Role of Forest Products in the Global Carbon Cycle: From the Forest to Final Disposal

12

Christopher Larson, Jeffrey Chatellier, Reid Lifset, and Thomas Graedel

Executive Summary

This chapter reviews the role of the production, use and end-of-life management of harvested wood products (HWPs) in the global carbon cycle. Harvested wood products can be long term reservoirs of carbon; however, solid wood products, paper, and paperboard manufacturing require large energy and heat inputs, and end-of-life pathways can further or hinder carbon sequestration, depending on management.

What We Know About Harvested Wood Products and the Carbon Cycle

- The global stock of carbon within forest products is estimated between 4,100 teragrams (Tg) carbon and 20,000 Tg carbon, with net sink rates estimated between 26 and 139 Tg carbon per year. The methods and assumptions used to estimate the role of HWPs in the global carbon cycle vary, resulting in a wide range of findings. Even assuming the high end of the estimates, these findings suggest that forest products are still are a minor component of the global carbon budget.
- Manufacturing processes operate on a mix of fossil energy and biomass energy, a by-product derived from wood waste. Emission reductions are achieved when energy generated from biomass displaces fossil fuel emissions.
- Newer wood products such as oriented strand board, laminated veneer lumber and I-joists use 80–216% of the energy needed to produce solid sawn lumber. It is unclear whether the lower density of newer wood product materials, given their increased strength and greater utilization of wood resources, offsets the energy intensity per unit of the newer materials.
- Paper products contain significantly more embodied fossil fuel (carbon) energy than solid wood products: 0.3–0.6 megagram carbon (MgC) in fossil energy used/MgC for virgin paper products vs. 0.07 MgC in fossil energy used /MgC for solid wood products in Finland in 1995. However, approximately 50% of U.S. paper production is manufactured using recycled paper as a feedstock. Recycled feedstock may reduce or increase GHG emissions relative to virgin pulping depending on the pulping process and energy sources.
- Global transport of wood and paper products is estimated to account for approximately 27% of total fossil carbon emitted within the manufacturing and distribution process.
- Several researchers assert that substitution of wood for other construction materials (e.g., steel and concrete) produces net GHG emissions

C. Larson
New Island Capital, 505 Sansone St, San Francisco, CA 94111

J. Chatellier
Forest Carbon Ltd, Bali, Indonesia

R. Lifset (✉) • T. Graedel
Yale School of Forestry & Environmental Studies, 360 Prospect St, New Haven, CT, USA
e-mail: reid.lifset@yale.edu

M.S. Ashton et al. (eds.), *Managing Forest Carbon in a Changing Climate*,
DOI 10.1007/978-94-007-2232-3_12,

reductions. These substitution effects may be up to 11 times larger than the total amount of carbon sequestered in harvested wood products annually. Quantification of substitution effects relies on many assumptions about particular counterfactual scenarios, most importantly linkages between increased/decreased forest products consumption and total extent of forestland.
- The end-of-life pathways of HWPs can augment GHG emissions reductions. Once discarded, HWPs can be burned for energy production, recycled or reused, or put in landfills, where the carbon can remain indefinitely due to anaerobic conditions. However, HWPs discarded in landfills create methane, a greenhouse gas that is 24 times more potent than CO_2, thus potentially offsetting gains from carbon storage.
- Inclusion of end-of-life pathways in HWP carbon stock calculation models is crucial, as failure to do leads to estimates with a high degree of error.

1 Introduction

Although many policy makers recognize the role forests play in carbon sequestration and climate change mitigation, to date there is no accepted methodology for quantifying and incorporating harvested wood products into global carbon budgets and carbon markets. While there is ample discussion surrounding sustainable forest management and the long-term sequestration of carbon in standing forests, the discussion rarely considers the life-cycle of wood and or the linkages between forest management and end markets for wood products.

Understanding the role of harvested wood products (HWPs) in the global carbon cycle is essential if appropriate policy concerning the treatment of HWPs as a carbon stock is to be implemented on a national or even international level under multi-lateral agreements in a post-Kyoto protocol regime (Rueter 2008). Studies that quantify current global stocks of HWPs vary greatly, as calculation methods are dependent on critical assumptions regarding product life, decay rates, and system boundaries (Pingoud et al. 2003; Green et al. 2006). A lack of data on the usage and disposal of HWPs adds to the difficulty of quantifying this global carbon stock (Kuchli 2008). Opinion on system boundaries is divided across the literature. The topic of landfills is a major part of this debate as models that include "end-of-life" within their system boundaries are intrinsically tied to assumptions made regarding the level of methane (CH_4) capture from landfills. The composition of materials in landfills has a significant impact on the magnitude of CH_4 generation, while the landfill design greatly influences the ability to capture landfill gases or convert methane passively through oxidation.

This chapter reviews the role of the forest products industry and harvested wood products within the context of global carbon stocks and flows, starting at the beginning of the HWP life cycle with production, turning then to the estimation of product life span and carbon stocks in products, and concluding with issues related to end-of-life management.

The chapter reviews the direct and indirect effects on greenhouse gas (GHG) emissions of production within the wood products and paper and paperboard sectors. It demonstrates the complexities of including wood products in use in such analyses by examining recent research in life-cycle assessment, manufacturing and use trend data, and literature on the impacts of materials substitution. The literature on the carbon stock of HWPs is reviewed and currently accepted research on the topic of product life spans and HWPs in landfills is summarized. The end-of-life pathways of HWPs and their carbon implications are then examined.

The chapter concludes by considering areas of further research, such as incorporation of the role of forest management in carbon sequestration via HWPs and further study of carbon benefits currently claimed by proponents of wood product substitution for more energy-intensive raw materials.

1.1 Overview and Framework for Assessing Role of HWPs in Carbon Cycle

From the perspective of global carbon stocks and flows, forest products are a heterogeneous

Table 12.1 A framework for evaluating the carbon profile of the forest products industry

Carbon Flux	Activity or Source
Direct emissions	Manufacturing
Indirect emissions	Transport
	Purchased power
	Landfill CH_4 emissions
Sequestration	Forests
	Products in use
	Products in landfills
Avoided emissions	Combined heat and power applications
	Product recycling
	Substitution effects

Adapted from Miner (2008)

Table 12.2 Emissions and sequestration estimates for the global forest products value chain

Value chain component	Est. Net emissions, Tg CO_2-eq. $year^{-1}$	Uncertainty[a]
Direct emissions: manufacturing	262	±20%
Indirect emissions: purchased power	193	±25%
Indirect emissions: transport	70	±50%
Indirect emissions: landfill-derived methane	250	−50% to +100%
Net forest sequestration	−60	±200%
Sequestration in forest products	−540	±50%
Avoided emissions: biomass fuels	−175	±200%
Avoided emissions: combined heat and power	−95	±200%
Avoided emissions: recycling	−150	±200%
Product substitution effects	Unknown	N/A

Modified after NCASI (2007)
[a]Uncertainty is based on professional judgment as presented in NCASI (2007)

group that consists of very short-lived products (e.g., newsprint) to very long-lived products (e.g., furniture, housing stock). Heterogeneity increases further due to different manufacturing processes, energy requirements for production, sources of energy within manufacturing, consumption patterns, end-of-life considerations, and substitution effects.

Miner (2008) presents a useful framework for evaluating the carbon profile of the forest products industry (Table 12.1). In general, direct and indirect emissions are quantified as positive GHG contributions, while sequestration and avoided emissions are quantified as negative GHG emissions—where negative emissions, i.e. reductions, are desired—relative to a business-as-usual scenario (Miner 2008). The net GHG profile is difficult to quantify because data for several of these processes are imprecise or unavailable. This is particularly true for sequestration and emissions within solid waste disposal sites as well as for substitution effects.

In light of the broad divestment of industrial timberland in the United States (Brown 1999) largely to timberland investment management owners, it is reasonable to ask whether forest carbon sequestration should be part of the carbon profile of the forest products industry. As many timberland buyers increasingly seek to manage, quantify, and monetize carbon sequestration benefits (Lippke and Perez-Garcia 2008) alongside traditional timberland management strategies, it is becoming increasingly important to apply a life-cycle framework to the industry, including the sequestration potential of forestlands as a source of raw material inputs. Similarly, it is also important to quantify products-in-use and in landfill sequestration since these are also key components of the life-cycle of forest carbon in use.

The National Council for Air and Stream Improvement, a U.S.-based industry-sponsored research group (NCASI 2007), presents estimates of net emissions of the forest products and forest carbon sequestration (Table 12.2). It is important to note that these figures do not take into account any product substitution effects, but do incorporate a large figure for forest carbon sequestration that may not be linked to the production of forest products. The largest areas of uncertainty in these estimates relate to transportation-related emissions, forest carbon sequestration, and methane emissions from landfilled forest products.

Forest products and the forest products industry are unique within the realm of carbon stocks and flows. First, industrial production of forest products typically uses a high proportion of its own

feedstock (of biomass derived from production byproducts) as its energy source. Nevertheless, the vast majority of direct fossil CO_2 emissions are still generated in the production phase. Second, once in use, most forest products do not generate CO_2 emissions; upon disposal, they can generate varying degrees of CO_2 and methane (CH_4) depending on decomposition rates.

Forest products are often considered to be less energy- and emissions-intensive substitutes for other building materials, particularly concrete, steel, and aluminum (Wilson 2005; Upton et al. 2008). These substitution effects may play a much greater role in global CO_2 reduction schemes than improvements within the forest products manufacturing process itself (Kauppi and Sedjo 2001; Miner 2008). However, while some researchers (e.g., Burschel et al. 1993) regard product substitution as important, they point out that changes in forest management are even more significant. Denman et al. (2007) note that terrestrial ecosystems, and forests in particular, sequester amounts equal to approximately 25% of total anthropogenic emissions. Thus, the impacts of industry on forestland extent, stocking rates, and land-use conversion must be included in a comprehensive analysis of the carbon footprint of the industry.

Long-lived wood products-in-use constitute a carbon sink (Skog 2008), as do some wood products within solid waste disposal sites (Skog and Nicholson 1998; Micales and Skog 1997). NCASI (2007) estimates that within the United States the total gross emissions through the forest products value chain in 2005 were 212 Tg[1] carbon dioxide equivalents (CO_2e)[2] per year, while the forest carbon pool in products (in-use and landfills) grew by 108.5 Tg CO_2e per year. In 2005 in the U.S., landfilled wood products constituted 3% of total carbon stocks within the forest sector, but accounted for 27% of carbon sequestration (defined as flux in total carbon stocks), which is estimated to average 162 Tg carbon per year (Woodbury et al. 2007). The global stock of carbon within forest products is estimated between 4,100 Tg carbon (Han et al. 2007) and 20,000 Tg carbon (Sampson et al. 1993; IPCC 1996), with net sink rates estimated between 26 Tg carbon per year (IPCC 1996) to 139 Tg carbon per year (Winjum et al. 1998). The Intergovernmental Panel on Climate Change (IPCC 2007) estimates the total global standing forest carbon stock to be 3,590,000 Tg carbon for vegetation only and 11,460,000 Tg carbon for forest biomass and soils . Others estimate that the total carbon stock of the terrestrial biosphere is 24, 770,000 Tg C, including non-forest stocks (Fischlin et al. 2007). Even assuming the high estimate of 20,000 Tg carbon for forest products, this suggests that they still are a minor component of the global carbon budget.

