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MODELING CARBON STORES IN OREGON AND WASHINGTON FOREST PRODUCTS: 1900–1992

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Abstract. A new model, FORPROD, for estimating the carbon stored in forest products, considers both the manufacture of the raw logs into products and the fate of the products during use and disposal. Data for historical patterns of harvest, manufacturing efficiencies, and product use and disposal were used for estimating the accumulation of carbon in Oregon and Washington forest products from 1900 to 1992. Pools examined were long- and short-term structures, paper supplies, mulch, open dumps, and landfills. The analysis indicated that of the 1,692 Tg of carbon harvested during the selected period, only 396 Tg, or 23%, is currently stored. Long-term structures and landfills contain the largest fraction of that store, holding 74% and 20%, respectively. Landfills currently have the highest rates of accumulation, but total landfill stores are relatively low because they have been used only in the last 40 years. Most carbon release has occurred during manufacturing, 45% to 60% lost to the atmosphere, depending upon the year. Sensitivity analyses of the effects of recycling, landfill decomposition, and replacement rates of long-term structures indicate that changing these parameters by a factor of two changes the estimated fraction of total carbon stored less than 2%.

1. Introduction

Eighteen years after Baes et al. (1977) first posed the question, uncertainty remains about the role of terrestrial biota in the global carbon cycle. On one hand, reconstruction of past land-use change indicates that the terrestrial biota is a net source of 0.4 to 2.6 Pg (Pg $= 10^{15}$ g) C year^{-1} (Dale et al., 1991; Dixon et al., 1994). On the other hand, 'deconvolution' studies (which estimate terrestrial flux by subtracting atmospheric increases and ocean uptake from the efflux of fossil fuels) indicate that the terrestrial biota is currently a small sink of less than 0.3 Pg (Post et al., 1990).

This discrepancy of 0.7–2.9 Pg C year^{-1} could be caused by several factors. First, uncertainty remains about the carbon uptake rate of oceans (Post et al., 1990; Tans et al., 1990; Watson et al., 1991). Second, major uncertainties also remain concerning land-use estimates. Some studies have indicated that carbon flux from non-tropical forests is close to being balanced (Houghton et al., 1987), others that some non-tropical areas may be carbon sinks (Kauppi et al., 1992; Kurtz et al., 1992; Turner et al., 1993). The differing estimates may result from the different definitions of the aerial extent of ecosystems, and different data for disturbance rates, carbon stores associated with living biomass (Brown et al., 1989), and soil carbon (Schlesinger, 1984; Post et al., 1982). Carbon stores in many ecosystems may change, as when fuel accumulates after fire suppression, without a change

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in land cover-type (Brown et al., 1991; Houghton, 1991). Finally, estimates of atmospheric fluxes may differ because major pools such as soil, woody slash, and forest products are treated inconsistently.

While many earlier studies have provided insight into ecosystem factors controlling carbon balance, they cannot be used for estimating atmospheric fluxes because they exclude forest products (Armentanto and Ralston, 1980; Cooper, 1983; Cropper and Ewel, 1987) or they have modeled them in a simple fashion with a constant rate of product loss to the atmosphere (Houghton et al., 1983; Harmon et al., 1990; Dewar, 1991; Hall and Uhlig, 1991; Kurz et al., 1992). The latter analyses, while more inclusive, contain many uncertainties and do not present the basis for determining the rates of manufacturing efficiency and forest-product life spans. Harmon et al. (1990) calculated the mix of products from published conversion factors that describe the flow of raw materials through the manufacturing process. Kurtz et al. (1992) used a similar approach to determine the mix and then modeled the long-term accumulation of these materials. Despite increased realism, neither of these two studies allowed for changes in manufacturing efficiencies, product use, or disposal over time.

To refine estimates of the carbon flux associated with land-use change, we have developed an analysis system that historically reconstructs the flow of carbon into and out of forest products. This paper describes this new model which is called FORPROD (Forest Products). While FORPROD is currently applied to the Pacific Northwest, it can be used in any region where basic timber-harvest and manufacturing data are available. FORPROD is part of a larger study designed to estimate the effect of land-use change and timber harvest on the carbon balance of Oregon and Washington. It estimates stores of carbon in forest products as part of the larger system of models that predicts changes in carbon stores within the forest ecosystem after timber harvest (Cohen et al., 1992, 1994). FORPROD considers (1) the amount of raw material (i.e., logs) that is converted to products (e.g., lumber) during manufacturing, and (2) the accumulation of forest products as they are used or disposed. Products considered by the model are lumber, plywood, paper (including paper board), mulch, and fuel. The fate of these major forest products in use as short- and long-term structures, paper supplies, mulch, open dumps, and landfills is followed. Processes considered during use are decomposition, replacement of structures, and recovery and recycling of disposed paper and wood into new products. The data come from Oregon and Washington, which have produced approximately 20% of the forest products in the United States for the last half century (Powell et al., 1993).

First we give an overview of harvest and manufacturing – the sources of data, assumptions about them, and conversions. Second, we describe the model. Third, the parameters tested in sensitivity analyses are discussed. We were particularly interested in the sensitivity of the model to historical change because such change has commonly been ignored in past studies. Given that manufacturing efficiency and the longevity of forest products and wastes have generally increased with time, carbon release may be substantially underestimated if the parameters are defined only by the most recent period. Fourth, harvested carbon is tracked through manufacture of products and disposal. Finally, we use the model to make a preliminary estimate of the carbon that has been stored in forest products produced in Oregon and Washington from 1900 to 1992.

2. Harvest and Manufacturing Overview

Our analysis included only the fate of logs harvested for industrial purposes within Oregon and Washington and not wood harvested for firewood, despite the fact that such carbon is rapidly released to the atmosphere. There are few statistics on the volume of firewood, as it is generally harvested on a small scale (i.e., for individual households). We also did not consider the fate of logs imported to Oregon and Washington for manufacturing. Our assessment of the effect of timber harvest on carbon sequestration in the two states is designed to couple changes in the forest ecosystem to the fate of the forest products, and to solve the flux to the atmosphere by mass balance. For this approach to work, however, we must consider a closed system; inclusion of carbon harvested outside the location of interest would create an open system that could not be internally balanced. Finally, we did not include the use of fossil fuels for harvesting and processing carbon into forest products. The release of fossil-fuel carbon is usually considered separately from releases related to land-use (Houghton et al., 1983; Dewar, 1991; Hall and Uhlig, 1991); we therefore follow this convention and consider only the fate of carbon produced within the forest ecosystem.

The model first converts harvested tree volumes to carbon and then estimates the fraction of raw materials converted to forest products. These values are then used by the carbon-stores portion of FORPROD to estimate the input rates to the various forest-product pools. The model can be used in two modes, one with a constant rate of manufacturing efficiency, the other with a time series of changing rates of manufacturing efficiency. In the standard simulation we used the latter approach.

2.1. HARVEST OF RAW MATERIALS

Predicting the mass of forest products produced for a given year first requires that the volume of boles harvested be entered into FORPROD. We therefore compiled published harvest statistics for Oregon and Washington from 1900 to 1992 (Johnson, 1941a, b; Moravets, 1949a, b; Wall, 1972; Warren, 1993). As FORPROD does not consider the fate of logs used for firewood, we did not include firewood in the analysis. Besides historical records, FORPROD can also use output from land-use models for the volume of trees removed from forests.

