NCASI

IMPACT. SCIENCE. SOLUTIONS.

November 2024

Technical Bulletin

Greenhouse Gas and Carbon Profile of the US Forest Products Industry: 1990 to 2020

Nº 1091

ncasi

NATIONAL COUNCIL FOR AIR AND STREAM IMPROVEMENT, INC.

Greenhouse Gas and Carbon Profile of the US Forest Products Industry: 1990 to 2020

N° 1091

November 2024

Prepared by

Reid Miner
NCASI Ret.

Barry Malmberg, PhD
NCASI

Adam Costanza
NCASI

Stephen Prisley, PhD
NCASI

For more information about this research, contact:

Barry Malmberg
NCASI
Principal Research Scientist
PO Box 271, Sheridan, WY 82801
(541) 249-3986
bmalmberg@ncasi.org

Kirsten Vice
NCASI
Vice President, Sustainability & Climate
Program and Canadian Operations
2000 McGill College Avenue, Sixth Floor
Montréal, QC H3A 3H3
(514) 907-3145
kvice@ncasi.org

To request printed copies of this report, contact NCASI at publications@ncasi.org or (352) 244-0900.

Cite this report as:

NCASI. 2024. *Greenhouse gas and carbon profile of the US forest products industry: 1990 to 2020*. Technical Bulletin No. 1091. Cary, NC: National Council for Air and Stream Improvement, Inc.

IMPACT. SCIENCE. SOLUTIONS.

EXECUTIVE SUMMARY

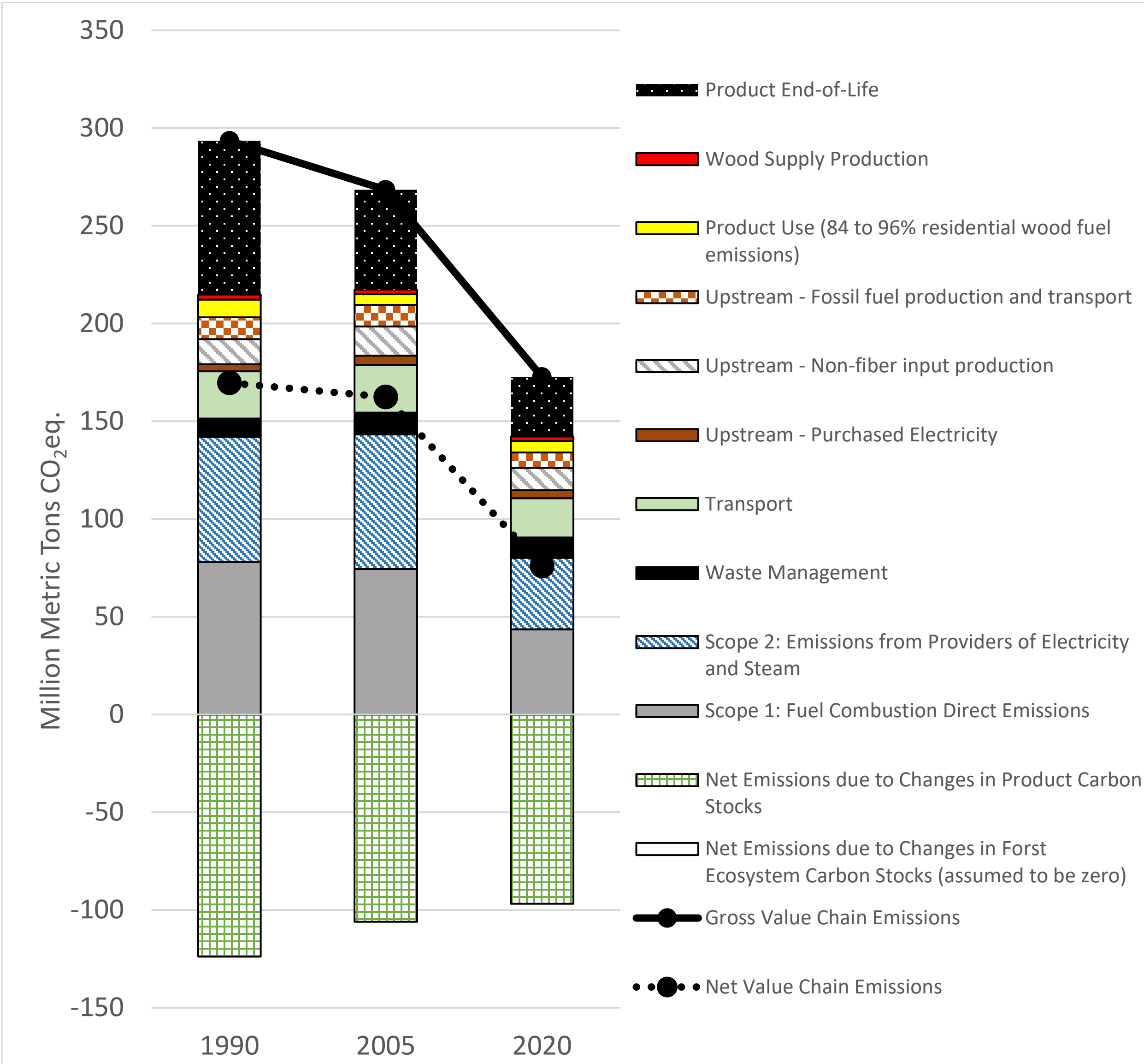
This report examines greenhouse gas emissions and sinks along the value chain of the US forest products industry for 1990, 2005, and 2020. The calculations improve, update, and expand earlier estimates for 1990 and 2005 (NCASI 2008; Heath et al. 2010). Total gross emissions for the US forest products industry value chain were 293, 268, and 172 million metric tons CO_2eq. in 1990, 2005, and 2020, respectively. Gross emissions in 2020 were 41% below 1990 emissions and 36% below 2005 emissions. The reductions were driven largely by (1) changes in energy sources used by the forest products industry and electricity producers, (2) reduced energy intensity in manufacturing, and (3) reduced methane emissions from products disposed in municipal solid waste landfills. Methane reductions were due to increased recycling and improved methane capture and destruction at municipal solid waste landfills.