However, Woodbury et al. (2007) assert that the forest products sector (including forest growth) provided net carbon sequestration equal to 10% of total U.S. CO_2 emissions in 2005. While forests accounted for 63% of net sequestration, changes in products-in-use and landfilled forest products accounted for 37% of net sequestration, implying that in 2005 the production and disposal of forest products was responsible for sequestering 3.7% of total U.S. CO_2 emissions. NCASI (2007) estimates that in 2005, 52% of gross emissions from the forest products industry was offset by carbon sequestration in products-in-use and products in landfill. It further estimates that annual forest growth on all private lands offset an additional 61% of gross emissions from the forest products industry. USEPA (2008) reports that total U.S. CO_2 emissions in 2005 were 7,130 Tg CO_2e, which, using figures from NCASI (2007), suggests that forest harvesting, clearance and disturbances (e.g. fire) represent only 1.8% of total U.S. emissions, and that forest products represent only 1.5% of total U.S. emissions.

A potentially broader set of carbon implications arises from the use of wood for energy or as a substitute for more carbon-intensive materials. These substitution effects have been explored extensively through comparisons of steel/aluminum/

[1]Teragrams – 1 Tg is equal to 1,000,000 Metric tons. All tons in this chapter are metric unless otherwise indicated.

[2]Carbon dioxide equivalent (CO_2e) is a measure for describing the climate-forcing strength of a quantity of greenhouse gases using the functionally equivalent amount of carbon dioxide (CO_2) as the reference.

Table 12.3 Production and production concentration of industrial roundwood, paper and paperboard, and wood fuel

	Industrial roundwood, 1,000 m^3 $year^{-1}$	Industrial roundwood, % of global production	Paper and paperboard, 1,000 ton $year^{-1}$	Paper and paperboard, % of global production	Woodfuel, 1,000 m^3 $year^{-1}$	Woodfuel, % of global production
Production, Top 10 Countries	1,175,185	71%	263,350	74%	1,061,620	60%
Production, Top 25 Countries	1,445,594	88%	331,510	94%	1,385,578	78%
Production, Total Global	1,645,681	100%	354,490	100%	1,771,978	100%
Top 10 Counties in Production	USA, Canada, Russia, Brazil, China, Sweden, Finland, Germany, Indonesia, France		USA, China, Japan, Canada, Germany, Finland, Sweden, South Korea, France, Italy		India, China, Brazil, Ethiopia, Indonesia, Dem. Rep. of Congo, Nigeria, Russia, USA, Mexico	

Derived from FAO (2004)

concrete vs. wood housing designs (Marceau and VanGeem 2002, 2008; Wilson 2005; Perez-Garcia et al. 2005; NCASI 2007) and use of biomass fuels (Sedjo 2008). There is less literature, however, on the effects of wood demand on maintaining tracts of forestland (Ince 1995). Together, these indirect effects may play a much larger role in GHG reduction than direct effects within the forest products sector.

1.2 Definition of Harvested Wood Products (HWPs) and Related Terms

Harvested wood products (HWPs) can be defined as wood-based materials that, following harvest, are transformed into commodities such as furniture, plywood, paper and paper-like products (Green et al. 2006). The term HWP is further simplified by the International Panel on Climate Change (IPCC) defining it as all wood material (including bark) that is transported off harvest sites. It does not include woody biomass, commonly referred to as slash or residual material, left at harvest sites (Pingoud et al. 2003).

Sawtimber refers to trees or logs large enough to be sawn into lumber. Once harvested, tree boles (i.e., the main stem of the tree) intended for human utilization are termed *roundwood*. The UN Food and Agriculture Organization (FAO) more formally defines roundwood as wood in its natural state after it has been harvested, including logs that have undergone minimal transformation and may be without bark, rounded, split, or roughly squared. Roundwood is used as either *woodfuel* or industrial roundwood, which is used to produce HWPs. Woodfuel, destined for heating, cooking and energy production, includes solids (fuelwood and charcoal), liquids (black liquor,[3] methanol, and pyrolitic oil) and gases from the gasification of these substances. *Fuelwood* is wood in the rough such as branches, twigs, logs, chips, sawdust and pellets, used for energy generation. Woodfuel is not analyzed here. While industrial roundwood and paper/paperboard production are concentrated in a few industrialized countries, fuelwood is less concentrated, and more prominent in lesser-developed countries (Table 12.3).

HWPs are categorized into two groups: *solid wood products* (SWPs) and *paper products*. Solid wood products consist of sawn wood and wood-based panels, typically measured in cubic meters. Paper products are defined as paper and paperboard (thicker, stronger and more rigid grades of paper) which are measured in dry tons (Green et al. 2006). In many cases, HWPs are further transformed into different product classes and categories throughout their lifecycle due to recycling (Pingoud et al. 2003).

[3]Black liquor is liquid residual from soda or sulfate pulping.

Table 12.4 Roundwood, pulpwood, woodfuel production and production ranking for selected countries and the global HWP industry

All values		Selected countries								
1,000 m^3 $year^{-1}$	Global total	US	Finland	Canada	Russia	Japan	Brazil	China	Zambia	Mexico
Industrial roundwood	1,645,681	414,702	49,281	196,667	134,000	15,615	110,470	95,061	834	6,913
% as Pulpwood	*32%*	*41%*	*51%*	*14%*	*40%*	*23%*	*43%*	*7%*	*0%*	*14%*
Woodfuel	1,771,978	43,608	4,519	2,901	48,000	114	136,637	191,044	7,219	38,269
Total roundwood	**3,417,660**	**458,310**	**53,800**	**199,568**	**182,000**	**15,729**	**247,107**	**286,105**	**8,053**	**45,182**
% of Roundwood as fuelwood	*52%*	*10%*	*8%*	*1%*	*26%*	*1%*	*55%*	*67%*	*90%*	*85%*
Global rank of industrial roundwood production		1	7	2	3	18	4	5	78	33

Modified from data in FAO (2004)

2 The Global Forest Products Industry

For the purposes of this review, we make a distinction between the forestland harvests for land clearing/forest conversion versus the continuous production of forest products such as sawtimber, paper/pulp, biomass, and other forest products. Land clearing (deforestation) is considered a primary driver of anthropogenic CO_2 emissions, accounting for between 17% and 20% of total global CO_2 contributions between 1990 and 2002 (WRI 2006; Watson et al. 2000). Drivers of deforestation include a variety of sources ranging from fuelwood consumption, illegal logging, and expansion of agricultural land (Stern 2006).

The total global annual volume of harvested wood products in 2006 was 3.42 billion cubic meters (m^3) according to the U.N. Food and Agriculture Organization (FAO 2007). About 1.65 billion m^3 was industrial roundwood, while 1.77 billion m^3 was fuelwood. Others, however, suggest that harvest was slightly lower (approximately 3 billion m^3 per year), with approximately 1.8 billion m^3 as industrial roundwood, and 1.2 billion m^3 as fuelwood (Nabuurs et al. 2007). These figures may differ on the total fuelwood harvest, since economic data typically do not include fuelwood. In 2006, developed countries accounted for 70% of total global industrial roundwood consumption (USA, Canada, European Union, Japan) (FAO 2007). However, the largest producers, in order, were the USA, Canada, Russia, Brazil, and China.

The percentage of roundwood used for woodfuel varies greatly by country. Developed countries typically report low percentages used for fuelwood, while lesser-developed countries generally report a higher proportion of roundwood as fuelwood (Table 12.4).

3 Direct Effects of Forest Product Harvest, Manufacturing, and Distribution

3.1 Wood Harvesting

Wood products result from harvesting trees from natural forests and plantations. The harvesting process generates significant amounts of by-product, such as branches, leaves and other unmerchantable biomass, which are often either burned or left in the forest to decompose. The proportion of merchantable to unmerchantable biomass varies by forest type, species, and age at harvest. Representative values cited in the literature for North American forests suggest that 20–40% of tree biomass remains in the forest after harvest (Côté et al. 2002; Finkral and Evans 2008).

The variability reflects the diversity of commercially harvested species and forest types, as well as economic factors and harvest technologies.

Sustainable management and the use of a formal management plan should be requirements for any forest to be included as a carbon sink under national and international GHG accords. As of 2007, approximately 90% of developed country forests were harvested under sustained yield objectives within a management plan, while only 6% of developing country forests were similarly managed (Nabuurs et al. 2007).

3.2 Wood Products Manufacturing

Solid wood products manufacturing uses a majority of all global industrial roundwood volume. Its direct manufacturing emissions are a small fraction of total industry emissions, unlike paper and pulp manufacturing, which create much higher direct emissions. Globally, in 2004, 68% of all industrial roundwood volume went to the manufacturing of a variety of solid wood products (FAO 2007). Using 2004 FAO data, NCASI (2007) estimates that the global solid wood products industry emits 25 million tons (Mt) of fossil CO_2 per year. Major categories include solid sawn lumber (softwoods and hardwoods), structural panels (plywood, oriented strand board [OSB]), non-structural panels (e.g., particleboard), engineered wood products (laminated veneer lumber, I-joists, glulam), and miscellaneous uses (telephone poles, railroad tracks). Many of these products, with the exception of packaging/pallets, are manufactured for durable purposes. Since durable goods usually have product lives of 40–80 years, solid wood products have the potential to sequester carbon for significant periods. Furthermore, much of the solid wood stream is then deposited in a solid waste disposal site, where it may be sequestered near-permanently (Skog and Nicholson 1998).

McKeever (2002) estimates that in 1998 the United States, which leads the world in consumption of solid wood products, consumed 0.23 billion m^3 of solid wood products in the following proportions: solid sawn lumber (62%), structural panels (18%), nonstructural panels (12%), engineered wood products (1%) and miscellaneous (8%).[4] Researchers focused on the carbon sequestration potential of the solid wood products sector have offered several conclusions related to product mix and manufacturing:

1. Since 1970, the rate of resource utilization (the percentage of roundwood that ends up in final product form) of the U.S. solid wood products industry has increased significantly, despite a recognized reduction in size and quality of roundwood inputs. Yields from raw materials have increased, and inputs of petroleum-based additives in engineered and panel products have decreased.
2. Since 1970, the product mix within the solid wood products industry has shifted from solid sawn lumber and plywood to a mixture of engineered wood products and OSB. This is likely due to changes in quality of roundwood inputs and demand for uniform, high-performance engineered products (Meil et al. 2007).
3. The industry produces a substantial amount of its energy needs through biomass electricity and heat production, which are often adjacent to manufacturing facilities.

3.3 Carbon Management Implications of Trends in Solid Wood Product Manufacturing

Within the wood products sector, there is a strong trend toward engineered products such as glue-laminated lumber, I-joists, and non-plywood structural panels such as oriented strand board. Proponents recommend these products for their load-bearing strength and uniformity relative to solid sawn wood (Meil et al. 2007). They can also be manufactured from small-diameter roundwood and/or scraps from other processes. Because these products have been allowed under the two major international building codes (IBC/IRC), are favored by builders for their uniformity and strength, and allow for greater economic utilization of harvested fiber, it is unsurprising that this

[4] Data include domestic and imported products.

