0	94	82	78	73	62
200+	0	103	91	88	70	59

Forest products

The bole C that is harvested and removed from a site is tracked in LANDCARB through the forest products sector using an approach similar to that used by Harmon et al. (1990b). Overall, we assume that 40% of the harvested bole C is injected into the atmosphere immediately after harvesting. This is to account for losses during primary and secondary manufacturing and incineration of short-lived forest products. The remaining 60% is added to the forest products inventory and is subjected to an annual loss of 2% to account for decay of buildings, fences and other wooden structures as well as incidental losses of paper and other forest products.

LANDCARB behavior at the stand level

The behavior of the LANDCARB model for a single stand on site-class 3 land is illustrated in Figure 1. After the harvesting of an old-growth stand at time zero, there is a large increase in the size of the dead labile pool resulting from the large input of logging slash. The size of the dead labile pool then declines rapidly because of decay and the minimal litterfall inputs from the very small postharvest living pool. The dead labile pool drops to a minimum value at year 90 before climbing back to the equilibrium value as litterfall inputs from the living pool increase between years 100 and 200. The living C pool drops to zero at the time of harvesting and returns to 90% of the preharvest level (LIVEMAX) in year 140. The forest products pool is initialized at time zero with 30% of the C from the preharvest living pool (50% of the living pool is removed from the site during harvesting, 60% of which enters long-term forest products). The decay rate reduces this initial forest products pool by 50% within 34 years and by 90% within 112 years. Total C storage (live + dead + forest products) returns to 90% of the preharvest level in 136 years. Similar total C dynamics occur for stands in each of the other site classes (Fig. 2).

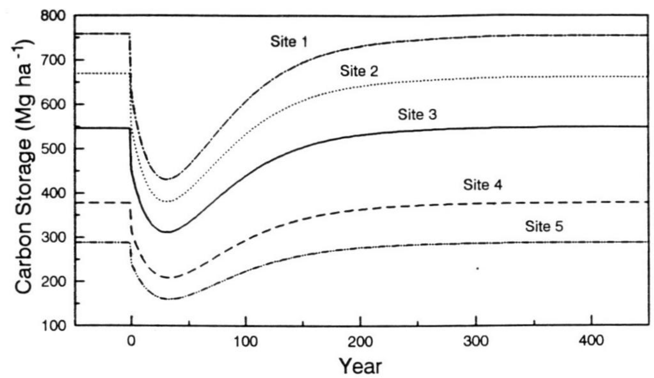

Figure 2. **Change in total C storage for stands on site-class 1–5 lands. Curves illustrate response following the clearcut harvesting of an old-growth stand at time zero. Output is from the LANDCARB model. See text for details.**

Model integration with spatial data

The equations describing C stores are dependent upon climate, soils and stand age. Calculations for an entire landscape require climate, soils and stand-age data for the entire study area. Constructing a C budget for a landscape requires information on changes in C storage between two or more points in time. Over the time scales considered here (19 years), this requires data on changes in stand age resulting from succession, timber harvesting and wildfire. Climate and soils are considered to remain constant over this period. We relied on the use of a site-class map (Isaac 1949) to integrate the influence of climate and soils on tree growth and decomposition rates (Plate 1). This map provides coverage for the entire region and corresponds well with observed patterns of forest productivity.

Natural and anthropogenic sources of heterogeneity

Maximum potential carbon storage

PNW forests provide an extreme example of how regional C densities can differ from the average continental-scale values used in global C budgets. For example, in their recent review of global terrestrial C budgets, Dixon et al. (1994) represented the continental United States using an average value of 170 Mg C ha^{-1} for the total C storage in all above- and below-ground living and detrital pools, including the O horizon and mineral soil (to a depth of 1 m). In contrast, old-growth forests on the most productive (site-class 1) lands in the PNW store an average of 753 Mg C ha^{-1}, exclusive of the mineral soil (Tables 1 and 3). Furthermore, even stands on the least productive (site-class 5) sites contain a minimum of 123 Mg C ha^{-1}, exclusive of the mineral soil (Fig. 2). The top 1 m of mineral soil on these site-class 5 lands would add an additional 104 Mg C ha^{-1} to total C storage (P. Homann, unpublished; derived from

Soil Conservation Service 1991). Thus, there is considerable potential for error propagation resulting from this regional- and local-scale variability.