While it is not possible to precisely determine the net overall effect of harvesting and forest management practices on forest carbon stocks, available data from the US Forest Service provide no evidence that forest ecosystem carbon stocks on wood-supplying lands are decreasing. Indeed, it appears likely that carbon stocks are increasing on land producing wood for the forest products industry, suggesting net removals of CO_2 from the atmosphere by wood-producing forests. Nonetheless, for the purposes of this study, net emissions of biogenic carbon from the forest attributable to the industry's activities on wood-producing land are assumed to be zero. This is the same conclusion reached in the previous study (NCASI 2008; Heath et al. 2010).

Removals of CO_2 from the atmosphere attributable to growth in stocks of carbon in forest products manufactured from domestically produced wood are calculated to have been 124, 106, and 97 million metric tons CO_2eq. in 1990, 2005, and 2020, respectively. The removals have declined over time primarily because of reduced product output. Including these removals in the calculations yields net transfers of CO_2eq. from the industry's value chain of 169, 162, and 75 million metric tons CO_2eq. in 1990, 2005, and 2020, respectively. Net transfers of greenhouse gases to the atmosphere in 2020 were 55% lower than in 1990 and 53% lower than in 2005. The graphical abstract shows the industry's profile in 1990, 2005, and 2020.

This update includes estimates for several types of emissions not included in the 2008 study. Where comparable estimates are provided in the two studies, the updated estimates are generally higher.

The annual inventory results, including those involving biogenic carbon stocks, are emissions in a single year resulting from production in that year and earlier years. Alternatively, one can also calculate the long-term storage of carbon in products manufactured in a single year. Based on radiative forcing over 100 years attributable to net emissions of biogenic carbon, products manufactured in 1990, 2005, and 2020 result in long-term storage of 45, 43, and 35 million metric tons of carbon equivalents, respectively (equivalent to 166, 156, and 128 million metric tons CO_2). Counteracting the benefits of this long-term biogenic carbon storage is the 100-year radiative forcing associated with biogenic methane emissions attributable to discarded forest products. The impacts of these methane emissions are estimated to be 86, 47, and 24 million metric tons of carbon equivalents for products manufactured in 1990, 2020, and 2020, respectively.



Greenhouse Gas and Carbon Profile of the US Forest Products Industry

NCASI

IMPACT. SCIENCE. SOLUTIONS.

SOMMAIRE

Le présent rapport examine les émissions et les puits de gaz à effet de serre de toute la chaîne de valeur de l'industrie des produits forestiers des États-Unis pour 1990, 2005 et 2020. Les calculs améliorent, actualisent et augmentent les estimations précédentes pour 1990 et 2005 (NCASI 2008; Heath et al. 2010). Les émissions totales brutes de la chaîne de valeur de l'industrie américaine des produits forestiers sont de 293, 268 et 172 millions de tonnes métriques de CO_2éq. en 1990, 2005 et 2020, respectivement. Les émissions brutes de 2020 sont 41% inférieures à celles de 1990 et 36% inférieures à celles de 2005. Les réductions sont dues en grande partie à (1) des changements dans les sources d'énergie utilisées par l'industrie des produits forestiers et par les producteurs d'électricité, (2) une réduction de l'intensité énergétique dans la fabrication, et (3) une réduction des émissions de méthane générées par les produits éliminés dans les sites d'enfouissement municipaux. La réduction des émissions de méthane sont dues à un recyclage accru et à un meilleur piégeage et une meilleure destruction du méthane dans les sites d'enfouissement municipaux.

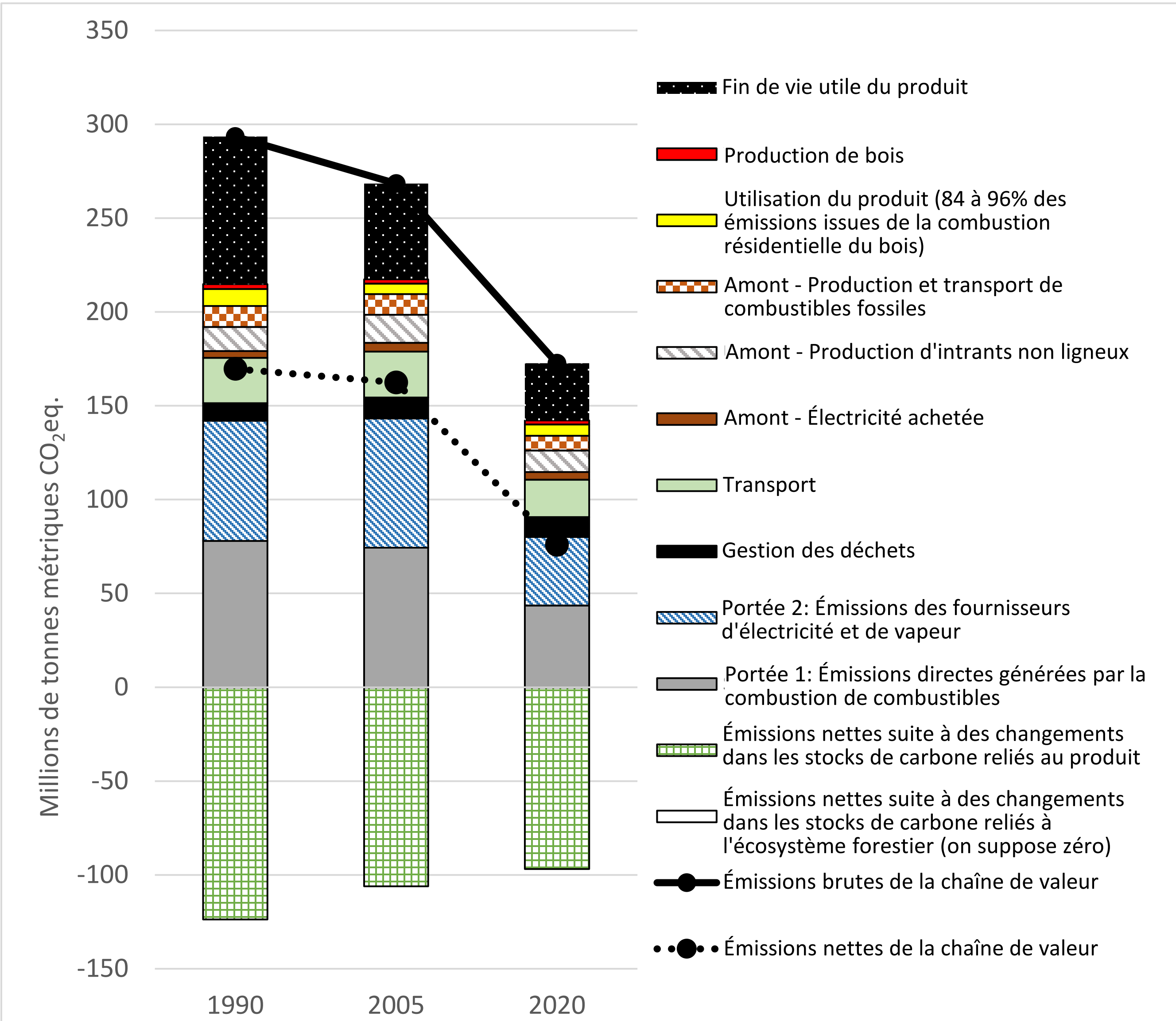
Bien qu'il ne soit pas possible de déterminer précisément l'effet global net de la récolte forestière et des pratiques d'aménagement forestier sur les stocks de carbone forestier, il n'y a pas d'évidence qu'il y a une réduction des stocks de carbone de l'écosystème forestier sur les terres productrices de bois lorsqu'on examine les données disponibles du Service américain des forêts. À vrai dire, il semble probable que les stocks de carbone soient en augmentation sur les terres qui fournissent le bois pour l'industrie des produits forestiers, ce qui semble indiquer qu'il y a absorption nette du CO_2 présent dans l'atmosphère par les forêts productrices de bois. Néanmoins, pour les besoins de la présente étude, nous supposons que les émissions nettes de carbone biogénique générées par la forêt qui sont attribuables aux activités de l'industrie sur les terres productrices de bois sont nulles (zéro). Il s'agit de la même conclusion à laquelle nous sommes parvenus dans l'étude précédente (NCASI 2008; Heath et al. 2010).