Table 12.5 Carbon dioxide (CO_2) emissions in the cradle-to-gate life cycle of a wood building product from the generation of the forest through product manufacturing, 2004

	Pacific northwest production				Southeast production				
Product	Glulam	Lumber	LVL	Plywood	Glulam	Lumber	LVL	Plywood	OSB
CO_2 emissions (biomass), kg/m^3	230	160	141	146	231	248	196	229	378
CO_2 emissions (fossil), kg/m^3	126	92	87	56	199	62	170	128	294
CO_2 emissions, total, kg/m^3	356	252	228	202	430	310	366	357	672
CO_2 emissions (biomass), kg/m^3	65%	63%	62%	72%	54%	80%	54%	64%	56%
CO_2 emissions (fossil), kg/m^3	35%	37%	38%	28%	46%	20%	46%	36%	44%
Total energy, MJ/m^3	5,367	3,705	4,684	3,638	6,244	3,492	6,156	5,649	11,145
Product yield, log to product		53%		51%		41%		50%	71%
Product yield, other wood inputs to product	82%		N/A		82%		N/A		
Description of other wood inputs	Dry, planed lumber		Veneer		Dry, planed lumber		Veneer		

Derived from Puettmann and Wilson (2005)
Note: I- joists are made of OSB and LVL, and could not be included in this table.
Glulam glue laminated timber beams, *LVL* laminated veneer lumber, *OSB* oriented strand board

is the fastest growing sub-segment of the solid wood products industry (Meil et al. 2007). By volume, these products made up about 1% of total U.S. roundwood consumption as of 1998. Sales growth of engineered wood products increased 30.2% over 2000–2004 (McKeever 2002). In contrast, the American Plywood Association projects that solid sawn lumber consumption will drop below 4 billion cubic feet (ft^3) in 2012, implying no growth in volume between 1998 and 2012.

Because solid wood manufacturing encompasses a mix of solid sawn wood and engineered wood products, it is worthwhile to examine the carbon footprint of each major segment. A series of studies by Wilson (2005) and Kline (2005) conducted as part of the CORRIM research program[5] provides carbon and energy consumption data for the production of various solid wood products (Table 12.5). CO_2 emissions by product type range from 202 to 672 kg CO_2/m^3, with U.S. Southern OSB production resulting in the highest emissions by volume[6] (Puettmann and Wilson 2005). Variability within a product type arises from differing regional energy sources and year of analysis. Solid sawn lumber production is not substantially lower in CO_2 emission/volume than the engineered wood products. However, Pacific Northwest plywood generated 24% lower CO_2 emissions than solid sawn wood.

Most engineered wood products contain (by mass) 5–15% in additives such as petroleum-based adhesives, waxes, and resins. These are created using more energy-intensive manufacturing processes. Because these products are stronger, less wood fiber is required within the construction process relative to solid-sawn lumber. For example, I-joists use approximately 62–65% of the wood fiber of a solid joist, but

[5]CORRIM is a research consortium focused on the environmental impact of the production, use, and disposal of wood and other bio-based materials. The Consortium includes US and Canadian research institution members and a number of contributing companies, associations and agencies related to the forest products industry.

[6]Meil et al. (2007) assert that this is largely due to the use of regenerative thermal oxidizer (RTO) units which are a critical element of air emissions control in OSB manufacturing.

their production is more energy-intensive. As a result, substitution of I-joists for solid-sawn lumber provides negligible opportunities for CO_2 emissions reduction (Perez-Garcia et al. 2005). Moreover, substitution of OSB for plywood reduces total carbon emissions only by 3–4%.

Resource utilization studies conducted in the U.S. in 1976 and again several decades later (Wernick et al. 1997; Meil et al. 2007) document increased utilization of by-products while providing interesting data on product yields from raw materials. In 1970, the softwood lumber industry had a 35% utilization rate by weight (e.g., conversion of raw logs into the primary product). This rose to 45% in 2000. Efficiency gains for softwood plywood showed a 7% improvement. However, a much greater proportion of plywood byproducts were used as raw materials for other products, such as nonstructural panels, rather than being burned or landfilled. In addition, over the same period, adhesive and resin content in plywood was reduced by 17% (Meil et al. 2007). The authors therefore calculate a reduction of 62.7 kg of fossil-derived CO_2/m^3 of softwood lumber produced in 2000 relative to 1970.

Biomass has been an important, carbon-neutral energy source for the forest products industry. Biomass is considered by Watson et al. (2000) and others to be a "carbon-neutral energy source" because it does not generate fossil carbon emissions. Within the forestry sector, forest regeneration is thought to offset carbon from energy production from biomass. Other researchers argue that such accounting masks important complexities (e.g., Luo et al. 2009; Walker et al. 2010). Regardless, within the IPCC framework, changes in forest regeneration are reported separately as land-use change (Watson et al. 2000; IPCC 2007), which makes it difficult for forest products industry bookkeeping to include the life cycle of their products for the purposes of calculating carbon stocks.

Over the past 30 years, the industry has improved utilization efficiency for materials by creating value from products once burned for energy, and by burning for energy products that were previously typically burned solely for disposal purposes (Wilson 2005). Historically, the industry burned bark and other "wet" residues in uncontrolled outdoor burners variously termed "teepee" or "beehive" burners, with significant particulate emissions and zero energy recovery. Only sawdust and planer shavings were converted into energy due to the cost and conversion efficiency of boilers. Today, in developed nations, it is more common for all residue, including bark, mill-ends, sawdust and shavings to be burned for the cogeneration of heat and electric power. These outputs are used to drive manufacturing processes within modern solid wood product mills. But they sometimes remain uncounted in carbon budgeting.

Two studies from the 1970s indicate that historically energy recovery was low. Grantham and Howard (1980) indicate that in 1970 25% of residual byproducts were used as fuel, and another 37% transferred to other facilities as raw materials. Corder et al. (1972) claim that 26% (for lumber) and 24% (for plywood) of byproducts were used for fuel in 1967. Between 1970 and 2000, bark and wet residues began to be used as fuel for combined heat and power applications at manufacturing sites.

3.4 Pulp and Paper Manufacturing

In 2004, the pulp and paper industry consumed approximately 32% of all industrial roundwood produced globally (FAO 2007). NCASI (2007) estimates that pulp and paper manufacturing processes globally emit 195–205 Mt of fossil CO_2 per year (compared to 25 Mt CO_2 per year for solid wood products). The pulp and paper industry generally produces products that are shorter-lived than the solid wood products segment, ranging from various grades of newsprint and paper to paperboard. The production of paper products from virgin fiber is considerably more energy-intensive than all solid wood products, since wood fiber must be converted (chemically or mechanically) from a mixture of cellulose, hemicellulose, and lignins into a cellulose-dominated pulp for papermaking. In 1998, the paper manufacturing industry ranked as the United States' fourth largest emitter of greenhouse gases,

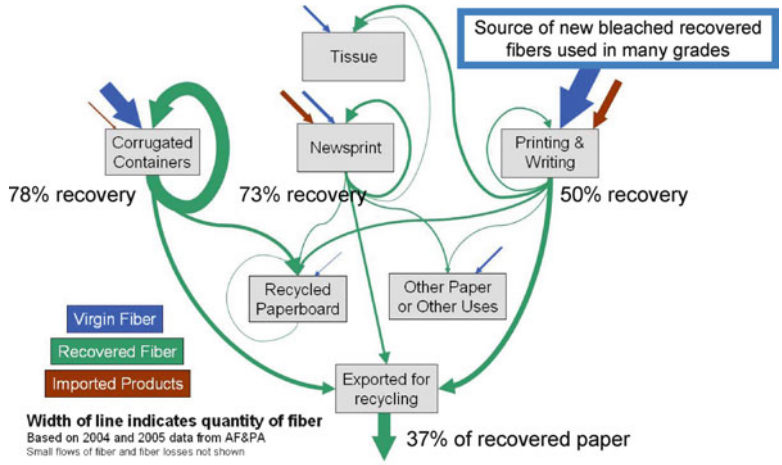

Fig. 12.1 The complex supply web of the forest products industry. Thickness of lines signify relative magnitude of flows. (*Source*: From Miner 2008. Reprinted with permission)

following petroleum, basic chemicals, and metals (EIA 2006). Using 1991 data, Subak and Craighill (1999) estimate that the paper and pulp industry directly and indirectly accounted for 1.3% of total global fossil carbon emissions in 1993.

Industry segments vary in production volumes, carbon intensity, manufacturing processes, and estimated service life. In 2006, the United States produced 41.8 million short tons of paper and 50.4 million short tons[7] of paperboard products. In contrast to flat or slowly growing markets for solid wood products (McKeever 2002), the U.S. paper and paperboard markets in total have been declining since 1999 (Irland 2008) likely the result of a transition away from newsprint consumption. Furthermore, production has dropped faster than consumption as significant industry segments have moved offshore (e.g., China now dominates global packaging markets) (FAO 2007).

International trade in pulp, paper, and paperboard products is considerably more developed than trade in raw sawtimber and solid wood products. A different set of nations is dominant within global production of paper and paperboard products (Table 12.3). Additionally, recycled fiber streams play a much greater role in paper manufacturing relative to solid wood products manufacturing (Falk and McKeever 2004).

Paper industry inputs vary by product type, and include (i) industrial roundwood, (ii) chips as a co-product of solid wood product manufacturing and (iii) recycled fiber. Certain products require more virgin fiber for tensile strength, while other products can be produced with predominantly recycled fiber. A NCASI (2007) report states: "Of the 352 million tons of paper and paperboard produced globally in 2005, 162 million tons, or 46%, was recovered rather than being disposed" (FAO 2006a). Miner (2008) also documented a complex fiber supply web within the industry (Fig. 12.1). Of the 100 million tons of paper consumed annually within the U.S, approximately 53.4 million was recovered for recycling in 2008. A 2008 press report from Forestweb (Irland 2008) indicates that paper manufactured from 100% recycled pulp results in 1,791 kg/ton of CO_2 emissions, vs. 4,245 kg/ton of CO_2 emissions from paper manufactured from virgin pulp. However, there is some debate over the role of recycled fiber in reducing GHG emissions within the industry. The de-inking and recycling process is energy-intensive, and typically involves 100% purchased power (vs. in-house biofuel-derived power in virgin pulp manufacturing). Some researchers suggest that the climate benefits of recycled material arise from the avoided CH_4

[7] One short ton = 2,000 pounds (lb) ≈ 907 kilograms (kg, SI) = .907 metric tons.

Table 12.6 Energy inputs and ratio of embodied** carbon in raw material vs. final product under a variety of pulping processes in Finland, 1995

	Total production, Gg year^{-1}	Direct fuels, MWh/Mg	Heat, MWh/Mg	Electricity, MWh/Mg	C in raw material/ C in final product
Mechanical					
GWP, B	801		0	1.55	1.2
GWP, NB	1,167		0	2.1	1.23
TMP, NB	923		−0.75	2.4	1.2
TMP, B	801		−1.17	3.37	1.24
CTMP	105		0.56	1.65	1.25
SCP	509		1.06	0.4	1.45
Chemical					
HSUP, B	2,174	0.39	3.07	0.69	2.46
SSUP, NB	680	0.52	2.77	0.57	2.56
SSUP, B	2,928	0.52	3.33	0.75	2.71
Recycled					
REC, NB	180	0	0	0.1	1.07
REC, B	272	0.25	0.17	0.4	1.17
Total	10,540				

From Pingoud and Lehtila (2002). Reprinted with permission

*In Finland, 51% of produced chemical pulp was dried in 1995 (Carlson and Heikkinen 1998). This is included in the energy demand figures

**Embodied energy, also known as embedded energy, refers to the energy consumed in the prior steps in the product chain.

Abbreviations used: *GWP* ground wood pulp, *TMP* thermo-mechanical pulp, *CTMP* chemi-thermo-mechanical pulp, *SCP* semi-chemical pulp, *HSUP* hardwood sulphate pulp, *SSUP* softwood sulphate pulp, *REC* recycled pulp, *B* bleached, *NB* unbleached, *MWh* megawatt hours. One megawatt-hour (MWh) ≈ 3.6×10^9 joules (J, SI) ≈ 3.412×10^6 British Thermal Units (BTU).

emissions from decomposing paper within landfills (Subak and Craighill 1999; NCASI 2007).