As FORPROD tracks the fate of carbon, the reported units of wood volume must be converted into carbon. The first step was to convert Scribner board feet to total cubic-foot volume (ft^3), which required data for the mixture of species and the size of the logs for the most exact conversion factors (Hartman et al., 1976). Unfortunately these data are not reported with harvest statistics, and therefore the conversion factors had to be approximated by regressing the reported cubic-foot volume of growing stock against the reported board-foot volume of saw timber (Bassett and Oswald, 1983; Gedney et al., 1986a, b, 1987, 1989; MacLean et al., 1992). That analysis gave an average conversion factor for common conifer species in Oregon and Washington of 0.234, within a range of 0.221 to 0.265, depending upon species. We used an average conversion factor weighted by the growing stock volume of each species reported in recent timber surveys (Gedney et al., 1986a, b, 1987, 1989; MacLean et al., 1992). The equation used to convert Scribner board foot volume (VolSbft) to cubic-foot volume (Volcft) was:

$$\text{Volcft} = 0.234 * \text{VolSbft}$$

The cubic-foot volume of wood harvested was then converted to the total cubic-meter volume (Volcm) by:

$$\text{Volcm} = 0.028 * \text{Volcft}.$$

The mass of organic matter harvested as wood (OGMWood) was calculated by multiplying cubic-meter volume by wood density (DenWood) of the major species:

$$\text{OGMWood} = \text{DenWood} * \text{VolCm}.$$

The density for current forest conditions was calculated by weighting the wood density of each species (Maeglin and Wahlgren, 1972) by the proportion of the growing stock it comprised in recent timber surveys (Gedney et al., 1986a, b, 1987, 1989; Maclean et al., 1992). The mean density of logs harvested east of the Cascade Mountains was 0.40 Mg m^{-3}; west of the Cascade Mountains it was 0.43 Mg m^{-3}. We then calculated density for earlier periods, finding that it has changed little over the last 50–60 years: 0.435 Mg m^{-3} for west-side forests (Andrews and Cowlin, 1934) and 0.389 Mg m^{-3} for east-side forests (Cowlin and Wyckoff, 1944). Finally, the mass of organic matter of wood (OGMWood) was multiplied by 0.52, the carbon content of coniferous wood, to convert the carbon mass of wood (CWood) (Wilson et al., 1987; Birdsey, 1992):

$$\text{CWood} = 0.52 * \text{OGMWood}.$$

2.2. LOG DISPOSITION

Once harvested, Oregon and Washington trees are used chiefly as saw logs for lumber production, veneer logs for plywood production, and pulp logs for paper

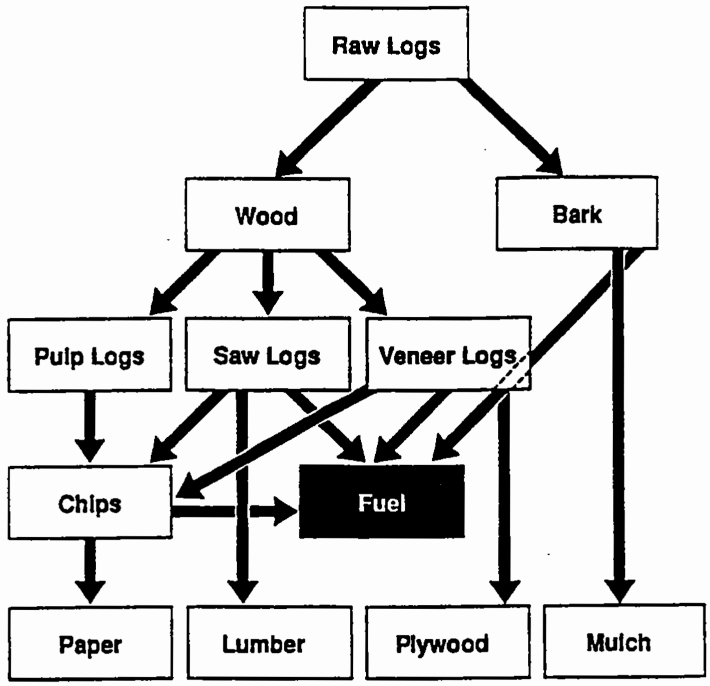

Figure 1. FORPROD flow of carbon through harvest and manufacturing. Fuels are released to the atmosphere in the year of harvest.

production (Figure 1). We excluded other minor uses, such as for railroad ties and poles (< 1% of total). We assumed that exported logs were used in the same way as logs used within the United States.

The equations for calculating carbon mass in saw logs (SawLog), veneer logs (VenLog), and pulp logs (PulpLog) were:

$$\text{SawLog} = \text{FSawLog} * \text{CWood}$$

$$\text{VenLog} = \text{FVenLog} * \text{CWood}$$

$$\text{PulpLog} = \text{FPulpLog} * \text{CWood}.$$

where FSawLog, FVenLog, and FPulpLog are the fractions of each used in any given year.

Changes in the use of logs over time was compiled from published harvest reports (Moravets, 1950; Gedney, 1956; Cowlin and Forster, 1965; Manock et al., 1970; Schuldt and Howard, 1974; Bergvall et al., 1975; Howard, 1984; Howard and Ward, 1988; Larsen, 1990, 1992; Howard and Ward, 1991). In years in which there were no reports, we used linear interpolation for estimating values.

2.3. BARK REMOVAL AND PROCESSING

Before logs are used for lumber, plywood, or pulp production, the bark is removed. The mass of carbon in bark (CBark) was calculated as:

$$\text{CBark} = \text{FBark} * (\text{CWood}/(1 - \text{FBark}))$$

where FBark is the fraction of logs that is bark. The fraction varies among species (Wilson et al., 1987), so averages were calculated by multiplying the portion of growing stock of a species by the fraction in bark. The values derived were 15% for logs east of the Cascade Mountains, 12% for logs west of the mountains.

Bark is currently used for mulch (BMulch), hogged fuel (BFuel), and chips (BChips) which were calculated as:

$$\text{BMulch} = \text{FMulch} * \text{CBark}$$

$$\text{BFuel} = \text{FBFuel} * \text{CBark}$$

$$\text{BChips} = \text{FBChips} * \text{CBark}$$

where FMulch, FBFuel, and FBChips are the fractions of bark being used for each in any given year. Historical patterns of bark use were compiled from the literature (Corder et al., 1972; Schuldt and Howard, 1974; Bergvall et al., 1975; Howard and Hiserote, 1978; Howard, 1984; Howard and Ward, 1988, 1991; Larsen, 1990, 1992). Linear interpolation was used when data were missing. We assumed that before 1960, when reporting began, bark was primarily used as fuel.

2.4. LUMBER PRODUCTION

The primary products produced from saw logs are lumber, chips for paper production, and hogged fuel. The rest is disposed waste. During lumber production, a large fraction of wood waste is generated in the form of slabs, sawdust, planer shavings, and defective lumber. We assumed that this material was disposed of as either chips or hogged fuel. In reality, some of it was either decomposed or incinerated without energy recovery. As the consequences for carbon stores of these two processes were similar to consequences for hogged fuels, we combined the three flows into a fuel category.

The production of lumber (Lumber), chips (SLChip), and hogged fuel (SLFuel) from saw logs was calculated as:

$$\text{Lumber} = \text{FLumber} * \text{SawLog}$$

$$\text{SLChip} = \text{FSLChip} * \text{SawLog}$$

$$\text{SLFuel} = \text{FSLFuel} * \text{SawLog}$$

where FLumber, FSLChip, and FSLFuel are the fractions of saw logs being used for each in any given year. Historical changes in saw-log processing efficiency (Hodgson, 1931; Corder et al., 1972; Lane et al., 1973a, Hartman et al., 1976; Willits and Fahey, 1988; Briggs, 1993) and waste disposition (Hodgson, 1931; Gedney, 1956; Cowlin and Forster, 1965; Manock et al., 1970; Corder et al., 1972; Schuldt and Howard, 1974; Bergvall et al., 1975; Hartman et al., 1976; Howard and Hiserote, 1978; Howard, 1984; Howard and Ward, 1988, 1991; Larsen, 1990, 1992) were compiled from historical summaries of the forest-products sector. In years without reported data, we used linear interpolation for estimating values.