Le calcul de la quantité de CO_2 dans l'atmosphère absorbée grâce à l'augmentation des stocks de carbone dans les produits du bois fabriqués à partir de bois récolté aux États-Unis a donné 124, 106 et 97 millions de tonnes métriques de CO_2éq. pour 1990, 2005 et 2020, respectivement. L'absorption a diminué au cours du temps en raison principalement d'une diminution de la production. En incluant cette absorption dans les calculs, on obtient des transferts nets de CO_2éq. de la chaîne de valeur de l'industrie de 169, 162 et 75 millions de tonnes métriques de CO_2éq. pour 1990, 2005 et 2020, respectivement. Les transferts nets de gaz à effet de serre vers l'atmosphère en 2020 sont 55% inférieurs à ceux de 1990 et 53% inférieurs à ceux de 2005. Le résumé graphique ci-dessous montre le profil de l'industrie en 1990, 2005 et 2020.

La présente mise à jour comprend des estimations pour plusieurs types d'émissions qui n'étaient pas incluses dans l'étude de 2008. Lorsqu'il y a des estimations comparables dans les deux études, les estimations actualisées sont généralement plus élevées.

Les résultats du calcul de l'inventaire annuel, y compris les résultats reliés aux stocks de carbone biogénique, sont des émissions d'une année donnée qui découle de la production dans cette même année et années précédentes. Une autre solution consiste à calculer le stockage à long terme du carbone dans les produits fabriqués au cours d'une seule année. En se basant sur le forçage radiatif mesuré au cours d'une période de 100 ans et attribué aux émissions nettes de carbone biogénique, on obtient un stockage à long terme de 45, 43 et 35 millions de tonnes métriques d'équivalents carbone pour les produits fabriqués en 1990, 2005 et 2020, respectivement (équivalent à 166, 156 et 128 millions de tonnes métriques de CO_2). Le forçage radiatif mesuré sur une période de 100 ans

associée aux émissions de méthane biogénique attribuable aux produits du bois neutralise les bénéfices de ce stockage à long terme du carbone biogénique. On estime l'impact de ces émissions de méthane à 86, 47 et 24 millions de tonnes métriques d'équivalents carbone pour les produits fabriqués en 1990, 2020 et 2020, respectivement.



Le profil du carbone et des gaz à effet de serre de l'industrie des produits forestiers des États-Unis

GREENHOUSE GAS AND CARBON PROFILE OF THE US FOREST PRODUCTS INDUSTRY: 1990 TO 2020

TECHNICAL BULLETIN NO. 1091
NOVEMBER 2024

ABSTRACT

This report examines greenhouse gas emissions and sinks along the value chain of the US forest products industry for 1990, 2005, and 2020. Total gross emissions for the US forest products industry value chain were 293, 268, and 172 million metric tons CO_2eq. in 1990, 2005, and 2020, respectively. Gross emissions in 2020 were 41% below 1990 emissions and 36% below 2005 emissions. The reductions were driven largely by (1) changes in energy sources used by the forest products industry and electricity producers, (2) reduced energy intensity in manufacturing, and (3) reduced methane emissions from products disposed in municipal solid waste landfills. Methane reductions were due to increased recycling and improved methane capture and destruction at municipal solid waste landfills.

If CO_2 removals attributable to increases in biogenic carbon stocks are netted against gross emissions, net transfers of greenhouse gases from the industry's value chain are calculated to have been 169, 162, and 75 million metric tons CO_2eq. in 1990, 2005, and 2020, respectively. This report also includes estimates of net removals of CO_2 from the atmosphere over 100 years attributable to products manufactured in 1990, 2005, and 2020.

KEYWORDS

carbon footprint, forest carbon, carbon storage, US forest products industry, greenhouse gas emissions, life cycle assessment

RELATED NCASI PUBLICATIONS

The greenhouse gas and carbon profile of the US forest products sector. 2008. Special Report No. 08-05.

This page intentionally left blank

LE PROFIL DU CARBONE ET DES GAZ À EFFET DE SERRE
DE L'INDUSTRIE DES PRODUITS FORESTIERS DES ÉTATS-UNIS :
1990 À 2020

BULLETIN TECHNIQUE N^{O} 1091
NOVEMBRE 2024

RÉSUMÉ

Le présent rapport examine les émissions et les puits de gaz à effet de serre de toute la chaîne de valeur de l'industrie des produits forestiers des États-Unis pour 1990, 2005 et 2020. Les émissions totales brutes de la chaîne de valeur de l'industrie américaine des produits forestiers sont de 293, 268 et 172 millions de tonnes métriques de CO_2éq. en 1990, 2005 et 2020, respectivement. Les émissions brutes de 2020 sont 41% inférieures à celles de 1990 et 36% inférieures à celles de 2005. Les réductions sont dues en grande partie à (1) des changements dans les sources d'énergie utilisées par l'industrie des produits forestiers et par les producteurs d'électricité, (2) une réduction de l'intensité énergétique dans la fabrication, et (3) une réduction des émissions de méthane générées par les produits éliminés dans les sites d'enfouissement municipaux. La réduction des émissions de méthane sont dues à un recyclage accru et à un meilleur piégeage et une meilleure destruction du méthane dans les sites d'enfouissement municipaux.