Using data from Finland's forest products industry, Pingoud and Lehtilä (2002) estimate that in 1995 across pulping processes and fiber sources, the proportion of fossil-based carbon emissions per wood-based carbon in end products (Mg carbon/Mg carbon) is 0.07 for sawn wood and 0.3–0.6 for paper in the manufacturing stage, suggesting that paper is 428–857% more fossil carbon intensive than sawn wood by mass. They also found that direct fuel, heat, and electricity demands for the production of 11 grades of pulp in Finland in 1995 can dramatically vary (Pingoud and Lehtilä 2002) (Table 12.6).

Chemical pulping uses either a kraft (sulfate) process or sulfite process to dissolve lignins, which are burned with other derivatives to recover pulping chemicals and to provide process heat (Côté et al. 2002). This process leaves cellulose fibers largely intact for high-quality papermaking. Mechanical pulping uses fiber more efficiently, yielding a lesser amount for biofuel as a process energy, and increasing the need for purchased electricity (Pingoud and Lehtilä 2002). Chemical processes result in 50–55% loss of fiber by weight, while recovery of recycled paper results in a 16–18% loss of fiber by weight. Fiber that does not end up in the final product is generally burned in the production process or landfilled (Côté et al. 2002). Industry-wide in the U.S., 56% of all energy needs are met with biofuel co-products (Davidsdottir and Ruth 2004). Farahani et al. (2004) have highlighted a new technology, black liquor gasification-combined cycle (BLGCC), which has the potential, under certain conditions, to fully offset energy usage within the chemical pulping process. In this case, using less recycled feedstock actually improves the GHG emissions profile by providing greater opportunities to use biomass and black liquor as energy feedstock.

In general, mechanical pulping is less energy-intensive, although, as noted, it also uses a greater proportion of purchased electricity in its

manufacture. Given the reputation for energy and process efficiency of the Nordic paper and pulp industry (Subak and Craighill 1999), the figures in Table 12.6 may not be globally representative, yet are among the few data points available on this topic.

Similar to trends within the solid wood products industry, the paper and pulp sector has experienced process, energy efficiency, and resource utilization improvements since 1970. IEA (2003) documents a 0.8% decrease in energy intensity of OECD-country paper and pulp making processes from 1968 to 1990. Nevertheless, as of 2002 in the U.S., the paper and pulp industry remains the second highest manufacturing sector on an energy intensity basis (with petroleum/coal as the highest) (Davidsdottir and Ruth 2004). This estimate does not take into account the relatively high proportion of energy derived from biomass fuels within the forest products industry, approximately 40% in the United States in 1998 (EIA 2008).

3.5 Carbon Implications of Transport and International Trade of Forest Products

Transportation of forest products, both as raw industrial roundwood and as consumer products, has been recognized as a significant potential source of fossil carbon emissions (Pingoud and Lehtilä 2002; NCASI 2007). Research indicates that some forest products can travel large distances prior to and following manufacture, via overland freight or cargo ship. Globally, NCASI (2007) estimates that product transport results in fossil carbon emissions of approximately 70 million tons CO_2 per year, or approximately 27% of total fossil carbon emitted within manufacturing and distribution processes. Pingoud and Lehtilä (2002) examined transportation related emissions in Finland, documenting a wide range of transportation modes and distances. Their research concluded that transportation from harvest site to mill, and from mill to consumer, accounted for 22% and 20% respectively of total fossil carbon emitted within manufacturing and distribution processes in 1995.

3.6 Indirect Effects of the Forest Products Industry on Carbon Emissions

As noted above, the forest products industry's contribution to total global GHG emissions is minor, despite its high energy intensity (NCASI 2007), partly due to its significant use of biomass fuels to power manufacturing processes, and its long-lived products, which sequester carbon in products-in-use and landfilled products. Beyond purchased power, transportation, and landfill methane emissions related to forest products, the forest products industry offers products that may be less fossil carbon-intensive than substitute materials such as concrete, aluminum, and steel. To the extent that increased use of forest products results in an expansion of timberlands operated on a sustained-yield basis, substitution effects may have a greater impact on net carbon sequestration beyond a comparison of the embodied energy within various substitutable building materials.

Gustavsson et al. (2006) describe four GHG emissions-related aspects to materials substitution: (i) emissions from fossil fuel use over the life cycle of the product (e.g., production, transportation, end use and waste management); (ii) replacement of fossil fuels with biomass energy within the production phase; (iii) carbon stock changes in forests, products-in-use and landfilled materials; and (iv) GHG emissions from industrial process reactions in such areas as cement and steel production. While it is impossible to accurately quantify all actual and counterfactual outcomes within this framework, Kauppi and Sedjo (2001) indicate that the range of possible substitution effects may be up to 11 times larger than the total amount of carbon sequestered in forest products annually. This suggests that minor changes in consumer preference for materials can have a big impact on the overall GHG emissions profile of the construction sector.

In many applications, forest products are substitutable with rival products, typically plastics, metals or concrete (Upton et al. 2008). Researchers have compared the carbon footprint of forest products relative to some of these materials, and

concluded that increased use of forest products within the construction sector would result in decreased GHG emissions (Wilson 2005; Upton et al. 2008). Currently, in the U.S., wood framing techniques are used in approximately 90% of new housing starts (Upton et al. 2008). This percentage is much lower in other regions of the world, particularly outside of North America and Northern Europe (Gustavsson et al. 2006).

In the lifetime of a house, there are two primary sources of carbon emissions: the construction of the structure, and the energy requirements to heat and cool the structure over its lifetime. It is difficult to compare wood vs. other building materials because alternative materials have different thermal characteristics. For example, the thermal mass associated with concrete buildings may reduce heating and cooling costs, thereby lowering carbon emissions during building operation, suggesting that the advantages of wood could be overestimated (Nishioka et al. 2000; Upton et al. 2008).

Upton et al. (2008) project that wood-framed single-family houses require 15–16% less total energy for nonheating and cooling purposes and emit 20–50% less fossil CO_2 to build than non-wood houses made of steel framing products. This conclusion relies on several key assumptions about the ratio of embodied energy in housing relative to energy expended to heat and cool the house over its lifetime, as well as assumptions regarding the fate of forests used or not used for the production of industrial roundwood (Upton et al. 2008). Wilson (2005) found that the wood-framed house had a global warming potential index (a measure of total GHG emissions, not energy usage, as in Upton et al. (2008)) 26% and 31% lower, respectively, than model steel and concrete house designs. These figures represent only the embodied energy within the production of the house, not its operation. These figures are supported by Gustavsson and Sathre (2006), who conducted a sensitivity analysis around uncertainties and variability within the production of both concrete and wood. Using plausible inputs, wood building materials had lower embodied energy costs relative to concrete in all cases analyzed.

Perez-Garcia et al. (2005) characterize the substitution effects throughout the value chain from forest to landfilled product. The product life span of wood used in housing and therefore the carbon that is sequestered is unchanged, regardless of the length of the forest rotation. What changes is the carbon stock in the forest. Furthermore, with substitution effects, the use of wood products offsets concrete or metal construction, providing a greater benefit than either the forest carbon pool or the forest product carbon pool. In short, intensive forest practices create a "positive carbon leakage" through greater use of wood products in the market place.

Several studies examining substitution posit that greater use of forest products will result in greater retention of working forestlands, or conversely, that less use of forest products will hasten conversion of working forestlands to other land uses (Wilson 2005; Perez-Garcia et al. 2005; Upton et al. 2008). Regardless of the validity of this assumption, it is important to recognize that each author implicitly or explicitly recognizes that carbon fluxes within forestlands are several orders of magnitude greater than any identified substitution effect. Thus, it is worth examining how and whether the forest products industry has any effect on the extent and condition of forestlands relative to other factors.

4 Estimate of Carbon in HWPS

Global estimations of yearly HWP production are derived from statistics collected by FAO on the production of roundwood. In the United States, the USDA Forest Service keeps statistics on roundwood harvests and HWP production based on data collected from government agencies and industry. The FAO reports that, globally, 1.65 billion m^3 of roundwood is extracted annually for HWP production (FAO 2007). In 2002, the United States produced approximately 425 million m^3, or 25%, of global roundwood intended for HWPs (Howard 2006). If this global roundwood production figure were converted to carbon, it would be very large. However, production losses occur as roundwood is processed into different products, and assumptions on the magnitude of these losses greatly influence the final calculations.

Table 12.7 Global productiovn of HWPs in 2000 according to FAOSTAT 2002

	Billion m^3/year	Pg C/year
Primary products		
Roundwood	3.1	0.71
Wood fuel	1.5	0.37
Industrial roundwood	1.6	0.34
Pulpwood (Round & Split)	0.48	0.11
Sawlogs + Veneer Logs	0.95	0.20
Other Indust Roundwd	0.15	0.03
Semi-finished products		
Sawnwood	0.42	0.09
Panels + Fibreboard	0.22	
	billion tons/year	
Paper + Paperboard	0.32	0.15

Source: Pingoud et al. (2003)

The associated carbon fluxes have been estimated by assuming that the approximate dry weight of coniferous wood is 0.4 tons/m^3 and non-coniferous is 0.5 tons/m^3 and that the carbon fraction in biomass is 0.5. In addition, the estimated charcoal production was 0.04 billion tons/year (metric tons per year). The production of wood residues was 0.06 billion m^3/year and chips and particles 0.16 billion m^3/year, these being mainly by-products of wood processing

First, it is assumed that roughly 50% of harvested roundwood logs is lost as residues (Gardner et al. 2004), which brings the total to 825 million m^3. Data from 2004 show that the paper products industry consumed 32% of the total roundwood production, which would account for 264 million m^3, while solid wood products accounted for 561 million m^3. However, these figures are further reduced when losses from final product finishing are taken into account. Skog and Nicholson (2000) assumed an 8% loss for solid wood products, and 5% for paper products, during finishing. This would mean that 516 million m^3 of solid wood products and 251 m^3 of paper products comprise the total annual global production of HWPs. Using the same assumptions on production losses, United States yearly production of HWPs would amount to 133 million m^3 of solid wood products (SWP) and 65 million m^3 of paper products (see Table 12.7).

In 1996, the world's forests produced 3.4 billion m^3 of harvested roundwood. About 1.9 billion m^3 (56%) of this harvest was fuelwood; the remainder (1.5 billion m^3) was industrial roundwood (e.g., sawlogs and pulpwood). The industrial roundwood corresponds to a harvesting flux of about 0.3 Gt C $year^{-1}$(FAO 2004).