2.5. PLYWOOD PRODUCTION

The primary products produced from veneer logs are plywood, hogged fuel, chips for paper production, and dimensional wood from cores left after veneer peeling. We combined plywood and lumber from peeler cores as one product. Wood waste resulting from plywood production was disposed as either chips or hogged fuels. As with sawlogs, veneer wood waste that was decomposed or incinerated was treated as hogged fuel.

The production of plywood (Plywood), chips (VLChip), and hogged fuel (VLFuel) from veneer logs (VenLog) was calculated as:

$$\text{Plywood} = \text{FPlywood} * \text{VenLog}$$

$$\text{VLChip} = \text{FVLChip} * \text{VenLog}$$

$$\text{VLFuel} = \text{FVLFuel} * \text{VenLog}$$

where FPlywood, FVLChip, and FVLFuel are the fractions of veneer logs being used for each in any given year. Historical changes in the efficiency of veneer-log processing (Corder et al., 1972; Lane et al., 1973b, Woodfin, 1973, Hartman et al., 1976; Adams et al., 1986; Brigs, 1993) and waste disposition (Gedney, 1956; Cowlin and Forster, 1965; Manock et al., 1970; Corder et al., 1972; Schuldt and Howard, 1974; Bergvall et al., 1975; Howard and Hiserote, 1978; Howard, 1984; Howard and Ward, 1988, 1991; Larsen, 1990, 1992) were compiled from summaries for the forest-products sector. In years without reported data, we used linear interpolation for estimating values.

2.6. PAPER PRODUCTION

During the processing of chips and pulp logs into paper, material is lost. The overall efficiency of paper production depends strongly on the process used. Although the efficiency of each pulping process has remained relatively constant with time, the proportion of paper produced by each process has changed markedly. To take into consideration these historical changes, we calculated a weighted average efficiency

Table I

Efficiency of the eight types of softwood pulping processes used to predict average efficiency

Pulping process	Efficiency (%)
Mechanical pulping	95
Soda pulping	87
Defibrating	85
Semichemical	70
Screenings/off quality	50
Sulfate-bleached	45
Sulfate-semibleached	50
Sulfate-unbleached	55
Sulfite-bleached	43
Sulfite-unbleached	48
Dissolving and special alpha	35

for Oregon and Washington and the United States by multiplying the efficiency (Smook, 1982) of each of the main categories of wood pulping processes (Table I) by the respective quantities of pulp produced each year.

The treatment of waste from wood pulp production also varies with the process. Waste from sulfite and sulfate pulping, the major processes in Oregon and Washington, is burned as fuel and to recover sulfur. Other waste is digested in settling ponds or disposed in landfills. In the model, we assumed that material not resulting in paper formation was burned, or that it decomposed rapidly in waste-water treatment.

The amount of paper produced each year (Papln) was calculated as:

$$\text{Papln} = \text{PapCR} * \text{EffPP} * (\text{PulpLog} + \text{SLChip} + \text{VLChip})$$

where EffPP is the efficiency of converting chips to paper for each year as determined above, and PapCR is the reduction in carbon content brought about by the paper-manufacturing process. For all forest products except paper, the carbon content was assumed to be equal to that of raw wood (52%). But since cellulose is the primary product of paper manufacturing, and the carbon content of pure cellulose is 23% lower than that of whole wood, carbon stored in paper products was adjusted to an average content of 40%.

2.7. WASTE DISPOSAL

Since 1900, paper and finished wood products have been disposed of in primarily four different ways: open dumps, sanitary landfills, incineration, and recycling. Flows to landfills, incinerators, and open dumps were determined from published reports. Records of the fraction of waste disposed of in open dumps were poor, so we noted when sanitary landfills began and when open dumps were closed to account for the transition from one type of disposal to the other (Collins, 1972; Baum and Parker, 1973; Waste Age, 1979; DeGreare, 1982; EPA, 1984; Liptak, 1991). For example, in the United States, the use of sanitary landfills was not accepted as the proper means to dispose of solid waste until after 1945 (Ham, 1972). We therefore assumed that prior to this time, solid waste was largely disposed in open dumps. The conversion rate of open dumps to sanitary landfills appears to have been low until the Resource Conservation and Recovery Act of 1976 (DeGreare, 1982; EPA, Office of Solid Waste, 1984). We therefore assumed that open dump conversion greatly increased after that point and was largely completed by 1980 (Collins, 1972; Liptak, 1991).

There are also few quantitative estimates of the amount of waste disposed by incineration. Data on the number of cities with incinerators indicate that their use increased between 1900 and 1940, but then declined as many municipalities converted to landfilling (Baum and Parker, 1973; Rathje, 1989). The first estimate of the fraction of waste incinerated is for 1960 when 30.8% of all municipal solid waste was disposed in this manner (EPA, 1990). We therefore assumed that at its peak in the 1940's, incineration would have accounted for a slightly higher fraction of waste. The decline in incineration appears to have continued from 1960 to the mid-1980's, with 20.6% incinerated in 1970, and 14.2% incinerated in 1988 (EPA, 1990). We assumed a linear rate of change over this period. Since 1988, an increase in the fraction of waste incinerated has been driven primarily by the need for waste volume reduction and energy recovery (Kiser, 1991; Schmidt, 1990).

Prior to 1960, the degree of paper recycling is difficult to document. We therefore assumed that 5% of all paper waste was recycled between 1900 and 1940, and that between 1940 and 1960 there was a linear increase from 5% to 18%, the latter value being the first reported value we could find (Liptak, 1991). After 1960 we used the time series reported by Franklin Associates (1988, 1993) and Rathje (1989) to estimate paper recycling rates.

3. Estimating Carbon Stores with FORPROD

Carbon stores in forest-product pools were tracked in short-term structures, long-term structures, paper supplies, mulch, and waste in open dumps and sanitary landfills (Figure 2). 'Mulch' refers to bark or sawdust that is composted or spread directly on the soil; 'open dumps' are disposal sites in which rates of biological decomposition and combustion are high; 'sanitary landfills' are sites with no combustion and low rates of biological decomposition. Changes in the pools are estimated with difference equations having a time step of 1 year. Input to short-term and long-term structures, mulch, and paper supplies are from the manufacturing subroutines previously described. Lumber and plywood production is divided into material added to short-term structures or long-term structures. The former include

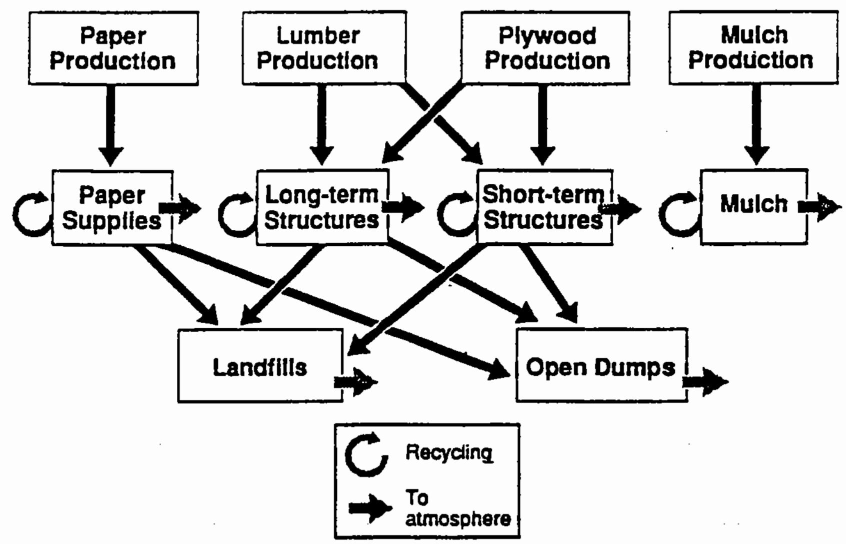

Figure 2. FORPROD flow of forest products during use and disposal. Recycling returns some material to the pool source and some to the atmosphere. All pools lose carbon to the atmosphere from decomposition or combustion.

wood in fences in decay-prone environments or in products such as pallets that have a short life span (< 20 years). The latter include buildings and other forms of wood with life-spans exceeding 20 years. All paper supplies, including paperboard, are tracked.