Si l'absorption du CO_2 attribuable à l'augmentation des stocks de carbone biogénique est déduite des émissions brutes, on obtient des transferts nets de gaz à effet de serre de la chaîne de valeur de l'industrie de 169, 162 et 75 millions de tonnes métriques de CO_2éq. pour 1990, 2005 et 2020, respectivement. Le présent rapport comprend aussi des estimations de l'absorption nette du CO_2 présent dans l'atmosphère sur une période de 100 ans attribuable aux produits fabriqués en 1990, 2005 et 2020.

MOTS-CLÉS

analyse du cycle de vie, carbone forestier, émissions de gaz à effet de serre, empreinte carbone, industrie américaine des produits forestiers, stockage de carbone

AUTRES PUBLICATIONS DU NCASI

Le profil du carbone et des gaz à effet de serre du secteur des produits forestiers des États-Unis. 2008. Rapport spécial n^{o} 08-05 (seuls le Mot du Président et le Résumé sont en français).

This page intentionally left blank

CONTENTS

TABLES

FIGURES

GREENHOUSE GAS AND CARBON PROFILE OF THE US FOREST PRODUCTS INDUSTRY: 1990 TO 2020

1.0 INTRODUCTION

Fifteen years ago, the National Council for Air and Stream Improvement, Inc. (NCASI) collaborated with the US Forest Service to develop a profile of US forest products industry greenhouse gas (GHG) emissions and sinks. The results of that study were released to NCASI member companies in NCASI Special Report No. 08-05 (NCASI 2008) and subsequently published in the peer-reviewed literature (Heath et al. 2010).

Much has changed since the publication of that study. Energy sources used to power industry and produce electricity have become less GHG-intensive. The production of many forest products has decreased while recovery rates of used products have increased, and many other changes have occurred. In addition, methods used to estimate emissions have evolved. In this report, the profile developed 15 years ago is updated using the most appropriate data and methods available today.

An attempt has been made to include all relevant emission categories addressed in the GHG Protocol Corporate Standard, which covers Scope 1 and 2 emissions (GHG Protocol 2004) and the GHG Protocol Value Chain (Scope 3) Standard (GHG Protocol 2011). For the purposes of this report, Scope 1 emissions are those from manufacturing and converting operations in the forest products value chain. Scope 2 emissions are those released by producers of the electricity and steam purchased by the forest products industry. Scope 3 emissions are all other emissions in the forest products industry value chain. For a complete description of the emissions Scope concept, see the GHG Protocol Corporate Standard (GHG Protocol 2004). The 15 categories in the Scope 3 Standard are shown in Table 1.1. Those included in this study are identified. Excluded categories are either assumed to be so small as to be immaterial (i.e., capital goods, business travel, and employee commuting) or are considered irrelevant to an industry-wide inventory focused on the manufacturing value chain (i.e., upstream leased assets, downstream leased assets, and franchises not associated with the value chain).

Table 1.1. GHG Protocol Scope 3 Categories Included in this Study

GHG Protocol Value Chain (Scope 3) Standard Category	Included in this Study?
1. Purchased goods and services	Included
2. Capital goods	Not included
3. Fuel- and energy-related activities (not in Scope 1 or 2)	Included
4. Upstream transportation and distribution	Included
5. Waste generated in operations	Included
6. Business travel	Not included
7. Employee commuting	Not included
8. Upstream leased assets	Included if part of the value chain
9. Downstream transportation and distribution	Included
10. Processing of sold products	Included
11. Use of sold products	Included

(Continued on next page.)

Table 1.1. Continued

GHG Protocol Value Chain (Scope 3) Standard Category	Included in this Study?
12. End-of-life treatment of sold products	Included
13. Downstream leased assets	Included if part of the value chain
14. Franchises	Included if part of the value chain
15. Investments	Included if part of the value chain

The structure of this study reflects, in large part, the way available data and methods are organized. The elements of this study and the approximate alignment with the 15 categories (Table 1.1) are shown in Table 1.2. The alignment is not exact, however. Details of the calculations for each element are contained in this report.

Net transfers of biogenic carbon to the atmosphere are estimated by "production accounting." Using this approach, the net transfers of biogenic carbon are calculated as the net changes in stocks of stored carbon attributable to wood harvested in the US (US EPA 2022a). Mathematically, this provides the same estimate as adding the flows of biogenic carbon to and from the atmosphere along the value chain associated with wood harvested in the US.

Most of this report is focused on annual inventories for the years 1990, 2005, and 2020. Annual inventories capture the emissions and removals in a single year (the year of the inventory) attributable to production in that year and all previous years. This report, however, also examines the long-term impacts associated with products manufactured in a single year (i.e., manufactured in 1990, 2005, and 2020). The two approaches provide fundamentally different types of information and should not be compared.

Table 1.2. Elements in GHG Profile and Approximate Comparison to GHG Protocol Categories

Element Description in this Study	GHG Protocol Category
Changes in forest ecosystem carbon stocks	Not included in Corporate or Scope 3 Standards
Changes in forest product carbon stocks	Not included in Corporate or Scope 3 Standards
Scope 1 Fuel Combustion	Corporate Standard and Scope 3 category 10 (processing of sold products)
Scope 2	Corporate Standard and Scope 3 category 10 (processing of sold products)
Waste management	Scope 3 category 5 (waste generated in operations)
Transport	Scope 3 category 4 (upstream transportation and distribution) Scope 3 category 9 (downstream transportation and distribution)
Upstream – Purchased electricity	Scope 3 category 1 (purchased goods and services)
Upstream – Nonfiber input production	Scope 3 category 1 (purchased goods and services)
Upstream – Fossil fuel production & transport	Scope 3 category 3 (fuel- and energy-related activities [not included in Scope 1 or Scope 2])
Product use	Scope 3 category 11 (use of sold products)
Wood supply production	Scope 3 category 1 (purchased goods and services)
Product end-of-life	Scope 3 category 12 (end-of-life treatment of sold products)

2.0 FOREST ECOSYSTEM CARBON

The US Forest Service develops periodic estimates of carbon stocks and fluxes in forest ecosystems. The US Environmental Protection Agency (US EPA) publishes these in its annual report on GHG emissions and sinks. Data for 1990 to 2020, taken from US EPA's Inventory of Greenhouse Gases and Sinks: 1990–2022 and the annexes to that inventory (US EPA 2024a; US EPA 2024b), are shown in Figure 2.1. The estimated net growth in carbon stocks in US forests in 2020 (208.6 million metric tons per year) is equivalent to removing 765 million metric tons CO_2 from the atmosphere annually (converting carbon to CO_2 equivalents (CO_2eq.) by multiplying by 44/12). These data include forests producing industrial roundwood and those that are not[1].