Estimates of the total carbon sequestered in HWPs globally vary widely from 4,200 Tg C (IPCC 2000) to 25,000 Tg C (Matthews et al. 1996). In another study, Harmon et al. (1990) suggest that global C stocks in long-lived products lie in the range 2–8 Pg C.

Similarly, estimates of the net annual sink from HWPs ranges from 26 to 139 Tg C/year in these same reports (IPCC 2000; Matthews et al. 1996). This compares to the 38,000 Tg CO_2e in estimated worldwide emissions in 2004 (IPCC 2007), which equates to 139,300 Tg C, thus the total amount of carbon sequestered annually in HWPs is small. There are several reasons to explain the wide range in these figures on HWP annual sink. First, estimates will vary based on the assumptions made about average production losses and wood densities. The choice of wood density can have considerable impact on the results (Stern 2008). Secondly, as described below, HWP stock estimates frequently do not distinguish between HWPs in use versus those in landfills (Pingoud et al. 2003). A standard methodology for converting HWP mass into carbon equivalents is needed to compare data reported from different countries along with better estimates for country-specific trends in landfill waste.

5 Calculating Useful Lifetimes of HWPs

The figures above give us a rough estimate of the potential yearly input to the global carbon stock of HWPs. However, since these calculations fail to recognize the finite life of HWPs, these rough estimates are inflated. Lifespans of HWPs vary significantly by product type and must be accounted for accordingly. The carbon embodied in short-lived products can be released quickly back into the atmosphere after rapid decomposition, while long-lived products can store carbon for many years. Some wood or paper items such as antiquities and historic buildings are expected to have very long lives (in excess of 100 years)

(Skog et al. 1998). However the majority of paper products have a high rate of retirement, lasting only weeks (Marland and Marland 2003).

The lifespan attributed to products has a major impact on the outcome of estimates on the stock of HWPs. Although it is critical to determine the lifespans of various HWPs, it is difficult due to a lack of data on product use and disposal (Stern 2008). In response, some believe that HWP lifespans should not be viewed as empirical but as parameter values used in models (Pingoud et al. 2003). Data on HWP use suggest that rate of retirement of HWPs from end uses is more or less constant for a period, then accelerates for a while near the median life, and finally slows down after the median life (Skog and Nicholson 2000). As a result, the average lifespan of HWPs is much shorter than some models would suggest (Pingoud et al. 2003). Because of the difficulty of determining lifespans, it is common to see conflicting values for the lifespan of the same products in different studies. For example, a review of studies has shown that the estimates of average lifespan of pallets range from 2 to 20 years (Pingoud et al. 2003).

Data on average lifespan can be used to model how HWPs are discarded and ultimately oxidized. Much of the literature uses the term decay to describe both the mathematical characterization of retirement of products from use as well as the biophysical decomposition of products (Dias et al. 2009). The use of the term decay in this way reflects the fact that researchers analyzing HWP lifespans frequently quantify both the length of time that a product is used and the time during which HWP generates carbon emissions during waste management using the same model. The decay parameters for products in use are nonetheless different from those out of use (in landfills, for example) where decay of HWPs may be halted almost completely. Most studies, however, do not separately model the decay of HWPs that are out of use (Pingoud et al. 2003). Instead these studies model the retirement of HWPs and assume that decomposition occurs at different rates as a function of the product's retirement function.

The type of decay model used has a significant impact on estimation of the HWP carbon stock as it determines the timing of carbon releases through oxidation during decomposition. Numerous methods for modeling the carbon release of HWPs exist. (Dias et al. 2009).

One method of modeling HWP oxidation is to assign an exponential decay rate to a product. This is often done by assigning each type of HWP a carbon half-life which represents the time in which half of the carbon embodied in the end-use product is no longer present and has been emitted back into the atmosphere. This exponential decay model assumes that 90% of the carbon in HWPs is released in 3.3 times the assigned half life. Under this model, carbon release begins immediately once a product is in use and occurs at a greater rate earlier on in the life of the product and slows as the product progresses through and end of life. Another approach assumes that products of this type all have the same age, which is set to the product's average lifespan. In the model, 100% of the carbon remains embodied in the HWP until it is discarded, at which time all the carbon in the HWP is then released into the atmosphere. A third method follows a linear model in which a percentage of the initial amount of carbon in the HWP is released each year. The year in which all the carbon has been released is the maximum lifespan of the HWP type. Half of the time needed to reach the maximum lifespan is the product's average lifespan. The emissions profile of these models can be linear, exponential or equal (Fig. 12.2) (Skog and Nicholson 2000).

The different methods in modeling carbon release from HWPs clearly show how assumptions concerning product lifespan can significantly alter estimates. The most rudimentary model is that which assumes products of a certain type have an equal age and release 100% of their carbon at the time of retirement. This model does not account for carbon that is released into the atmosphere from products that are discarded before reaching their average lifespan. This method may mask carbon emissions that are occurring from HWP end-of-life processes and may inflate estimates of the annual increase in the HWP carbon stock. The linear and exponential decay functions both have carbon emissions occurring from the start of a

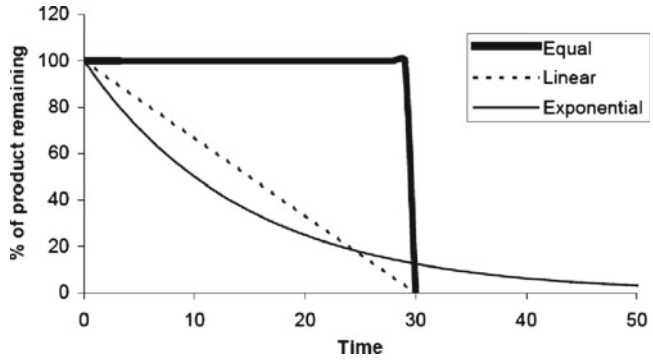

Fig. 12.2 A graphical representation of how carbon release is modeled using different methods of incorporating HWP product life into stock calculations (*Source*: Pingoud et al. 2003. Reprinted with permission)

product's life, which accounts for products that are discarded much earlier than those reaching the average lifespan. The exponential function creates a scenario where carbon emissions occur much faster in the beginning and slow as a product gets closer to reaching its average lifespan. HWP retirement most likely follows this decay function more closely as HWP retirement accelerates before reaching median life and finally slows down after the median life (Skog and Nicholson 2000). It must be noted that these decay functions do not effectively model conditions in landfills or bioenergy facilities.[8] Thus, they should only be used to model the rate of HWP retirement from use, which could then be incorporated into a larger model that more accurately portrays carbon emissions from HWP once they are discarded.

Determining accurate HWP lifespan values in order to create models that simulate real life conditions is difficult given a lack of data. There is room for vast improvement in reporting methods. Better data on product life for industrial uses of HWPs such as pallets may become available in the future as companies begin to label them with bar codes containing a pallet's age. Carbon markets may also encourage companies to keep better data on product life as they may in the future be able to sell temporary carbon credits based on their HWP stock. Efforts to develop global reporting standards and data sets on product life spans are underway (Murakami et al. 2010; Oguchi et al. 2010) Inclusion of HWPs in climate mitigation policy will require increased reporting which will lead to better data, allowing for more accurate product lifespans (Kuchli 2008).

6 End-of-life Pathways for HWPS

Harvested wood products can take several different pathways when they are discarded (CEPI 2007). Recent research has expanded the system boundaries of analysis to account for the different end-of-life pathways which can postpone carbon release of HWPs, store carbon indefinitely, displace fossil fuels, or even produce emissions at a significant level. HWPs can be recycled, burned (with or without energy recovery), composted, or disposed of in a dump or landfill (Fig. 12.3). Each of these pathways has different implications for carbon emissions. Calculations that do not account for these pathways are not accurately capturing the carbon effects. This is especially true in regards to the production of CH_4 resulting from the landfilling of HWPs. Research that includes end-of-life pathways has shown that from 2000 to 2005 the

[8]For example, carbon emissions from incineration of HWPs is immediate whereas, as described below, the release from landfilled products follows a longer and more complicated path. Composting of HWPs present an intermediate case.

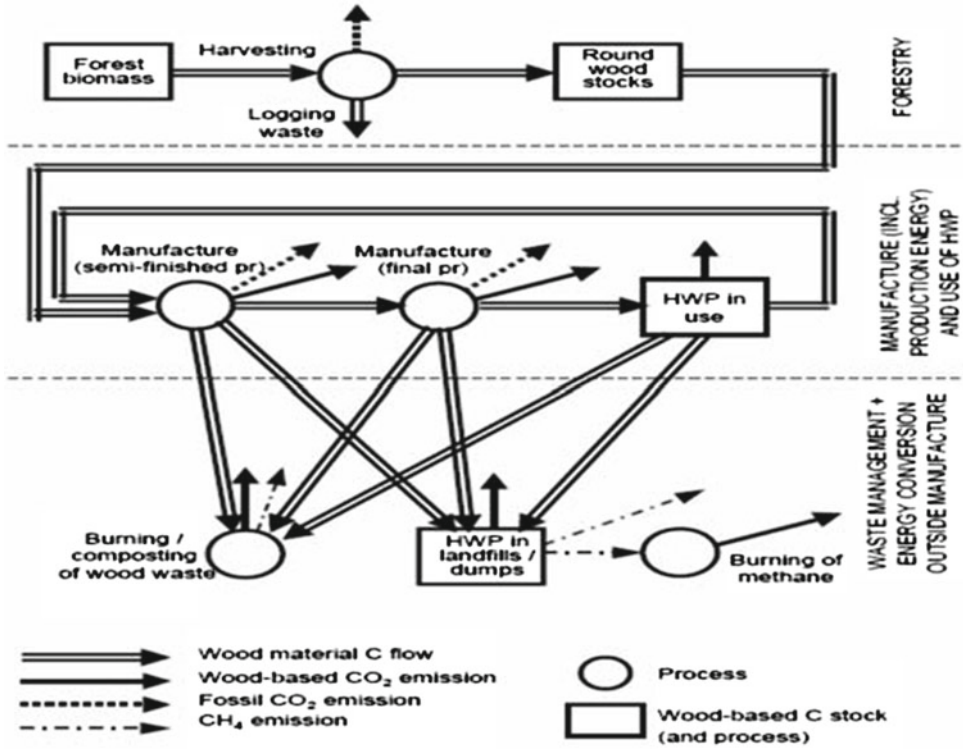

Fig. 12.3 Schematic representation of a lifecycle of HWP (*Source*: Pingoud et al. 2003)

global HWP stock had an average net increase of 147 Tg C/year, which is equivalent to 540 Tg CO_2/year (Miner 2008). These findings are at the higher end of the range compared to earlier studies due to the study's assumptions on landfills.

6.1 Burning HWPs

HWPs have the potential to be burned as a fuel. Short-lived wood products follow this pathway more often than long-lived products. Skog and Nicholson (2000) estimate that in 1993 in the United States, over 24% of paper and paperboard waste (after recycling) was burned. Although burning discarded wood or paper for energy is a carbon-emitting activity, it may result in lower net emissions if it has displaced more carbon-intensive fuel types (i.e., substitution effect). Using discarded HWPs for energy also reduces the amount that is put in landfills thus reducing the production of potent CH_4 gas. In order to evaluate whether burning HWPs for energy is superior to burning an alternative energy, a comparison of the two fuel chains must use a consistent methodology and a consistent definition of system boundaries.