Waste (W) lost in disposal and decomposition of products is influenced by the rate of recycling and recovery into new products. We assumed that products recovered from a given source would be used in a similar way, that is, that paper would be recovered as paper, short-term structures as short-term structures, and long-term structures as long-term structures.

The following sections give the assumptions and equations for each FORPROD pool. Table II summarizes the values of the parameters used in the standard simulation.

3.1. MULCH

The change in mulch stores (Mulch) are calculated as:

$$\Delta\text{Mulch} = \text{MulIn} - \text{MulDK} * \text{Mulch}$$

Where MulIn is the annual input of mulch and compost as predicted from manufacturing functions and MulDK is the decomposition-rate constant. We assumed a decomposition-rate constant of 0.10 year^{-1}, a value slightly higher than that for bark in a natural setting (Harmon and Sexton, 1995).

Table II

Values for decomposition, replacement, and recycling parameters used in the standard simulation (See text equations in Section 3 for details of forest-product pools)

Process pool	Parameter	Value
Decomposition-rate constants		
Mulch	MulDK	0.10 $year^{-1}$
Short-term structures	SSDK	0.05 $year^{-1}$
Long-term structures	LSDK	0.01 $year^{-1}$
Open dump	DumpDk	0.30 $year^{-1}$
Landfill	LFillDk	0.005 $year^{-1}$
Replacement-rate constants		
Short-term structure	SWasteMax	0.10 $year^{-1}$
Long-term structures	LWasteMax	0.01 $year^{-1}$
Paper	PWasteMax	0.60 $year^{-1}$
Recycling recovery rate		
Short-term wood structure	WRcvr	90%
Long-term wood structure	Wrcvr	90%
Paper	PRcvr	90%

3.2. SHORT-TERM STRUCTURES

The change in short-term structures (SStr) is a function of input from lumber and plywood and loss from decomposition in use and replacement:

$$\Delta \text{SStr} = \text{SSIn} - \text{SSDK} * \text{SStr} - \text{SWaste} * \text{SStr}$$

where SSIn is the input from lumber and plywood, SSDK is the in-place decomposition-rate constant of short-term structures, and SWaste is the rate constant of replacement. SSIn is estimated from the production of lumber and plywood.

Because there is little direct data for the fraction of lumber and plywood used in short-term structures, we estimated that all wood used for shipping and half of the wood used in manufacturing would be used in short-term structures, which would mean 18% and 5%, respectively, were used in short-term structures between 1962 and 1986 (Haynes, 1990), such that

$$\text{SSIn} = 0.18 * \text{Lumber} + 0.05 * \text{Plywood}.$$

SWaste is a function of the rate of recycling (WRcycl) and rate of recovery into new forest products (WRcvr) so that

$$\text{SWaste} = \text{SWasteMax} * (1 - \text{WRcycl} * \text{WRcvr})$$

where SWasteMax is the maximum rate of replacement of short-term structures. This equation reduces the flow of waste to zero only when all material is recycled and completely recovered to new products. In this analysis we assumed that 95% of all short-term structures would be replaced within 30 years (SWasteMax = 0.10 year^{-1}), and that the decomposition-rate constant would be 0.05 year^{-1} (95% decomposing in 60 years). We also assumed that 90% of the recycled material would be recovered as 'new' short-term structures and that the remaining 10% would be disposed in open dumps, landfills, or incinerators.

3.3. LONG-TERM STRUCTURES

The change in long-term structures (LStr) is a function of input from lumber and plywood and loss from decomposition in use and replacement:

$$\Delta \mathrm{LStr} = \mathrm{LSIn} - \mathrm{LSDK} * \mathrm{LStr} - \mathrm{LWaste} * \mathrm{LStr}$$

where LSIn is the input from lumber and plywood, LSDK is the in-place decomposition-rate constant of long-term structures, and LWaste is the rate constant of replacement. LSIn is estimated from the production of lumber and plywood, so that

$$\mathrm{LSIn} = 0.82 * \mathrm{Lumber} + 0.95 * \mathrm{Plywood}$$

where coefficients are the compliment of those used to predict the fraction going to short-term structures.

LWaste is a function of the rate of lumber and plywood recycling and rate of recovery into new forest products so that

$$\mathrm{LWaste} = \mathrm{LWasteMax} * (1 - \mathrm{WRcycl} * \mathrm{WRcvr})$$

where LWasteMax is the maximum rate of replacement of long-term structures. As with short-term structures, this equation reduces the flow of waste to zero only when all material is recycled and completely recovered to new products. In this analysis we assumed that 95% of all long-term structures would be replaced within 300 years (LWasteMax = 0.01 year^{-1}) (Marin, 1978), and that the in-place decomposition-rate constant would be 0.01 year^{-1} (95% decomposing in 300 years). We also assumed that 90% of the recycled material would be recovered as 'new' long-term structures and that the remaining 10% would be disposed in open dumps, landfills, or incinerators.

3.4. PAPER STORES

The changes in stores of paper supplies (Paper) are a function of input from paper production (PapIn) and loss from disposal (PWaste):

$$\Delta \mathrm{Paper} = \mathrm{PapIn} - \mathrm{PWaste} * \mathrm{Paper}$$

PWaste is a function of the rate of paper recycling (PRcycl) and rate of recovery into new paper products (PRcvr), so that

$$\text{PWaste} = \text{PWasteMax} * (1 - \text{PRcycl} * \text{PRcvr})$$

where PWasteMax is the maximum rate of paper disposal. This equation reduces the flow of paper waste to zero only when all paper is recycled and completely recovered into 'new' paper. In this analysis we assumed that 95% of all paper supplies would be replaced within 5 years (PWasteMax = 0.60 year^{-1}), that 90% of recycled paper would be recovered as 'new' paper, and that 10% of the recycled paper would be disposed in open dumps, landfills, or incinerators.

3.5. OPEN DUMPS

Before the advent of sanitary landfills, paper and wood products in open dumps underwent rapid decomposition or combustion. The model accounts for the transition from open dumps to sanitary landfills. Changes in open dump stores (Dump) are a function of input from short- and long-term structures and paper supplies minus the removal from decomposition and combustion:

$$\Delta\text{Dump} = \text{LDump} * \text{LStr} + \text{SDump} * \text{SStr} + \text{PDump} * \text{Paper} - \text{DumpDk} * \text{Dump}$$

where LDump, SDump, and PDump are, respectively, the flows of waste from long- and short-term structures and paper to dumps, and DumpDk is the combined decomposition and combustion-rate constant for material in open dumps. The flow of waste into dumps depends on the amount of waste incineration and the flow of waste to landfills. For example, LDump, the rate at which long-term structural waste is added to dumps, is calculated as

$$\text{LDump} = \text{FWDump} * \text{LWaste} * (1 - \text{WoodIncin})$$

where FWDump is the fraction of wood waste going to dumps, LWaste is the rate at which long-term structures are replaced (as calculated under loss from long-term structures), and WoodIncin is the fraction of wood being incinerated. The other flows to dumps are calculated in a similar manner.

In this analysis we assumed that 95% of the material added to open dumps would decompose or be burned within 10 years, therefore we used a DumpDk rate-constant of 0.30 year^{-1}.

3.6. LANDFILLS

In modern sanitary landfills, solid waste is strongly compacted, covered, or capped with a layer of soil in a dry, anaerobic, and acidic environment. Little or no decay takes place (Rathje, 1989; Liptak, 1991), thus little carbon reenters the atmosphere.