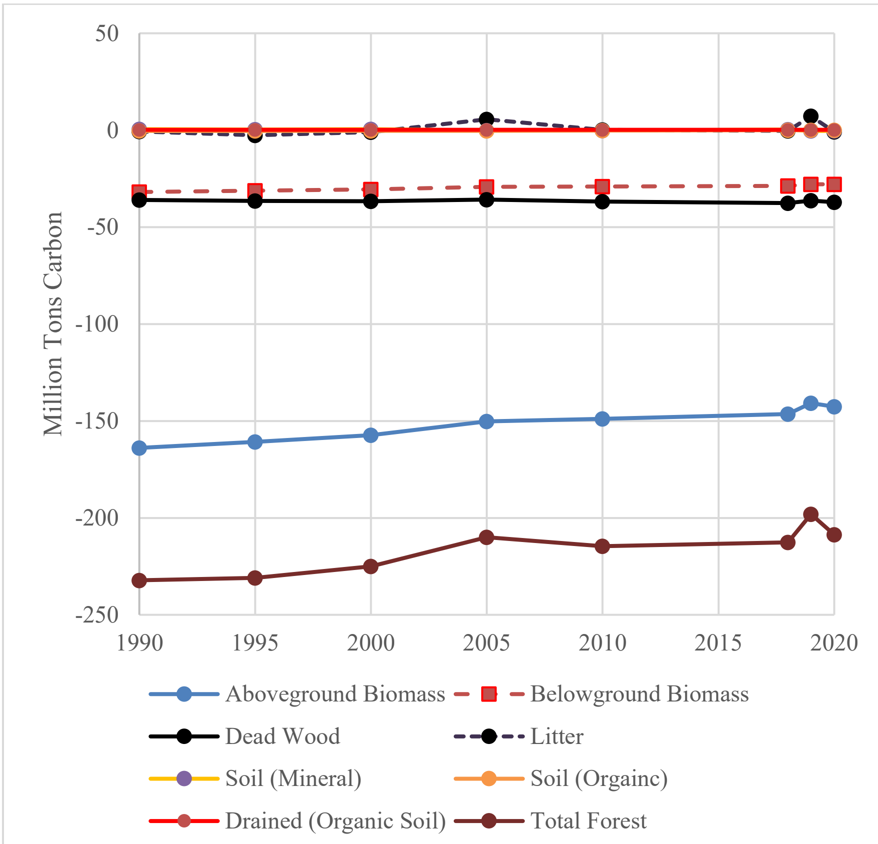


Figure 2.1. Annual Fluxes in Forest Ecosystem Carbon in the US: 1990–2020 [Note: Negative numbers = net removals of carbon from the atmosphere]

For the previous GHG and carbon profile (NCASI 2008; Heath et al. 2010), the US Forest Service examined this question at a level of detail not possible with publicly available data. The focus was on forests that provide most of the wood for the forest products industry, namely private timberland under industrial and nonindustrial ownership. In 2016, private timberlands produced 89% of wood

[1] These data are for forest land remaining in forest. The forest carbon analysis in this report does not include the effects of land-use change. These effects are small. Since 1990, the gains and losses of forest land in the US have resulted in a net loss of 5 to 7 million metric tons of forest ecosystem carbon per year (calculated from data in US EPA 2022a).

produced on all timberlands in the US (Table 35 in Oswalt et al. 2019). The previous analysis of carbon stocks on private timberland found that:

> "…because wood production is a more important management objective on industrial timberland than on most other types of private timberland, carbon stocks are not accruing on industrial timberland in the same way they are accruing on private timberland generally. Timberlands in total gained carbon between 2000 and 2005 (354 Tg CO_2eq./yr or about 0.7%/yr … while industry-owned lands lost a small amount 11 Tg CO_2eq./yr or about 0.1%/yr …
>
> A complete assessment of the effects of the industry's activities on forest ecosystem carbon… should encompass not only carbon stocks on industry-owned land, but also the indirect effects on carbon stocks on other land. Unfortunately, it is not possible to identify and precisely quantify these indirect effects. Nonetheless, given the large increases in carbon stocks on nonindustrial private timberland, it appears reasonable to assume that over the period included in this analysis the industry's activities did not cause decreases in forest ecosystem carbon stocks or increases in atmospheric CO_2. For purposes of this study, therefore, it was assumed that net emissions attributable to the forest products industry's activities in the forest were zero for the period covered by the analysis." (NCASI 2008, 4)

Since 2005, as shown in Figure 2.2, domestic harvesting declined initially then rebounded so that by 2020, it was nearly the same level as 2005 but still below 1990 levels.

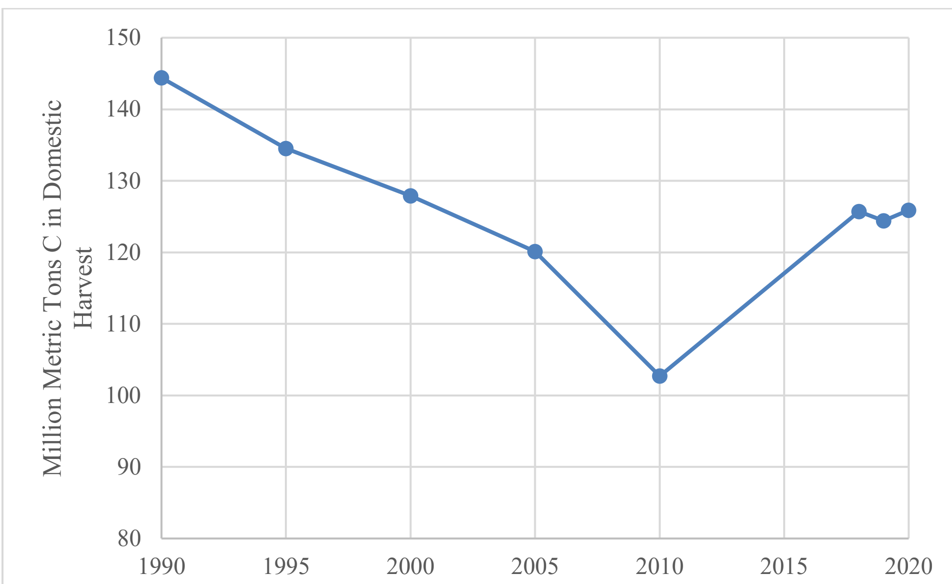


Figure 2.2. US Domestic Harvest – 1990 to 2020 [*Source*: US EPA (2024b)]