Using satellite data to monitor changes in stand age

Much of the uncertainty in the C budget for forest systems results from inadequate knowledge of the forest age-class distribution and uncertainty as to the changes in this distribution over time. This uncertainty may exist even when forest inventory data are present. For example, a recent C budget for the continental United States (Turner et al. 1995) used inventory data to derive forest age-class distributions. For some land management categories in some parts of the country, such as USDA Forest Service lands in the Rocky Mountain region and throughout the eastern United States, Turner et al. (1995) assumed that the age-class distribution was similar to that on adjacent private lands. Since the age-class structure on private lands is often quite different from that on USDA Forest Service lands, this could lead to a significant error in the overall estimates of stores and fluxes. In another example, a C budget was constructed using data from an inventory of Canadian forests and another independent inventory of Canadian peatlands (Kurz et al. 1992). These data were not spatially explicit and the degree of overlap in forested peatlands could not be determined; this uncertainty could also have led to significant error.

The shortcomings of inventory data can potentially be overcome using satellite data. For example, within our study area, Cohen and Spies (1992) and Cohen et al. (1995) demonstrated that forest composition, stand age and structure can be mapped using Thematic Mapper (TM) data. Initially, they used unsupervised classification to separate four forest-cover classes: open (< 30%), semiopen (30–85%), closed-canopy mixed conifer-hardwood (> 85%) and closed-canopy conifer (> 85%). Then, statistical models were developed for distinguishing among successional stages within the closed-canopy conifer class. Three successional stages were identified: young (< 80 years old), mature (80–200 years old) and old-growth (> 200 years old). Accuracy of predictions for the three age classes was 75%, whereas an overall accuracy of 82% was achieved for all six forest-cover classes. All class boundaries were selected to represent ecologically meaningful transitions that could also be distinguished with a high degree of accuracy. We used these structural classes to represent "structural" age classes (Table 4). On average, these structural ages correspond to time since last disturbance; however, on suboptimal sites, regeneration may proceed slowly following a disturbance. In these cases, the structural age may lag behind the chronological age. These six structural age classes were then used to construct C storage maps. The number of years represented by these classes varied somewhat, with the highest resolution in the three youngest age classes. Fortuitously, these classes provide the highest temporal resolution in the portion of the successional trajectory where total C storage is changing most rapidly (Fig. 1).

Table 4. **Percentage of area by forest age class for 811,788 hectares of forested lands from 1972 to 1991. Age classes represent "structural age" (after Cohen et al. 1994); see text for explanation of structural age classes.**

Structure class	Age class (yr)	1972	1976	1984	1988	1991
Open	0–10	9.1	11.5	11.0	11.6	9.1
Semiopen	11–20	11.0	11.2	17.3	19.7	24.4
Closed-mix	21–30	6.8	6.7	5.7	5.9	5.8
Young	31–80	21.0	20.7	20.1	19.4	18.9
Mature	81–200	17.7	16.8	14.9	13.7	13.1
Old-growth	200+	34.4	33.1	31.1	29.9	28.8

Satellite data for our study area were acquired for 1972, 1976, 1984, 1988 and 1991. Imagery for the first two years came from the Landsat Multispectral Scanner (MSS), while imagery for the latter three years came from the Landsat TM sensor. The Cohen et al. (1995) age-class map for our study area was developed using the 1988 image. A second age-class map was developed independently using the 1972 MSS data. A change detection algorithm was used to develop maps of the area harvested during each of the four periods. Wildfires within the study area during the period from 1972 to 1991 were minimal and are ignored in this analysis. The 1972 and 1988 age-class maps and the timber-harvest maps were then used to derive age-class maps for 1976, 1984 and 1991 (described in Cohen et al. in review).

The largest declines in area between 1972 and 1991 were for the mature and old-growth age classes (Table 4). Since these are the age classes with the largest total C storage, harvesting of these sites results in the largest net release of C to the atmosphere (Fig. 2). Net loss of mature and old-growth forest area averaged 4618 ha^{-1} yr^{-1} (0.57% of the study area) during the period from 1972 to 1991.

Total carbon flux: Mass balance considerations

The LANDCARB model was used to produce a single map of total C storage for each of five years (1972, 1976, 1984, 1988 and 1991). For each of these years separate maps were also produced to illustrate storage patterns in the living, detrital and forest products C pools. Differencing these data layers over any two periods yields a map of C flux for the relevant pool. These C flux values are presented in Figure 3 and a map of total C flux for the study area over the period from 1972 to 1991 is presented in Plate 2.