Changes in landfill stores (LFill) are calculated as the difference between input from paper, short- and long-term structures, and the material decomposed:

$$\Delta \text{LFill} = \text{LLFill} * \text{LStr} + \text{SLFill} * \text{SStr} + \text{PLFill} * \text{Paper} - \text{LFillDk} * \text{LFill}$$

LLFill, SLFill, and PLFill are, respectively, the flows of waste from long- and short-term structures and paper to landfills, and LFillDk is the decomposition-rate constant for material in landfills. Although much of the carbon leaving landfills is in the form of methane (CH_4), no differentiation is made in the model. We did not partition flows from landfills into CO_2 and CH_4 for several reasons. First, there are few data on the rate of CO_2 versus CH_4 production during the course of decomposition. Second, even if decomposers produced only CH_4 in landfills, an undetermined and potentially large fraction may be converted to CO_2 by energy recovery or other combustion processes. As these uncertainties have no influence on carbon stores, we have defered this aspect of the problem until better data are gathered.

The flow of waste into landfills is calculated in a similar manner to the flow into open dumps and depends on the amount of waste incineration and the alternative flow of waste to open dumps. For example, PLFill, the rate paper is added to landfills, is calculated as:

$$\text{PLFill} = (1 - \text{FPDump}) * \text{PWaste} * (1 - \text{PaperIncin})$$

where FPDump is the fraction of paper disposed in open dumps, PWaste is the fraction of paper replaced as calculated under paper stores, and PaperIncin is the fraction of paper waste incinerated. The other flows to landfills are calculated in a similar manner.

4. Sensitivity Analyses

The simulation using the best estimate of parameters is called the 'standard run'. Some of the parameters (see Table II) used in this run were constant over the entire period, whereas others varied over time. The variations in the latter set of parameters represented the best or most likely historical reconstruction of trends over the simulation period. Additional simulation runs were made to test the sensitivity of the model to parameters of major concern. The details of each run are described in the following sections, named after the parameter that was tested. In all of these tests, standard-run values were used except for the parameter in question.

4.1. TRANSITION TO LANDFILLS

To assess the sensitivity of simulations to the flow of waste to open dumps versus landfills, we considered three scenarios: our best reconstruction of the time of transition from open-dump to landfill disposal, a transition 5 years earlier than estimated, and a transition 10 years earlier than estimated. Unless specifically noted, the standard run was the 'best reconstruction' scenario.

4.2. LANDFILL DECOMPOSITION

The effect of three rates of decomposition of waste in landfills was explored because there are no quantitative measurements of this process and we were unable to determine an upper limit on the expected lifetime of landfill material. We therefore examined a low-decomposition scenario in which 95% decomposition occurred within 1200 years, a medium-decomposition scenario in which it occurred in 600 years, and a high-decomposition scenario in which it occurred in 300 years. The high, medium and low rate-constants were 0.01, 0.005, and 0.0025 year^{-1}, respectively. The medium landfill decomposition rate-constant was used for the standard run.

4.3. RECYCLING RATES

Although recycling rates for paper have been compiled annually since 1960 (Franklin Associates, 1993) the actual rates are debatable because some 'recycled' paper may be used for fuel or products subject to high rates of decomposition (e.g., hydromulch). We explored the effects of recycling by doubling and halving the reported rates.

4.4. LONG-TERM STRUCTURE REPLACEMENT

There are few estimates of replacement rates of long-term structures. Longevities of 100–150 years are often used (Harmon et al., 1990), but the only rigorous survey we found indicated a longevity of 300 years (Marcin, 1978). We used three rate-constants of replacement to examine the effects of this parameter: 0.01 year^{-1} (the standard run), one half of that value (0.005 year^{-1}), and double that value (0.02 year^{-1}).

4.5. TEMPORAL VARIATION

In the standard run, the values of some parameters, such as manufacturing efficiencies and waste disposal, varied over the simulation period. In many past studies, the values were held constant over the simulation period. In this set of simulations, we explored the effect of holding the parameters constant. Two fixed sets of values derived from the standard run were used: parameters specific to 1970, and parameters specific to 1990.

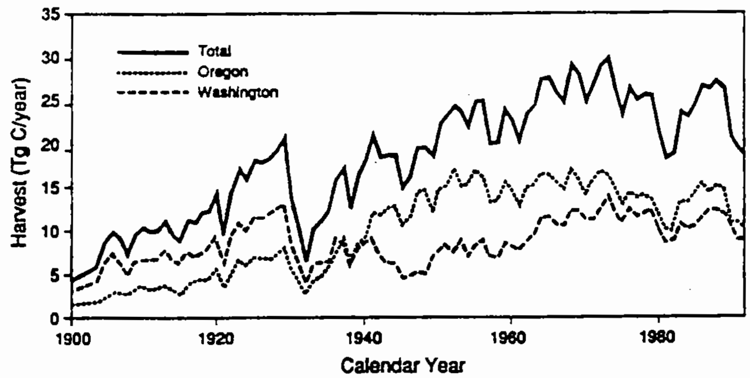

Figure 3. Historical reconstruction of the total wood and bark carbon harvested in Oregon and Washington between 1900 and 1992.

5. Harvest and Manufacturing Patterns, 1900–1992

The total amount of tree harvest from Oregon and Washington forests from 1900 to 1992 for use in wood products excluding firewood is estimated to be 1,692 Tg (Tg $= 10^{12}$ g). The amount of carbon removed increased from an estimated 4 Tg year^{-1} in 1900 to a high of 29.9 Tg year^{-1} in 1973 (Figure 3). Since 1945, the harvest of carbon for the wood products industry from these two states has averaged 23.9 Tg year^{-1}. Fluctuations in harvest have been primarily due to economic cycles in the United States, the largest fluctuation occurring during the Great Depression in the 1930's.

5.1. RAW LOG USE

The primary use of harvested logs has been as saw logs for lumber production (Figure 4). The use of veneer logs for plywood production was relatively minor until the 1950's, when building construction increased. Pulp logs have been a minor component of the timber harvest in Oregon and Washington throughout the entire period, peaking in 1962 at approximately 16% of all logs. Since 1960, pulp logs have comprised an average of 9.8% of all logs harvested in Oregon and Washington.

5.2. BARK USE

Bark has been used primarily for fuel (Figure 5). In the mid-1960's a growing market for bark mulch arose, and since 1965 it has averaged 14.5%. Most of the remainder has been burned as hogged fuel or waste.

5.3. SAW LOG USE

The largest change in saw log production over the last 90 years has been in the use of mill waste and not in milling efficiency, as one might assume (Figure 6A).

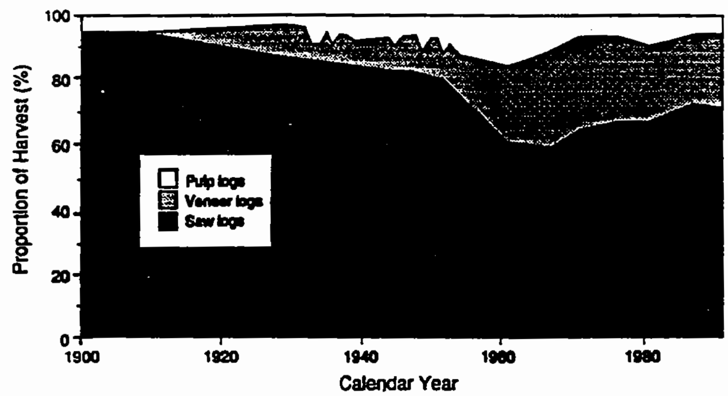

Figure 4. Historical reconstruction of the utilization of raw logs for lumber, plywood, and paper production in Oregon and Washington between 1900 and 1992.