The effects of harvesting on forests can be examined using net forest stock change (in timber volume or in carbon terms). Data from Table 36 in Oswalt et al. (2019) show that, in 2016, stocks in seven of the eight regions reported by the Resources Planning Act Assessment were increasing, with losses in the Intermountain region due to high levels of mortality from fire and insect damage. The regions with the highest increases in forest stock were the areas with highest harvest levels (which leads to increased areas of younger, rapidly growing forests), as shown in Figure 2.3.

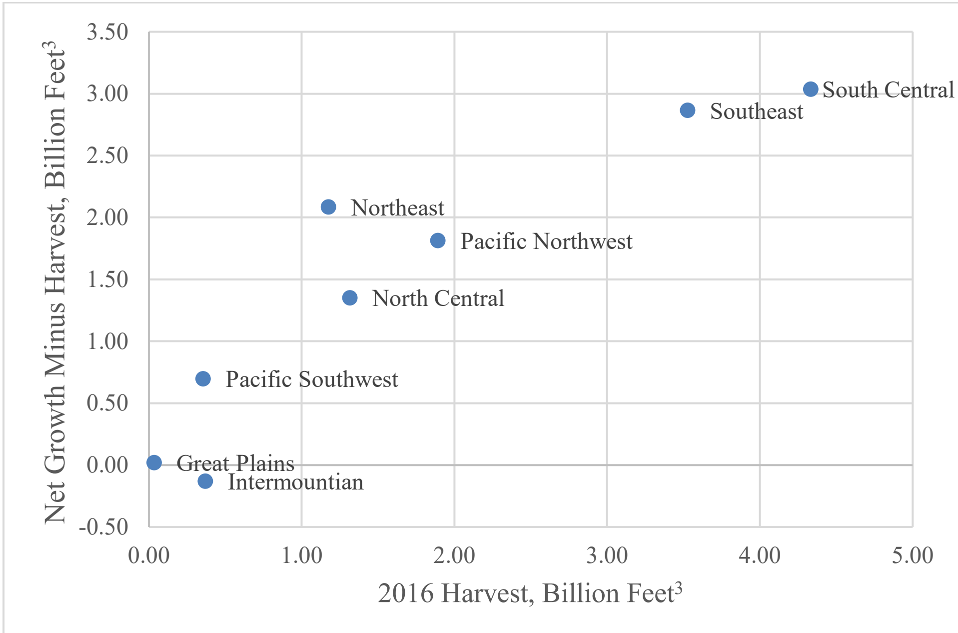


Figure 2.3. Net Forest Stock Change in Timberlands and Harvests for Eight US Regions

The harvesting and forest data available since 2005, therefore, provide no reason to suspect that carbon stocks are declining on the land that produces wood for the forest products industry. As a result, for purposes of this study, as in the earlier study, it is concluded that net emissions of biogenic carbon attributable to the forest products industry's activities in the forest can be conservatively assumed to be zero for the period covered by the analysis.

3.0 CARBON STORED IN FOREST PRODUCTS

Much of the carbon removed from forests by industry is transferred into products. These products store carbon during use and sometimes continue to store carbon in landfills. Small amounts of carbon are also stored in landfills receiving forest products manufacturing wastes, but the amounts of carbon are small and are not included in this report. Emissions of methane from industry landfills, however, are included.

US EPA publishes estimates of annual changes in stocks of carbon in forest products in its annual inventory of US GHG emissions and sinks. These estimates are developed for US EPA by the US Forest Service. For the 2008 profile, NCASI partnered with the US Forest Service to develop disaggregated estimates that allowed the role of paper products and wood products to be examined separately. In this update, however, the estimates published by US EPA (US EPA 2024a; US EPA 2024b) are being used directly. The estimates in the 2008 profile have been updated to this new approach, allowing more appropriate, albeit non-disaggregated, comparisons to earlier years.

In addition, this update includes estimates of the projected long-term benefits of carbon stored in products manufactured in 1990, 2005, and 2020.

3.1 Carbon Stored in Products in Use

After manufacturing, forest products remain in use for periods varying from days to centuries. Carbon remains stored during this time, delaying its return to the atmosphere. If carbon is added to the pool of products in use faster than it is being removed by the retirement of older products, stocks of carbon in this pool grow.

Estimates of annual changes in carbon held in forest products in use are reported annually in the US inventory of GHGs and sinks. The values in Table 3.1 are from that report. The estimates and their derivation are detailed in US EPA 2024b, Annex 3, Section 3.13.

US EPA's analysis shows that annual changes in stocks of carbon in use are small compared to the amounts already in use. The amount leaving the pool of products in use is relatively constant compared to the year-to-year change in inputs to this pool (i.e., new production). As a result, changes in industry production have a large impact on the annual changes in carbon stocks in use.

Table 3.1. Stocks of Carbon in US Forest Products in Use [*Source*: US EPA (2024b) Table Annex A A-201; US EPA (2024a) Table 6-10]

Year	Net C Additions, Million Metric Tons C/yr	Total C Stocks, Million Metric Tons C
1990	14.9	1,249
2005	11.6	1,447
2020	8.8	1,530

The estimates shown in Table 3.1 were developed for US EPA by the US Forest Service using the "production approach." Under the production approach, any carbon stock change attributable to wood harvested in the US is included in the US's results. This is true even if the wood or forest products are exported and used in other countries. Conversely, changes in domestic stocks of carbon are not included if they are associated with wood produced elsewhere.

The primary reason for the decline in annual additions between 1990 and 2005 was decreasing domestic harvest. The correlation between domestic harvest and annual changes in stocks of carbon in products in use is evident in Figure 3.1.