Our primary interest was to quantify the net exchange of C between the atmosphere and all components of the terrestrial ecosystem (live, detrital and forest products). This total C flux over the period from 1972 to 1991 averaged a release (source) of 1.13 Mg C $ha^{-1}yr^{-1}$ from the terrestrial system to the atmosphere (Fig. 3). Many earlier C budget studies (Armentano and Ralston 1980, Moulton and Richards 1990, Kauppi et al.

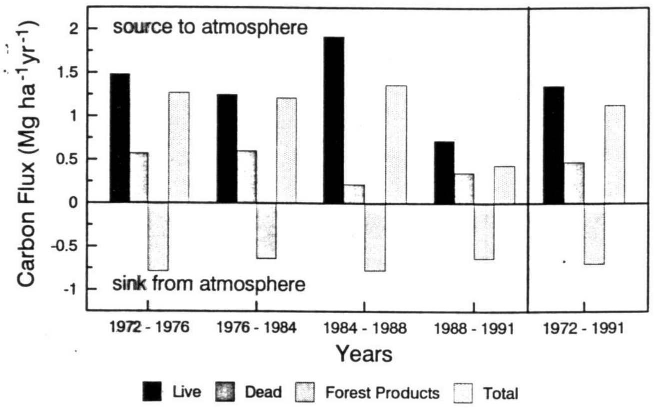

Figure 3. **Carbon budget for 811,789 hectares of forested land between 1972 and 1991. Total C flux from this area over this period was 17.5 Tg or 1.13 Mg ha^{-1} yr^{-1}.**

1992) did not adequately deal with the detrital flux or with the decomposition and incineration of forest products. Doing so involves ignoring mass balance and calculating ecosystem C flux as live flux (net growth) minus harvest. For our study area over the period from 1972 to 1991, such a calculation yields an average sink of 0.068 Mg C ha^{-1} yr^{-1}. This figure does not accurately represent the total net C flux for the terrestrial system. The failure of recent studies to adequately consider all components of the terrestrial C budget has added much confusion to an already challenging problem.

Temporal variability in carbon flux

Total C flux values for our study area ranged from a sink of 4.7 Mg C ha^{-1} yr^{-1} to a source of 15.8 Mg C ha^{-1} yr^{-1} (Table 5). The variability was determined primarily by which portion of a site's successional trajectory (Fig. 2) was captured during our 1972–1991 sampling period. Since young and mature stands store less C than old-growth stands, harvesting them results in a somewhat smaller release of C to the atmosphere. Over our 19-yr study interval, the largest C sources were old-growth stands on the most productive sites (site classes 1, 2 and 3) that were harvested between 1972 and 1976. Regardless of site class, harvested stands undergo their largest net loss of C (live, detrital and forest products) during the first 25–30 years after harvesting (Fig. 2). Hence, stands harvested at the beginning of our 19-yr study were near their minimum C storage levels in 1991. Within the next five to ten years, the net loss of C from these stands should stop. These stands will then begin to accumulate C, although it will require an additional 200 years for them to approach preharvest C storage levels. Stands harvested towards the end of our study were also sources but have not yet had time to release as much C to the atmosphere. Stands harvested prior to 1972 were somewhat smaller sources since they had already lost

Table 5. **Overall (a) frequency distribution of C flux values and distributions by ownership (b) and site index (c). Values in the first line of data represent C flux in absolute amounts (Mg C ha^{-1} yr^{-1}). All other values in panels (a), (b) and (c) are expressed as percentages of the 811,788-ha forested study area.***

	(a) C flux, 1972–1991 (Mg C ha^{-1} yr^{-1})									
	<— Sinks			—>	<— Sources			—>		
	-5.0	-4.0	-2.0	-0.25	0.25	2.0	4.0	8.0	16.0	Total
Overall	2.7	14.7	15.1	27.6	4.0	15.9	13.6	6.4		100.0
	(b) By ownership									
BLM	0.3	1.8	1.2	2.0	0.4	1.4	1.2	0.5		8.7
Forest Service	1.1	4.6	7.4	16.6	1.3	7.0	5.8	2.6		46.5
Private	1.3	7.9	3.9	5.2	2.2	6.9	6.5	3.2		37.1
Wilderness	0.0	0.3	2.5	3.6	0.0	0.4	0.1	0.0		7.0
	(c) By site index class									
Site 1	0.6	0.0	0.4	0.7	0.2	0.2	0.6	0.3		3.0
Site 2	2.1	0.0	1.2	2.4	0.4	0.4	2.4	0.7		9.7
Site 3	0.0	14.7	9.4	18.8	3.1	13.2	9.9	5.3		74.4
Site 4	0.0	0.0	3.1	3.7	0.2	1.3	0.6	0.1		9.0
Site 5	0.0	0.0	1.1	1.9	0.1	0.8	0.2	0.0		4.0