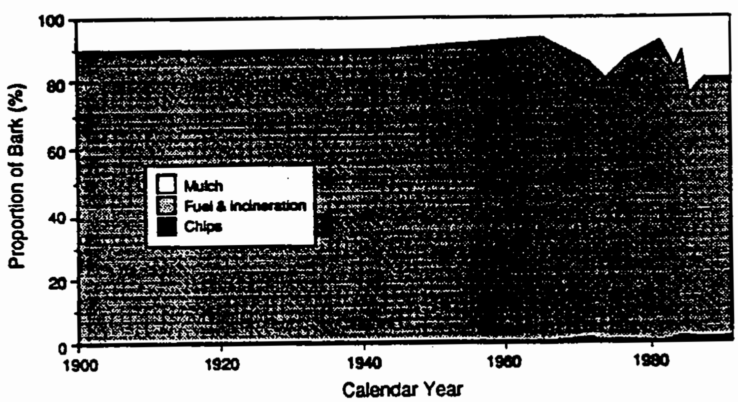

Figure 5. Historical reconstruction of the disposition of bark waste in Oregon and Washington between 1900 and 1992.

In the first half of this century, the percentage of lumber produced from saw logs in Oregon and Washington was approximately 52%. After 1945, production fluctuated, but generally declined to 40% by 1950. A decline in efficiency between 1945 and 1950 was due to additional processing, such as planing that reduced the amount of lumber by 10% (Corder et al., 1972), redefining of board-foot lumber measurement to smaller dimensions, and use of logs of smaller diameter. After the 1950's, saw mill efficiency has gradually increased because of technological improvements (Adams et al., 1986).

Saw log residue, averaging 48% of the wood, was primarily burned as waste or as hogged fuel until the late 1940's. In the mid to late 1940's, the use of the residue for pulp increased with the introduction of the sulfate pulping process. A tightening log supply and technological improvements in log barkers and chippers made it possible to use the saw log residue from Douglas-fir [*Pseudotsuga menziesii*

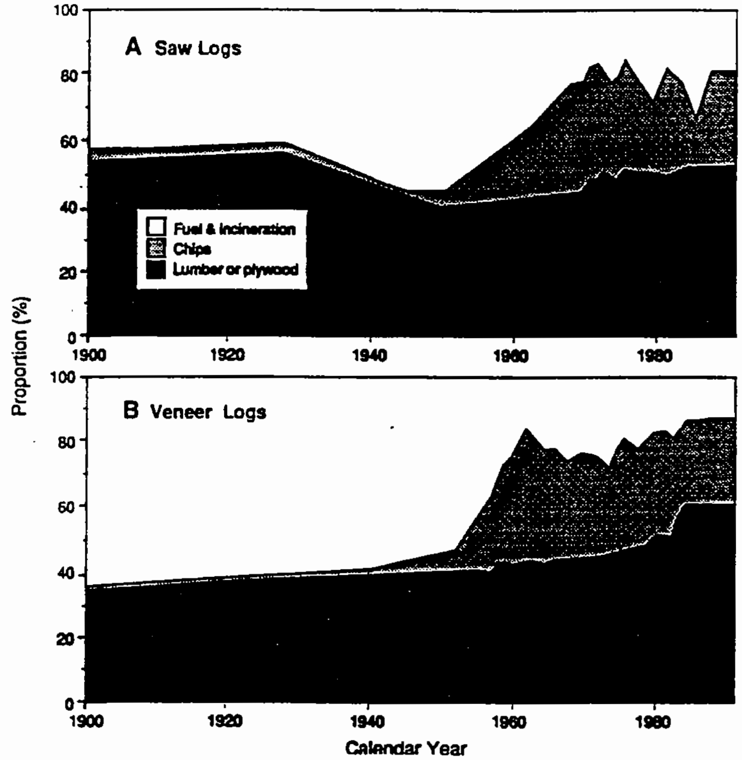

Figure 6. Historical reconstruction of the disposition of (A) saw logs and (B) veneer logs in Oregon and Washington, between 1900 and 1992.

(Mirb.) Franco] for pulping (Gedney, 1956). The use of log waste residue for chips increased until 1960, and since that time has averaged 27%.

5.4. VENEER LOG USE

As with saw logs, the largest change in veneer log use involved the disposition of the waste (Figure 6B). The percentage of plywood produced from veneer logs in Oregon and Washington slowly increased with efficiency from 40% in 1940 to 50% in 1980. Since the mid-1980's, technological improvements have increased the efficiency of plywood mills to approximately 61%.

Veneer log residue was primarily burned as waste or used as hogged fuel until the late 1940's, when there was an increase in the chipping of wood waste for paper production for the same reasons as for chipping of saw log residues. The fraction of veneer logs used for chips increased until 1960, and since that time has averaged 31%.

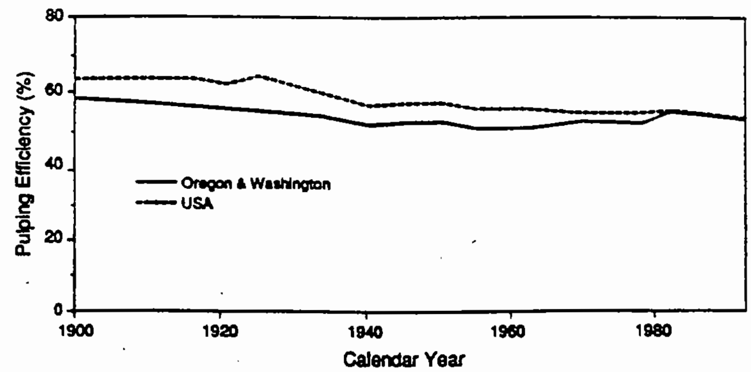

Figure 7. Estimated changes in papermaking efficiency between 1900 and 1992 in (A) Oregon and Washington and (B) the entire United States.

5.5. PAPER PRODUCTION

The overall efficiency of producing pulp for paper products declined slightly from approximately 60% to 52% between 1900 and 1992 (Figure 7). This is a trend not only for Oregon and Washington, but for the United States as a whole. The efficiency decline is due to an increase in paper production from sulfate pulping processes and a decrease in the proportion of mechanical pulping.

5.6. COMBINED PRODUCTION

The combined mass of forest products manufactured from 1900 to 1992 is primarily associated with the change in the mass harvested (Figure 8), product mass ranging from 2.04 Tg year^{-1} in 1900 to a high of 17.01 Tg year^{-1} in 1973. Changes in manufacturing efficiency and use of milling waste have also been important, and in some periods have counteracted the influence of harvest levels on production. During 1930 to 1950, for example, harvest levels increased 4-fold, but overall manufacturing efficiency (defined as the ratio of product output to harvest) declined from approximately 50% to 40%. Since 1950, the overall manufacturing efficiency has increased steadily (approximately 61% in 1992) because of changes in individual manufacturing efficiencies and use of wood waste for paper production, and the increase has partially offset the generally lower harvests during 1975 to 1992.

As might be expected from the disposition of raw logs, lumber has been the primary forest product from the two states over the period examined, although the proportion of lumber in total products has declined from 89% in 1900 to 53% in 1992. The decrease in fraction of total output has been caused, in part, by the increase in plywood production, which has remained at approximately 20% of total output since 1960. Construction materials have therefore been the major output over the period, ranging from 73% to 89% of total production. Perhaps the largest cause of the decreased importance of lumber has been the increase in paper production

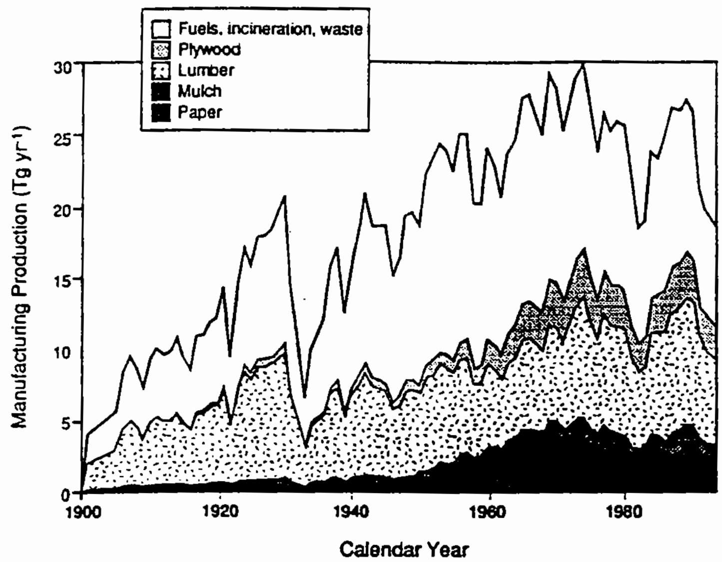

Figure 8. Historical reconstruction of manufacturing production for logs harvested in Oregon and Washington between 1900 and 1992.

since 1950. Before then, paper comprised < 10% of the total product output. The increased use of wood waste for chips after 1950, however, greatly increased paper production to a peak 29% of total production in the 1960's. Since then, paper has been approximately 20% of all product output.