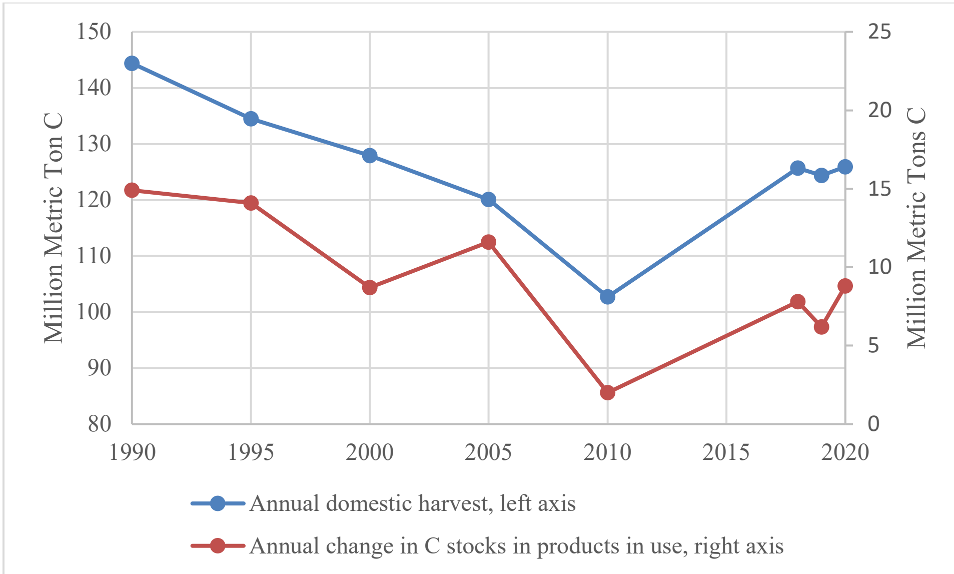


Figure 3.1. Changes in Harvested Wood and Product-in-Use Carbon

Between 1990 and 2006 the decline in domestic wood production was gradual, but the 2007–2009 recession caused a dramatic downturn in harvesting of wood, which had the expected impact on changes in stocks of carbon in products in use. Wood production and in-use carbon stocks subsequently increased.

The years examined in the 2008 profile were 1990 and 2004/2005. For this update, 2020 is added and, where possible, 2005 is used as a historical comparison (instead of 2004/2005). The stocks of carbon in the products-in-use pool grew by 14.9 million metric tons C in 1990, 11.6 million metric tons C in 2005, and 8.8 million metric tons C in 2020. The growth in stocks of carbon in products in use represented removals of CO_2 from the atmosphere of 54.6, 42.5, and 32.3 million metric tons of CO_2 in 1990, 2005, and 2020, respectively.

The stock change values for 1990 and 2005 in this update differ from those reported in NCASI 2008. The primary reason is that, in recent inventories, US EPA updated its past estimates based on updated analysis by the US Forest Service. US EPA lowered the 1990 increase in stocks of in-use carbon to 14.9 from 17.7 million metric tons C. Likewise, the estimated increase for 2005 has been reduced to 11.6 from 12.1 in earlier US EPA inventory reports.

3.2 Carbon Stored in Products in Landfills

Forest products are recycled or discarded after use, with some going to landfills. Landfills gain carbon as new discards are added and lose carbon as products degrade into a mixture of methane and CO_2. The annual balance between gains and losses determines whether stocks of biogenic carbon in landfills increase or decrease. US EPA's annual inventories of GHG emissions and sinks include estimates of landfill forest product carbon stock changes (US EPA 2024a; US EPA 2024b). For this profile, we use those estimates and update the 2008 profile (NCASI 2008; Heath et al. 2010) values to allow consistent comparisons with previous periods. As for products in use, these calculations are developed by the US Forest Service using the production approach. Under the production approach, any carbon stock change in landfills attributable to wood harvested in the US is included in the US's results. This is true even if the landfills are in other countries. The changes in landfill carbon stocks attributable to forest products produced in the US are shown in Table 3.2.

Table 3.2. Stocks and Stock Changes of Carbon in Forest Products in US Landfills
[*Source*: US EPA (2024b)]

Year	Net C Additions, Million Metric Tons C/yr	Total C Stocks, Million Metric Tons C
1990	18.8	646
2005	17.3	906
2020	17.6	1,165

As is the case for carbon in products in use, carbon stocks in landfills are much larger than the annual net changes. The net changes, however, are much less variable than changes in stocks of carbon in products in use. This is primarily because inputs to landfill carbon stocks are far less variable than inputs to the pool of products in use[2]. Net additions to landfill carbon stocks in 1990, 2005, and 2020 were 18.8, 17.3, and 17.6 million metric tons C per year, respectively. These additions correspond to net removals of CO_2 from the atmosphere of 68.9, 63.4, and 64.5 million metric tons CO_2 per year in 1990, 2005, and 2020, respectively.

3.3 Total Carbon Storage in Forest Products

In Figure 3.2, the contributions of wood products and paper to carbon stock changes are shown separately. The data series stops in 2013 because after that date, US EPA stopped showing the separate contributions of wood and paper. Annual changes in carbon stocks in products in use are highly dependent on changes in production rates for both wood and paper products. Indeed, during economic slowdowns, stocks of carbon in products in use can decrease (i.e., stocks change less than zero). Annual increases in stocks of carbon in wood products in use are consistently larger than those for paper due to the short time in use for paper products. Stocks of carbon in products in landfills increase annually over the entire record. This is because carbon in products continues to be added to landfills at a rate faster than decomposition removes carbon from landfills.

Total annual changes in carbon stocks attributable to forest products are shown in Figure 3.3. While in the early 1990s net additions to products in use were comparable to additions to landfills, carbon stock changes due to products in use decreased over time. The total change in carbon stocks attributable to forest products has also decreased over time. The total net additions in units of carbon and CO_2 are shown in Tables 3.3 and 3.4. These additions were 33.8, 28.9, and 26.4 million metric tons C per year in 1990, 2005, and 2020, respectively. These correspond to total net removals from the atmosphere of 124, 106, and 96.8 million metric tons CO_2 per year in 1990, 2005, and 2020, respectively. The 1990 and 2005 estimates are very close to those in NCASI (2008) and Heath et al. (2010). This is not surprising as both are based on data and analysis from the US Forest Service.