* *Note: Last columns in panels (b) and (c) do not total 100% due to round-off error.*

some of their original C prior to the beginning of our study. Stands harvested more than about 30 years before 1972 were the biggest C sinks in our study area. These stands were near their minimum total C storage levels in 1972 (Figs. 1 and 2) and then rapidly accumulated C throughout our study period.

Effect of management on carbon flux

C flux is strongly influenced by land management (Table 5). Nearly all of the wilderness areas in our study were either simulated to be in equilibrium or were small sinks for atmospheric C. The small source areas in the wilderness appear to have resulted either from natural disturbances or from windthrow or slash fires that burned into a wilderness from adjacent land. A majority of the USDA Forest Service-managed lands were also either in equilibrium or were small sinks for C. USDA Forest Service-managed lands also included some area as either large sources or large sinks; however, there was a higher proportion of BLM-administered and privately held lands in these C flux classes. These results are consistent with the observation that the conversion from primary to secondary forest was occurring more rapidly on the BLM-managed and privately held lands than on USDA Forest Service-managed lands (Spies et al. 1994, Cohen et al. 1995).

Conclusions

The results presented here illustrate how remotely sensed data, used in conjunction with other spatial data and a metamodel, can be used to quantify the exchange of C between the atmosphere and terrestrial ecosystems. The spatial and temporal heterogeneity in the terrestrial system makes it necessary to independently model C flux at a very large number of points on the ground (Solomon et al. 1993). This requires detailed data on forest condition and change. Remote sensing offers the only practical means to systematically map and monitor changes in forest condition over large areas and across various land-ownership categories.

The use of simple metamodels can provide a powerful means to quantify ecosystem behavior at large spatial scales. The development of complex models can often serve as a useful heuristic device and complexity is often necessary to adequately capture the dynamics of small reference stands. At larger spatial scales, the idiosyncratic behavior of individual stands may become less important as the goal shifts to describing the behavior of the larger system. If new driving factors do not come into play at this larger spatial scale (O'Neill et al. 1986), the average behavior of a large number of stands can be described by means of a simplified metamodel. The computational simplicity of a metamodel makes it possible to examine the response of a very large area using relatively small grid cells.

Model validation always presents significant challenges and is often particularly problematic when the aim is to model the response of large geographic areas over several decades. Although we have discussed the problems that may result from the use of inventory data to drive ecosystem C models, these data can be invaluable for model validation. We are compiling timber-harvest data for use in model validation. Data on the volume of timber harvested annually on both public and private lands are routinely compiled at the county level. These data are also compiled for subcounty administrative units on public lands. These harvest records provide an independent check of our estimates of harvested C, which, in turn, will provide an indirect assessment of the accuracy of living and detrital pools.

We conclude that our study area has been a substantial source for atmospheric CO_2 during the period from 1972 to 1991 (Fig. 3, Plate 2 and Table 5). This conclusion is not consistent with continental- and global-scale studies (the latter reviewed in Dixon et al. 1994), which conclude that northern, temperate forests have been C sinks over this period. Additional work will be required to determine whether the results from our study area are representative of the region as a whole and to predict the C budget for this region in the future. Our work demonstrates that forest management practices can have a major impact on regional C budgets. If management plans currently under consideration for public lands (FEMAT 1993) are implemented, little primary forest will be harvested during the coming years. On federally managed public lands where harvesting will continue, rotation lengths approaching 200 years will frequently be used. Proposed harvesting guidelines will result in reductions in the amount of C removed from the harvest site through retention of live trees and coarse woody debris. This practice also has been increasingly emphasized on private lands in recent years. Improved silvicultural practices on both public and private lands also are resulting in better regeneration success. With respect to forest products, improved utilization standards both in the manufacturing sector and in the marketplace are likely to reduce the proportion of harvested C that goes into short-term storage pools. Although considerable work remains to be done, these trends suggest that the amount of C released from PNW forests may be substantially reduced in the coming years.

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