5.7. WASTE DISPOSAL

In the first 70 years of this century, open dumps and incineration were used for most of the wood and paper products disposed (Figure 9). The conversion of open dumps to sanitary landfills appears to have been gradual between 1940 and 1970 and then rapid into the 1980's.

After a long period of decline between 1940 and 1985, during which the fraction of waste incinerated apparently dropped from 35% to 5%, the fraction started to increase, 17% being incinerated in 1991 (Kiser, 1991) and 25% incineration predicted for 1992 (Schmidt, 1990).

Recycling of paper waste in the United States has increased gradually since 1960, when records began to be kept. The percentage of paper and paperboard recycled in the U.S. has steadily risen: 18.1% in 1960 (Liptak, 1991), 20.6% in 1970, 26.7% in 1980, and 38.1% in 1992 (Franklin Associates, 1993). The recycling of wood waste appears to have been minimal until the late 1980's. For example, Portland, OR, has shifted from recycling none of its wood waste in 1985 to 18% in 1992.

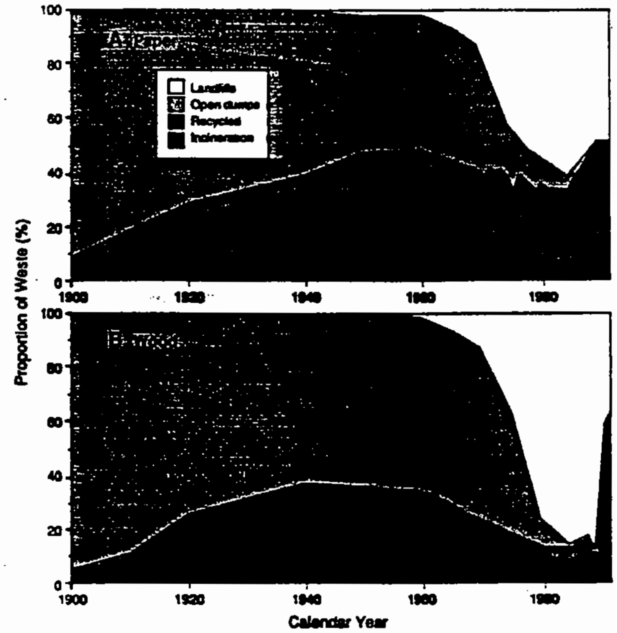

Figure 9. Historical reconstruction of the fate in Oregon and Washington between 1900 and 1992 of (A) paper waste and (B) wood waste.

6. FORPROD Estimates of Carbon Stores, 1900–1992

6.1. RESULTS WITH STANDARD SIMULATION

Of the 1,692 Tg of carbon that has been harvested between 1900 and 1992, the standard simulation indicated that 396 Tg or 23% remains in storage. The largest storage pool has been long-term structures, which, by the end of the period examined, comprised 74% of the total stores (Figure 10). Although landfills rank second, that pool comprised a smaller fraction (20%) than we originally anticipated, probably because landfills have been a major disposal site only for the last two decades. All other pools together contained 6% of the total stores, and some pools, such as paper and mulch, contained less than 1%.

The analysis indicates that, despite nearly a century of timber harvest, few forest product pools have reached a steady state. The overall rate of increase of forest-product carbon stores from 1900 to 1992 was 4.3 Tg year^{-1}. From 1972 to 1992, the rate was 6.02 Tg year^{-1}, indicating that, if anything, the rate of forest-product accumulation is increasing, largely because of the growth of the landfill pool, which had average net accumulations of 0.33 Tg year^{-1} between 1952 and 1972 and 3.46 Tg year^{-1} between 1972 and 1992. In contrast, the net accumulation rate in long-term structures has increased only slightly over those two periods, from 3.2 Tg year^{-1} to 3.65 Tg year^{-1}.

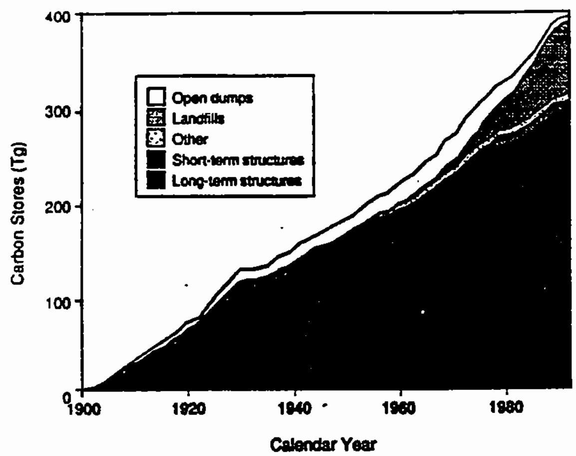

Figure 10. Accumulation of carbon in forest products produced in Oregon and Washington between 1900 and 1992, as estimated by FORPROD.

6.2. EFFECT OF LANDFILL DISPOSAL

Forest-product stores were affected more by the transition from open dumps to landfills than by landfills decomposition-rate constants. Shifting the time of transition forward 5 years and 10 years from the standard scenario gave predictions of 405 and 414 Tg, respectively (Table III), an increase of 2.3% and 4.5%, respectively, over the standard simulation store of 396 Tg. Relative to the cumulative harvest, the discrepancy is even smaller ($< 1\%$), indicating it had little effect on the overall result.

6.3. EFFECT OF LANDFILL DECOMPOSITION

The sensitivity analysis indicated that the landfill decomposition-rate constant, one of the most difficult parameters to estimate, did not greatly influence the overall result (Table III). The rate constants of 0.0025, 0.005, and 0.01 year^{-1} yielded total forest-product stores of 398, 396, and 393 Tg, respectively, as of 1992, a change of $\pm 1\%$ of total stores and $< 0.1\%$ of the cumulative harvest.

6.4. EFFECT OF RECYCLING

Increasing and decreasing the recycling rates had an unanticipated result (Table III). Although doubling the rate increased paper stores from 3.97 Tg to 4.56 Tg, it lowered total stores from 396 to 389 Tg. Halving the recycling rate had the opposite effect, increasing total stores to 400 Tg. Modifying the rate of paper recovery did not modify this trend. This counterintuitive result stems from the fact that paper in landfills lasts much longer than paper as product, so that there is a slight carbon

Table III

Effect of varying selected parameters on estimates of carbon stores in forest products

Test	Total stores	Percent change from standard run
Standard run	396	–
Landfill decomposition		
0.0025 $year^{-1}$	398	+0.5
0.010 $year^{-1}$	393	−0.8
Recycling		
Halved	400	+1.0
Doubled	389	−1.8
Landfill transition		
5 years earlier	405	+2.2
10 years earlier	414	+4.5
Long-term structure replacement		
0.005 $year^{-1}$	422	+6.6
0.02 $year^{-1}$	357	−9.9

gain without recycling. Relative to the total store, the increase is minor (< 1%) and might be offset by the effects of a reduced demand for virgin fiber.