6.2 Recycling

Recycling programs prolong the lifespan of carbon in HWPs, which keeps carbon stored in the product chain and extends carbon sequestration benefits. Recycling processes typically transform HWPs into products of lower wood content. This process can be repeated until the HWP is used to create bioenergy or otherwise disposed. This is known as a cascade effect (Kuchli 2008). HWP

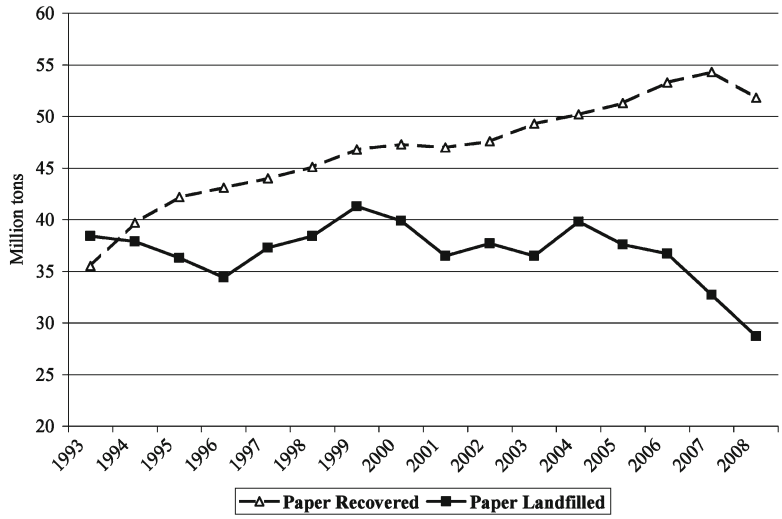

Fig. 12.4 Paper recovery vs. landfilling in the U.S. 1993–2007 (*Source*: Created with data from the American Forest & Paper Association, http://www.paperrecycles.org/stat_pages/recovery_vs_landfill.html)

recycling can thus reduce the rate of landfilling. This in turn reduces the amount of CH_4 produced by HWPs in landfills (CEPI 2007). This is particularly true for paper products, as these materials produce higher levels of CH_4 than landfilling of solid wood products (Skog et al. 2004).

As HWPs cascade into products of lower wood densities, however, their viability to be recycled is reduced. Once paper has reached a very low grade, such as tissue, it can no longer be recycled. Not surprisingly, in part because of the cascading effect and the downcycling of HWPs, low-grade paper products typically constitute a third of municipal solid waste (MSW) in landfills (Pingoud et al. 2003; EPA 2008).

The type of HWP plays a major role in whether or not it will be recycled. At the moment, recycling is only seen as a viable option for paper products.[9] The EPA reported that in 2007, 83 million tons (U.S.) of waste paper and paperboard were generated, of which 45 million tons (U.S.) (or 54%) were recovered through recycling (EPA 2008). In contrast, the recycling rate for HWPs used in construction is significantly lower. In 2007, the United States recycled only 1.3 million tons of durable wood products from the nearly 14 million tons generated (9%) (EPA 2008). This huge disparity in recycling rates is due to the nature of the products themselves. Newspaper is easily sorted and collected, while wood from construction demolition is very difficult to separate and re-use. Notably, data from the National Council for Air and Stream Improvement (NCASI) shows that while paper recovery is rising rapidly, the amount of paper products in landfills has decreased only nominally (Fig. 12.4) (Miner 2008). Still, reductions in the amount of HWP landfilled are expected to occur over time as HWP recycling processes modernize and become fueled by residue losses from the recycling process.

6.3 Landfills

Landfills have been criticized for their negative environmental impacts since the beginning of the environmental movement. Today, however, there are those in the scientific community who suggest that landfills could potentially act as a carbon *sink* for HWPs because HWP decomposition

[9]There is some niche market recycling of solid wood products—lumber from old buildings, etc. and some efforts to reclaim hardwood from pallets. (Technically, this is reuse and not recycling.)

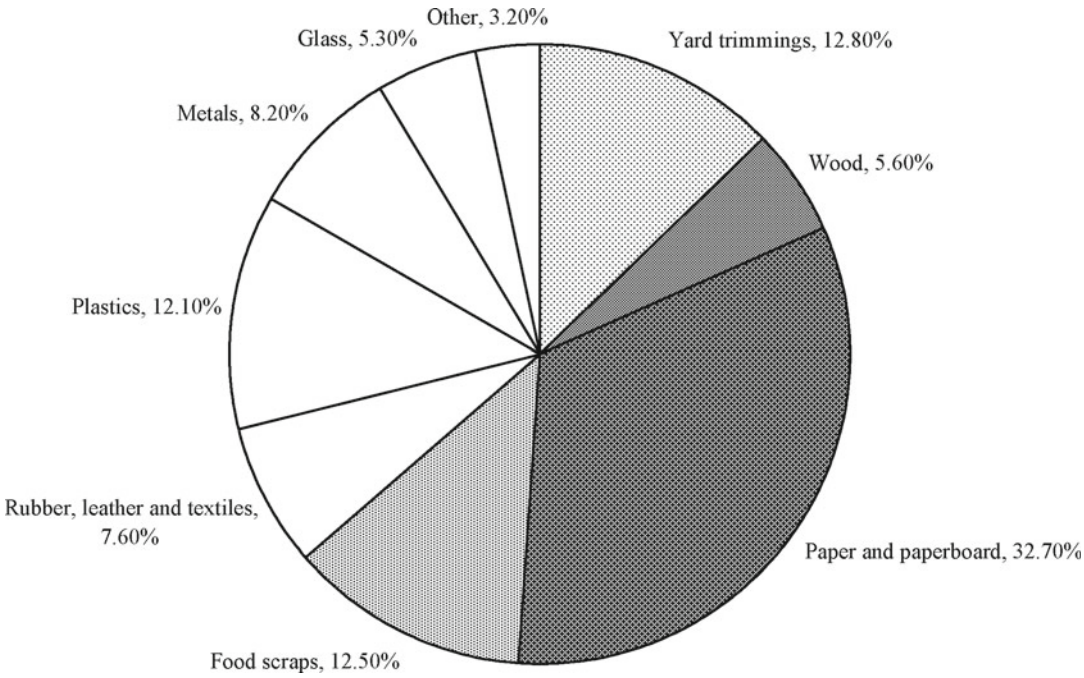

Fig. 12.5 Composition of waste materials destined for final disposal as Solid Waste in the United States, 2007 (*Source*: Data from EPA 2008)

can be very slow. Modern landfills are typically engineered to minimize the infiltration of water and the absence of moisture impedes biodegradation. Studies have shown that most wood products, when disposed of in a modern landfill, will experience a very slow decay (Bogner et al. 1993; Ximenes et al. 2008). This finding can have significant implications for calculating the stock of carbon in HWPs because it is estimated that biomass materials, such as paper, food, and wood, constitute about 63% of the municipal solid waste (MSW) in the U.S. (Fig. 12.5) (EPA 2008). The high proportion of HWPs in landfills further supports the case to expand the boundaries of analysis to include HWP end-of-life pathways. If CH_4 is captured and used for energy, carbon emission reductions can occur as carbon remains locked in HWPs at the same time that energy generated from landfill gases can displace fossil fuel emissions from traditional energy sources. Despite the attractiveness of using landfill gases for fuel, recent estimates indicate that only around 5 Tg C is captured worldwide, versus 15–20 Tg C of annual emissions from landfills (Spokas et al. 2006; Willumsen 2004). The large discrepancy between landfill gas (LFG) production and capture is best understood by analyzing how landfills work as well as current disposal practices. This may also help to forecast the likely impact from policies under debate to encourage or permit increased landfilling of HWPs.

6.4 Landfill Science

In a landfill, solid waste is buried. While this allows some biodegradable fractions of the waste to decompose via a complex series of microbial and abiotic reactions, the anaerobic conditions prevent a significant amount of decomposition. CH_4, or methane, is formed by methanogenic microorganisms under anoxic conditions, either

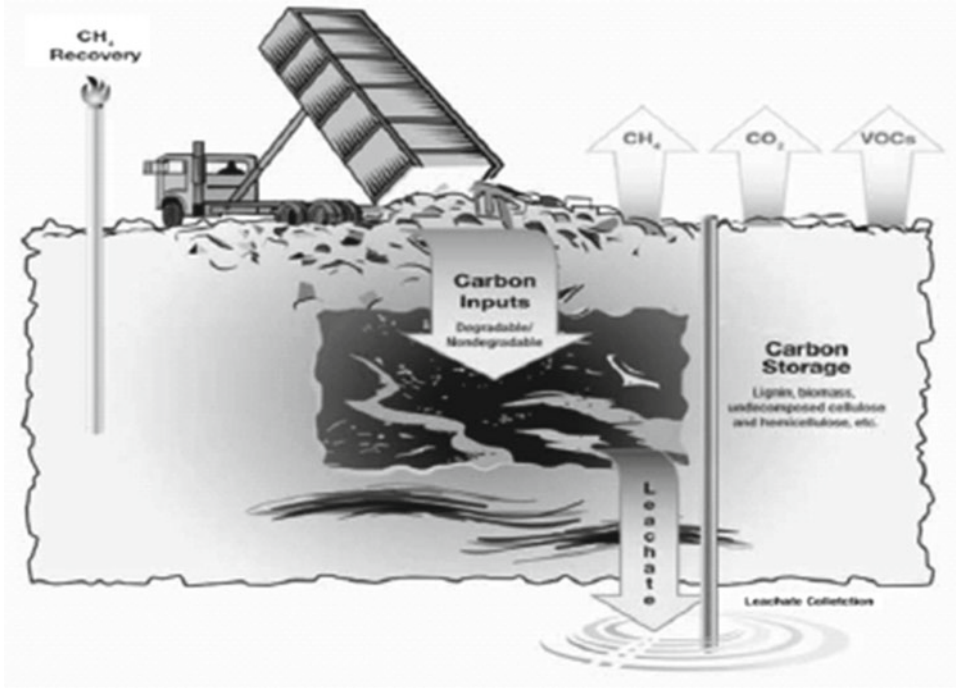

Fig. 12.6 Landfill carbon mass balance (*Source*: EPA 2006)

through the direct cleavage of acetate into CH_4 and carbon dioxide or the reduction of CO_2 with hydrogen (Fig. 12.6) (Spokas et al. 2006).

Since new layers of waste cannot be instantly covered, the waste is exposed to oxygen which allows white-rot fungus to decay wood. This type of decay, however, is limited because the available oxygen is rapidly consumed by the fungus, leaving only anaerobic bacteria. While anaerobic bacteria can break down hemicellulose and cellulose, these organisms cannot reach these materials if they are enclosed in lignin (Skog and Nicholson 2000). As a result, solid wood placed in landfills experiences low rates of decay. In newsprint, however, lignin content is only 20–27% and chemically-pulped paper has virtually no lignin, which results in a greater likelihood of decay than solid wood products, despite anaerobic conditions. Still, both wood and paper products experience low decay rates; in general, less than 50% of the carbon in these products is estimated to be ultimately converted to CO_2 or CH_4 (Table 12.8) (Skog and Nicholson 2000).