6.5. EFFECT OF LONG-TERM STRUCTURE REPLACEMENT

The rate constant of long-term structure replacement had the greatest effect on FORPROD simulations (Table III). Decreasing the rate constant from the standard simulation value of 0.01 $year^{-1}$ to 0.005 $year^{-1}$ caused the 1992 total forest-product stores to increase from 396 Tg to 422 Tg, a change of 6.6%. Although this is a large increase in forest-product stores, it only represented a 1.5% increase relative to the cumulative harvest. Likewise, an increase in the replacement rate constant to 0.02 $year^{-1}$ resulted in a decrease in overall stores to 357 Tg, and a 10% decrease of forest-product stores.

6.6. EFFECT OF TEMPORAL VARIATION

The effect of holding efficiency rates constant varied with the data period (Figure 11). When 1970 values were used, overall stores were close to those with the standard simulation, a total of 364 Tg in 92 years. This is an 8% underestimate, probably due to the lower use of landfills in the 70's. Much larger discrepancies were introduced with the 1990 values, which gave a total store of 594 Tg in 92 years, 50% larger than values with the standard simulation.

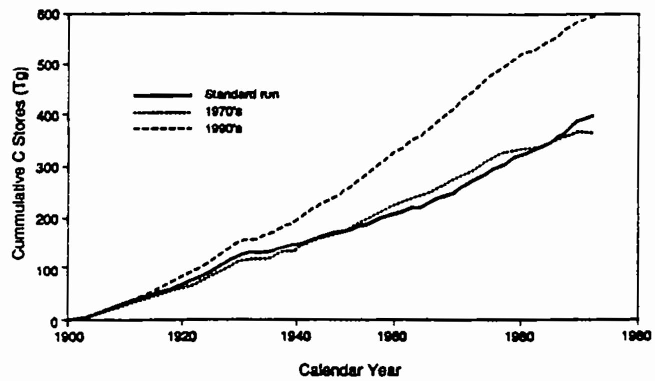

Figure 11. Total accumulation of forest products estimated by FORPROD when parameters varying with time in the standard run (see Figures 4–8) are held constant.

The pools that differed most among these three simulations were long-term structures and landfills. The 1990 rates gave the largest difference among stores in long-term structures: 273, 295, and 345 Tg with 1970, standard, and 1990 values, respectively. The difference for landfill stores was even greater: 43, 78, and 232 Tg with 1970, standard, and 1990 data, respectively. The latter results indicate that while knowing the exact time of transition from open dumps to landfills is not important, modeling with landfills as the sole disposal site is untenable.

In an earlier study, Harmon et al. (1990) estimated that 45% of harvested carbon ends up in long-term storage pools with an average loss of 1.5% year^{-1}. Applying those values here indicates a total carbon store in 1992 of 401 Tg. While total stores over the 92-year period estimated with the two methods are comparable, the earlier study overestimated carbon stores in some years after 1938 by as much as 54 Tg. Although the overall trend is the same, results for a given year may be significantly inaccurate.

7. Discussion and Conclusions

The overall carbon balance of a region depends on net changes in carbon pools such as living vegetation, detritus, soils, and forest products. As our analysis for pools other than forest products is incomplete, it would be misleading to calculate an overall balance. Nonetheless, our analysis of the forest products pools is important for reconstruction of regional carbon flux. It indicates that, despite the large mass of carbon (1,692 Tg) harvested in Oregon and Washington, only a small fraction (23%) is currently stored in forest products. This fraction is probably higher than average for the United States because paper production is more important in other regions (e.g., in the Southeast). The fraction of net stores is probably also high relative to that found in developing regions where manufacturing efficiency is low and decay in use may be greater.

The estimated rate of current accumulation of forest products manufactured from logs harvested in Oregon and Washington was approximately 6 Tg year^{-1}. Our analysis indicates that, far from being in balance, forest products resulting from harvest in the Pacific Northwest will continue to accumulate if harvest levels remain constant. For comparison with other studies of forest products, this absolute accumulation rate can be placed in relative terms with respect to harvest mass and current product stores. Conversion to relative terms indicates a net accumulation of approximately 25% of harvest mass and a growth rate of the forest-products pool of approximately 2% year^{-1}. These values are considerably lower than estimates for the Canadian forest sector (Kurz et al., 1992) of a net accumulation of 50% of harvest mass and a growth rate of the forest-products pool of 4% year^{-1}. The difference may be attributable to the predominance of paper as a product and of landfills as a store in the Canadian forest sector. A study of future forest-product stores from timber harvest in Finland (Karjalainen et al., 1995), in which current harvest levels were extrapolated 50 years into the future, indicated that approximately 38% of the harvest would be in net storage and that forest-products pool would have a relative growth rate of 1.5% year^{-1} from years 30 to 50.

Given the early stage of forest-product modeling, it is difficult to determine whether these differences are due to the dynamics of the systems or to variation in the modeling approaches. The former would be more interesting and meaningful; however, differing assumptions about waste deposition (i.e., transition from open dumps to landfills), landfill decomposition rates, and recycling may obscure real differences in system dynamics. We can distinguish some differences by comparing the manufacturing efficiency estimated by the models. Karjalainen et al. (1995) estimate an overall efficiency of 68%, and Kurz et al. (1992) give individual efficiencies for products that indicate an overall efficiency of 38%, if lumber and paper production (approximately 16.6 Tg) is divided by harvest mass (44 Tg). Our study, which includes past as well as current periods, estimates a quite comparable range of 40% to 61%. It is interesting that the study showing the lowest manufacturing efficiency (Kurz et al., 1992) had the greatest rates of accumulation and net storage. These differences in system response are therefore likely to be caused by treatment of forest products in use rather than in manufacturing.

While it is possible to use average rates of manufacturing efficiency and waste disposition over a given period of interest, our analysis indicates that this method introduces major inaccuracies in temporal patterns of accumulation, particularly when the transition from open-dump to landfill waste-disposal is not included. This may partially explain the large proportion (50%) of forest products found to be stored in landfills by Kurz et al. (1992), who assumed landfills since 1946 were the primary waste deposition site. If one assumes that landfills are not important until 1970 (as in the United States), then they would store about half the value estimated by Kurz et al. (1992) and comprise 25% of the total stores. Our sensitivity analysis of wastes generated from Oregon and Washington wood products indicate a similar effect. The assumption that landfills were the primary deposition site increased the share of forest products stored landfills from 20% in the standard simulation to 39%.

In contrast to its sensitivity to the transition from open dumps to landfills, the FORPROD model was relatively insensitive to the rates of recycling, landfill decomposition, and long-term structure replacement. Doubling and halving those parameters led to less than 10% change in total stores of forest products. Karjalainen et al. (1995) performed a sensitivity analysis similar to ours by altering product life-spans, recycling rates, and landfill decomposition rates. They found that changing the product life-span 10% resulted in $< 3\%$ change in total stores. Similarly they found that increasing recycling 50% increased stores $< 2\%$, and that doubling the flow to landfills from 25% to 50% of all waste increased total stores 10%. The largest change resulted from increasing the landfill decomposition rate from 1% to 10% year^{-1}, which decreased stores 20%.

The insensitivity of forest-products models to most parameters may be due to the fact that substantial amounts of carbon are lost to the atmosphere during manufacturing, when approximately 40% to 60% is lost within a few years of harvest, leaving a relatively small fraction to be stored for a long period. These models may also be insensitive to these parameters because they generally involve internal transfers to pools that sequester carbon. The sensitivity of the models to the manufacturing parameters is fortuitous, because those parameters have the best historical data. In contrast, the fate of paper and wood wastes appears to be a key focus for future research. Once the uncertainty regarding paper and wood waste is resolved, the role of forest products in the overall global carbon balance can be assessed.

Acknowledgements

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