Table 12.8 Estimated maximum proportions of wood and paper that are converted to CO_2 or CH_4 in landfills

Product type	Maximum carbon converted (%)
Solid wood	3
Newsprint	16
Coated paper	18
Boxboard	32
Office paper	38

Source: Skog and Nicholson (2000)

Emissions created from anaerobic conditions are referred to generally as "landfill gas" (LFG) and encompass multiple gases, predominantly CO_2 and CH_4. According to Skog and Nicholson (2000), the proportion of carbon that is emitted as CO_2 and CH_4 in the gaseous product of MSW in landfills is skewed towards CH_4 at a rate of 1.5:1. Other studies suggest that the proportional difference between the two is not as great and that 1:1 should be used for commercial purposes (Johannessen 1999; Themelis and Ulloa 2007).

It is also important to note that emissions of various greenhouse gases occur on different temporal scales. On the one hand, CO_2 is released

quickly as decomposition occurs while oxygen is still present in the system. Studies estimate that half of the total CO_2, is emitted in the first 3 years while the rest is emitted continually over time (Skog and Nicholson 2000). Methane, on the other hand, is released very slowly over time once all the oxygen is depleted, with half the total CH_4 emitted in approximately 20 years (Micales and Skog 1997). Moreover, Skog and Nicholson claim that 10% of the CH_4 is converted to CO_2 by microorganisms as it moves out of the landfill, which makes the landfill cover a de facto converter. According to Johansson, the conversion capacity for a landfill top cover varies depending on soil texture, moisture content, and the amount of organic matter available in the soil. Covers with porous soils and organic matter have achieved complete oxidation of methane (Johannessen 1999).

While LFG generation poses a problem in terms of carbon emissions, high LFG generation levels are desirable for operators of LFG recovery systems, particularly since such systems are capital intensive and often financed by energy sales. Although theoretically, 1 ton of biodegradable carbon can produce 1,800 m^3 of LFG, in practice, this number is much lower because of uneven and incomplete biodegradation. As a result, 200 m^3 is generally accepted as the maximum volume of LFG produced from 1 ton of land filled MSW (Johannessen 1999). Several factors influence the rate of capture to total volume of LFG generated. These include LFG losses to the atmosphere through the surface or through lateral gas migration; pre-closure loss due to decomposition of organic material under aerobic conditions; aerobic decomposition of the near-surface layer (e.g., air intrusion due to gas extraction); and washout of organic carbon via leachate (Johannessen 1999). All of these can reduce the potential LFG capture rate, and often tip the balance of whether landfills reduce emissions from carbon storage or serve as large sources of carbon emissions.

As of the beginning of this century there are more than 350 landfills in the United States with gas recovery plants, and more than 1,100 worldwide (Spokas et al. 2006). These landfills are very diverse with respect to the amounts of material placed in the landfill, the type of material, degradation rate, and LFG capture system. Moreover, within individual landfills, decomposition rates can vary even in adjacent areas of a landfill (Micales and Skog 1997). This variation makes it difficult to assign an average capture rate to all landfills (CEPI 2007). As one example, the EPA's Waste Reduction Model (WARM) uses a default value of 75% LFG capture rate. Compared to other reports, this figure is higher than average and likely varies greatly from region to region within the United States (Themelis and Ulloa 2007). Other studies are more conservative and claim that normal recovery rates are thought to range from 40% to 50% by volume (Johannessen 1999). In this case, even landfills with advanced cover systems are thought to recover just slightly over 60% of the LFG generated. However, a more recent study in France found LFG recovery rates ranged from 41% to 94% of the theoretical CH_4 production and were highly dependent on the engineered cover design (ADEME 2008). It further suggested that average LFG recovery rates could exceed 90% by excluding the poorest performing cover design from the study.

LFG generation and capture rates vary across a temporal scale. This has led the French environment agency (ADEME) to create different default values to account for landfill design and stage of operation with values ranging from 35% to 90% recovery (Spokas et al. 2006). The literature on this subject clearly shows that there is a high level of uncertainty when it comes to calculating emissions from landfills. However, industry experts believe that methane emissions from wood products in landfills will become a smaller part of the total carbon foot print from HWPs as technology improves and more LFG is captured (Miner 2008).

In 2007, 3.7 billion m^3 of methane was captured from landfills in the United States, of which 70% was used to generate thermal or electrical energy (Themelis and Ulloa 2007). The rest of the captured methane was flared since it was thought to have no economic value. Flaring of LFG and using it in energy production reduces the methane content to carbon dioxide and water (Johannessen 1999). Despite the fact that flaring reduces the potency of the methane, it still

produces high levels of CO_2 emissions. It must also be noted that there are nearly 1,400 landfills in the United States (EPA 2008) that do not capture and flare any biogas. It is likely that HWPs in these sites are generating high levels of CH_4 emissions.

Including end-of-life conditions in HWP carbon stock models is critical due to the large potential emissions from landfills. Carbon released during end-of-life processes does not follow the simple decay functions most often used to model HWP retirement and discard. As described above, landfills may have varying conditions which will have a large impact on HWP carbon stocks. How these landfills are incorporated into HWP carbon stock accounting is key. In the United States, for example, only about 20% of 1,754 landfills are currently capturing LFG (EPA 2008). This figure raises serious doubts on the default LFG capture rate of 75% used by the EPA in the WARM model. Unfortunately, unrealistic default LFG capture rates have the potential to lead to significant miscalculation of the role of HWPs not only on a country basis but globally. Policies that promote or permit landfilling of HWPs could be aligned with policies that require high percentage LFG capture rates to ensure net emission reductions.

7 Management and Policy Implications

As policymakers focus on the role of forests and HWPs in mitigating climate change, additional research is needed to fully understand the relationships among climate policy, the forest products industry, consumers, and forests. Management and policy implications are summarized below.

7.1 Management Implications

- *Forestlands*. The potential to sequester carbon in forests is much larger than the potential to sequester carbon in forest products. Minor changes in forest extent have much greater impacts on GHG emissions than the forest products industry. Some researchers (Kauppi and Sedjo 2001; NCASI 2007) refer to the beneficial role that the forest products industry plays in maintaining sustained-yield forestland.
- The production and use of HWPs may postpone carbon emissions as carbon is stored in HWPs for a period after the initial harvest of roundwood. If the production of HWP exceeds the rate of retirement, then the amount of carbon bound in the HWP stock increases.
- Current methods for estimating carbon in HWPs are highly variable. A lack of data on product use makes it difficult to model HWP stocks; even assumptions on average wood density can significantly alter estimates of the conversion of HWP mass into carbon.
- *Substitution*. Each major building materials industry (wood, steel, and concrete) has published studies suggesting that their products are superior from the perspective of climate change mitigation. Given that climate considerations are currently an externality, more research is needed to understand what factors drive materials selection and whether a carbon price signal is sufficient to overcome these factors.
- *Landfills*. From the perspective of greenhouse gas emissions landfills with effective landfill gas collection could potentially be an acceptable final destination for discarded HWPs since HWPs have shown to have very low rates of decay in landfills. The production of LFG also fits into the "cascaded use of HWPs" framework because it can be converted into energy, displacing fossil fuels and further reducing global emissions. At the same time, it must be demonstrated that unintended consequences such as increased emissions elsewhere in the wood product life cycle are not triggered by strategies intended to enhance the HWP carbon stock at end of life.

7.2 Policy Implications

- The rise of biomass energy use in the forest products industry, as well as increasing utilization of wood products, has been driven

by several factors. These include the competitive nature of the industry and the need to lower costs while seeking new sources of revenues, particularly for by-products and co-products that had historically not generated an economic return to the industry. Under certain economic conditions, however, forest products manufacturers may be inclined to alter manufacturing processes, which could result in incremental emitting activities under certain scenarios, particularly if it lowers costs for a profit maximizing entity.

- To date, policymakers have not fully considered the role harvested wood products can play in climate change mitigation and have not linked forest management practices to the full life cycle of harvested wood products. Incentives could be considered to support the use of recycled materials, to encourage such activities as product substitutions, industrial energy efficiency, and to encourage biomass fuel sources.
- There are many factors that will favor or disfavor wood as a construction material or energy source. These include relative price, technology, economic growth, policy, market efficiency, socioeconomic factors, and quality and quantity of energy and materials (Gustavsson et al. 2006). Recognizing that wood products are still largely a cyclical industry driven by global GDP, policies could begin to introduce longer- term, secular demand for wood products that encourage investment in wood that is both economically and environmentally sound.
- Recycling should be promoted heavily in policy intended to enhance the HWP carbon stock since recycling postpones carbon emissions of even short-lived HWPs. Recycling also fits very well in the "cascaded use of HWPs" concept where HWP are transformed multiple times within a tight recycling chain and finally converted into bioenergy.

References

ADEME, 2008. State-of-the-practices and implementation recommendations for non hazardous waste management using bioreactors landfills. French Environment and Energy Management Agency (Agence de l'Environnement et de la Maîtrise de l'Energie). http://www2.ademe.fr/servlet/getDoc?cid=96&m=3&id=51261&p2=17618&ref=17618

Bogner JE, Spokas K (1993) Landfill CH_4: rates, fates and global carbon cycle. Chemosphere 26:369–386

Brown R (1999) Timberland shedding trend: why are paper firms selling their land? Northern Logger Timber Processor 48:2

Burschel P, Kursten E, Larson BC, Weber M (1993) Present role of German forests and forestry in the national carbon budget and options to its increase. Water Air Soil Pollut 70:325–340

Carlson E, Heikkinen P (1998) Energy Consumption by the Finnish Pulp and Paper Industry and Comparison of Electricity Prices in Certain European Countries (Report in Finnish, Abstract in English), The Finnish Pulp and Paper Research institute, Sustainable Paper Report 21. 81 p

CEPI (2007) Framework for the development of carbon footprints of paper and board products

Corder SE, Scroggins TL, Meade WE, Everson GD (1972) Wood and bark residues in Oregon: trends in their use, research paper 11. Oregon State University, Forest Research Laboratory

Cote WA, Young RJ, Risse KB, Costanza AF, Tonelli JP, Lenocker C (2002) A carbon balance method for paper and wood products. Environ Pollut 116:S1–S6

Davidsdottir B, Ruth M (2004) Capital vintage and climate change policies: the case of US pulp and paper. Environ Sci Policy 7:221–233

Denman KL, Brasseur G, Chidthaisong A, Ciais P, Cox P, Dickinson RE, Haugustaine D, Heinze C, Holland E, Jacob D, Lohmann U, Ramachandran S, da Silva Dias PL, Wofsy SC, Zhang X (2007) Couplings between changes in the climate system and biogeochemistry. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Avery KB, Tignor M, Miller HL (eds) Climate change 2007: the physical science basis, contribution of working group I to the fourth asssessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, pp 499–587

Dias AC, Louro, Margarida, Arroja L, Capela I (2009) Comparison of methods for estimating carbon in harvested wood products. Biomass and Bioenergy 33:213–222

Energy Information Administration (2008) Forest products industry analysis brief. http://www.eia.doe.gov/emeu/mecs/iab98/forest/. Accessed 30 Dec 08

EPA (2006) Solid waste management and greenhouse gases: a life-cycle assessment of emissions and sinks, 3rd edn. United States Environmental Protection Agency, Washington, DC

EPA (2008) Municipal solid waste in the United States: 2007 facts and figures. U.S. Environmental Protection Agency, Office of Solid Waste, EPA530-R-08-010

Falk RH, McKeever DB (2004) Recovering wood for reuse and recycling: a United States perspective. In: Gallis C (ed) Proceedings of management of recov-

ered wood recycling, bioenergy and other options. University Studio Press, Thessaloniki, pp 29–40
FAO (2004) FAOSTAT. United Nations Food and Agriculture Organization. http://faostat.fao.org/site/628/default.aspx
FAO (2007) State of the world's forests. United Nations Food and Agriculture Organization, Rome
Farahani S, Worrell E, Bryntse G (2004) CO_2-free paper? Resour Conserv Recycl 42:317–336
Finkral AJ, Evans AM (2008) Effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. For Ecol Manag 255:2743–2750
Fischlin A, Midgley GF, Price JT, Leemans R, Gopal B, Turley C, Rounsevell MDA, Dube OP, Tarazona J, Velichko AA (2007) Ecosystems, their properties, goods, and services. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE (eds) Climate change 2007: impacts, adaptation and vulnerability. Contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, UK, pp 211–272
Gardner D, Cowie A, Ximenes F (2004) A new concept for determining the long-term storage of carbon in wood products. Cooperative Research Centre for Greenhouse Accounting
Grantham JB, Howard JO (1980) In: Sarkanen KV, Tillman DA (eds) Logging residues as an energy source in biomass conversion, vol 2. Academic, New York
Green C, Avitabile V, Farrell EP, Byrne KA (2006) Reporting harvested wood products in national greenhouse gas inventories: Implications for Ireland. Biomass and Bioenergy 30:105–114
Gustavsson L, Sathre R (2006) Variability in energy and carbon dioxide balances of wood and concrete building materials. Build Environ 41:940–951
Gustavsson L, Madlener R, Hoen HF, Jungmeier G, Karjalainen T, KlÖhn S, Mahapatra K, Pohjola J, Solberg B, Spelter H (2006) The role of wood material for greenhouse gas mitigation. Mitig Adapt Strateg Glob Change 11:1097–1127
Han FXX, Lindner JS, Wang CJ (2007) Making carbon sequestration a paying proposition. Naturwissenschaften 94:170–182
Howard J (2006) Estimation of U.S. timber harvest using roundwood equivalents In: McRoberts RE, Reams GA, Van Duesen PC, McWilliams WH (eds) Proceedings of the sixth annual forest inventory and analysis symposium; 2004 September 21–24; Denver, CO. Gen Tech Rep WO-70 Washington, DC: U.S. Department of Agriculture Forest Service 126 p
Ince PJ (1995) What Won't Get Harvested, Where and When: The effects of increased paper recycling on timber harvest. PSWP working paper #3. New Haven: Yale University School of Forestry & Environmental Studies.
International Energy Agency (IEA) (2003) Key world energy statistics 2003. International Energy Agency, Paris
IPCC (1996) Climate change 1995: impacts, adaptations and mitigation of climate change. Cambridge University Press, Cambridge, UK
IPCC (2000) Robert T. Watson, Ian R. Noble, Bert Bolin, Ravindranath NH, David J. Verardo, David J. Dokken (eds) Cambridge University Press, UK. pp 375
IPCC (2007) Forestry. In: Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge/New York
Irland L (2008) Professor of Forest Finance, Yale School of Forestry and Environmental Studies, personal communication
Johannessen LM (1999) Guidance note on recuperation of landfill gas from municipal solid waste landfills. World Bank Urban Development Division
Kauppi P, Sedjo R (2001) Technological and economic potential of options to enhance, maintain, and manage biological carbon reservoirs and geo-engineering. In: Climate change mitigation, IPCC working group panel III report, pp 301–345
Kline DE (2005) Gate-to-gate life-cycle inventory of oriented strandboard production. Wood Fiber Sci 37:74–84
Kuchli C (2008) Chair's conclusions and recommendations In: Hetsch S (ed) Harvested wood products in the context of climate change policies, United Nations Palais des Nations, Geneva, Switzerland
Lippke B, Perez-Garcia J (2008) Will either cap and trade or a carbon emissions tax be effective in monetizing carbon as an ecosystem service? For Ecol Manag 256:2160–2165
Luo L, Evd V, Huppes G, Udo de Haes HA (2009) Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol. Int J LCA 14:529–539
Marceau ML, VanGeem MG (2002) Life cycle assessment of an insulating concrete form house compared to a wood frame house. PCA R&D SN2571. Skokie, Illinois, Portland Cement Association
Marceau ML, VanGeem MG (2008) Comparison of the life cycle assessments of an insulating concrete form house and a wood frame house. PCA R&D SN3041. Skokie, Illinois, Portland Cement Association
Marland E, Marland G (2003) The treatment of long lived, carbon containing products in inventories of carbon dioxide emissions to the atmosphere. Environ Sci Policy 6:139–152
Matthews RW, Nabuurs GJ, Alexyeyev V, Birdsey RA, Fischlin A, Maclaren JP, Marland G, Price DT (1996) Evaluating the role of forest management and forest products in the carbon cycle. In: Apps MJ, Price DT (eds) Forest ecosystems, forest management and the global carbon cycle. NATO ASI Series Vol. 40, pp. 293–301. Springer-Verlag, Berlin Heidelberg
McKeever DB (2002) Domestic market activity in solid wood products in the United States, 1950–1998. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station
Meil J, Wilson J, O'Connor J, Dangerfield J (2007) An assessment of wood product processing technology

advancements between the CORRIM I and II studies. For Prod J 57:83–89
Micales J, Skog K (1997) The decomposition of forest products in landfills. Int Biodeterior Biodegrad 39:145–158
Miner R (2008) The carbon footprint of forest products. In: GAA environmental workshop, Geneva, 17–19 June 2008
Murakami S, Oguchi M, Tasaki T, Daigo I, Hashimoto S (2010) Lifespan of commodities, part I: The creation of a database and its review. J Ind Ecol 14(4):598–612
Nabuurs GJ, Pussinen A, van Brusselen J, Schelhaas MJ (2007) Future harvesting pressure on European forests. Eur J For Res 126:391–400
National Council for Air and Stream Improvement (2007) The greenhouse gas and carbon profile of the global forest products industry. NCASI Spec Rep 07–02:1–32
Nishioka Y, Yanagisawa Y, Spengler JD (2000) Saving energy versus saving materials: Life-cycle inventory analysis of housing in a cold-climate region of Japan. J Ind Ecol 4(1):119–136
Oguchi M, Murakami S, Tasaki T, Daigo I, Hashimoto S (2010) Lifespan of commodities, part II: Methodologies for estimating lifespan distribution of commodities. J Ind Ecol 14(4):613–626
Perez-Garcia J, Lippke B, Comnick J, Manriquez C (2005) An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. Wood Fiber Sci 37:140–148
Pingoud K, Lehtila A (2002) Fossil carbon emissions associated with carbon flows of wood products. Mitig Adap Strateg Glob Change 7:63–83
Pingoud K, Soimakallio S, Perala AL, Pussinen A (2003) Greenhouse gas impacts of harvested wood products: evaluation and development of methods. Espoo 2003. VTT Technical Research Center of Finland, Research Notes 2189, 120 p. + app. 16 p
Puettmann ME, Wilson JB (2005) Life-cycle analysis of wood products: cradle-to-gate LCI of residential wood building materials. Wood Fiber Sci 37:18–29
Rueter S (2008) Model for estimating carbon storage effects in wood products in Germany. In: Hetsch S (ed) Harvested wood products in the context of climate change policies, United Nations Palais des Nations, Geneva, Switzerland
Sampson RN, Apps M, Brown S, Cole CV, Downing J, Heath LS, Ojima DS, Smith TM, Solomon AM, Wisniewski J (1993) Workshop summary statement—terrestrial biospheric carbon fluxes—quantification of sinks and sources of CO_2. Water Air Soil Pollut 70:3–15
Sedjo RA (2008) Biofuels: think outside the cornfield. Science 320:1419–1420
Skog K (2008) Sequestration of carbon in harvested wood products for the United States. For Prod J 58:56–72
Skog K, Nicholson GA (2000) Carbon sequestration in wood and paper products. In: Joyce LA, Birdsey R, technical editors 2000. The impact of climate change on America's forests: a technical document supporting the 2000 USDA Forest Service RPA Assessment. Gen. Tech. Rep. RMRS-GTR-59. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, p 79–88
Skog KE, Nicholson GA (1998) Carbon cycling through wood products: the role of wood and paper products in carbon sequestration. For Prod J 48:75–83
Skog KE, Pingoud K, Smith JE (2004) A method countries can use to estimate changes in carbon stored in harvested wood products and the uncertainty of such estimates. Environ Manag 33:S65–S73
Spokas K, Bogner J, Chanton JP, Morcet M, Aran C, Graff C, Golvan YML, Hebe I (2006) Methane mass balance at three landfill sites: what is the efficiency of capture by gas collection systems? Waste Manag 26:516–525
Stern N (2006) Stern review on the economics of climate change. UK Office of Climate Change. http://www.hm-treasury.gov.uk/stern_review_final_report.htm
Stern T (2008) Contribution of the Austrian forest sector to climate change policies – industrial marketing implications. In: Hetsch S (ed) Harvested wood products in the context of climate change policies, United Nations Palais des Nations, Geneva, Switzerland
Subak S, Craighill A (1999) The contribution of the paper cycle to global warming. Mitig Adap Strateg Glob Change 4:113–136
Themelis NJ, Ulloa PA (2007) Methane generation in landfills. Renewable Energy 32:1243–1257
United States Environmental Protection Agency (2008) Inventory of U.S. greenhouse gas emissions and sinks: 1990–2006. USEPA #430-R-08-005
Upton B, Miner R, Spinney M, Heath LS (2008) The greenhouse gas and energy impacts of using wood instead of alternatives in residential construction in the United States. Biomass Bioenergy 32:1–10
Walker T, Cardellichio P, Colnes A, Gunn J, Kittler B, Perschel B, Recchia C, Saah D (2010) Biomass sustainability and carbon policy study. NCI-2010-03. Manomet Center for Conservation Sciences, Brunswick
Watson R, Noble IR, Bolin B, Ravindrath NH, Verardo DJ, Dokken D (2000) IPCC Special Report on Land Use, Land-Use Change and Forestry. IPCC, Geneva
Wernick IK, Waggoner PE, Ausubel JH (1997) Searching for leverage to conserve forests: the industrial ecology of wood products in the United States. J Ind Ecol 1(3):125–145
Willumsen H (2004) Number and types of landfill gas plants worldwide. Presentation to World Bank, November 18. http://deponigas.dk/uploads/media/Number_and_Types_of_LFG_Plants_Worldwide.pdf. Accessed 13 Oct 2011
Wilson J (2005) Documenting the environmental performance of wood building materials. Wood Fiber Sci 37:1–2

Winjum JK, Brown S, Schlamadinger B (1998) Forest harvests and wood products: Sources and sinks of atmospheric carbon dioxide. For Sci 44(2): 272–284

Woodbury PB, Smith JE, Heath LS (2007) Carbon sequestration in the US forest sector from 1990 to 2010. For Ecol Manag 241:14–27

World Resources Institute (2006) Climate analysis indicators tool (CAIT) on-line database version 3.0. World Resources Institute, Washington, DC. Available at. http://cait.wri.org

Ximenes FA, Gardner WD, Cowie AL (2008) The decomposition of wood products in landfills in Sydney, Australia. Waste Manag 28:2344–2354