[2] From 1990 to 2020, annual changes in landfill carbon stocks ranged from 6% less to 9% greater than the mean change. The range for changes in stocks of carbon in products in use was much larger: from 19% below the mean to 76% above the mean.

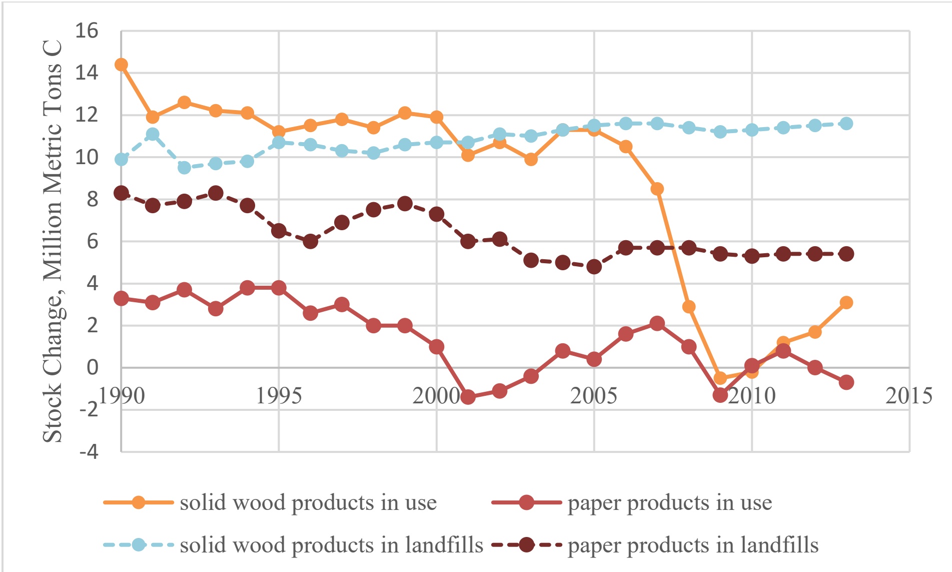


Figure 3.2. Contributions of US Wood and Paper Products to Changes in Stocks of Carbon in Products in Use and in Landfills [*Source*: US EPA (2015)]

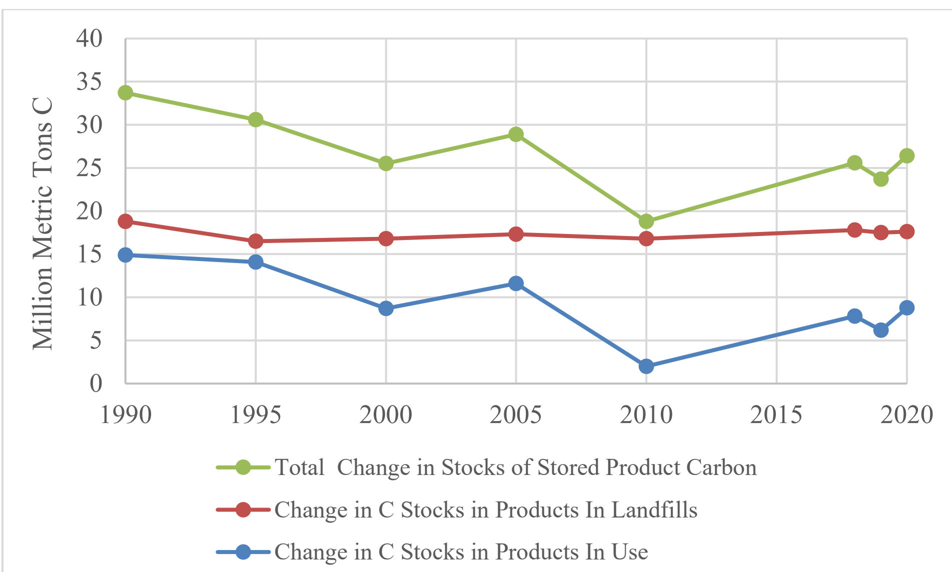


Figure 3.3. Net Additions to Carbon Stocks in Forest Products Produced from Wood Harvested in the US [*Source*: US EPA (2024b)]

Table 3.3. Net Additions to US Carbon Stocks in 1990, 2005, and 2020 [*Source*: US EPA (2024b)]

Year	Net C Additions, Million Metric Tons C/yr			Fraction of 1990 Total	Fraction of 2005 Total
	In Use	In Landfills	Total		
1990	14.9	18.8	33.8	1.00	
2005	11.6	17.3	28.9	0.86	1.00
2020	8.8	17.6	26.4	0.78	0.91

Table 3.4. Net Emissions Attributable to Changes in Carbon Stocks in US Forest Products [*Source*: US EPA (2024b)]

Year	Million Metric Tons CO_2eq.[a]		
	Attributable to Products in Use	Attributable to Products in Landfills	Total
1990	−54.6	−68.9	−123.9
2005	−42.5	−63.4	−106.0
2020	−32.3	−64.5	−96.8

[a] Negative sign indicates a net removal of CO_2 from the atmosphere.

3.4 Accounting for Long-Term Storage in a Single Year's Production

The aforementioned storage calculations were developed using annual inventory accounting. Annual changes in carbon stocks attributable to current and past year's production are reported in the inventory for the year in which the stock changes occur. The benefits associated with carbon storage in products can, however, also be characterized by calculating the long-term impacts of carbon in a single year's production. There are several approaches for presenting long-term benefits of stored carbon. Two approaches are used here. One approach is simply to report the amounts of carbon remaining stored for 100 years following the product's production. The other is based on the radiative forcing associated with losses of product carbon over 100 years. The second approach produces a metric referred to, in this report, as "effective long-term storage."

To calculate effective long-term storage, the cumulative radiative forcing associated with losses of stored carbon from the manufactured product is calculated over 100 years following manufacturing. This "positive" 100-year radiative forcing associated with lost carbon stocks is added to the "negative" radiative forcing associated with the addition of the original product to stocks of stored carbon (see Fearnside et al. (2000) for a similar approach). The results represent the net radiative forcing impact of additions to, and losses from, the stocks of stored carbon over 100 years, expressed in units of stored carbon or CO_2eq.

The starting point for the calculations is information on the amount of carbon that remains stored in forest products in use over time. Smith et al. (2005) provides data, shown in Figure 3.4, on the fraction of original product carbon residing in the in-use pool in the 100 years following manufacturing.

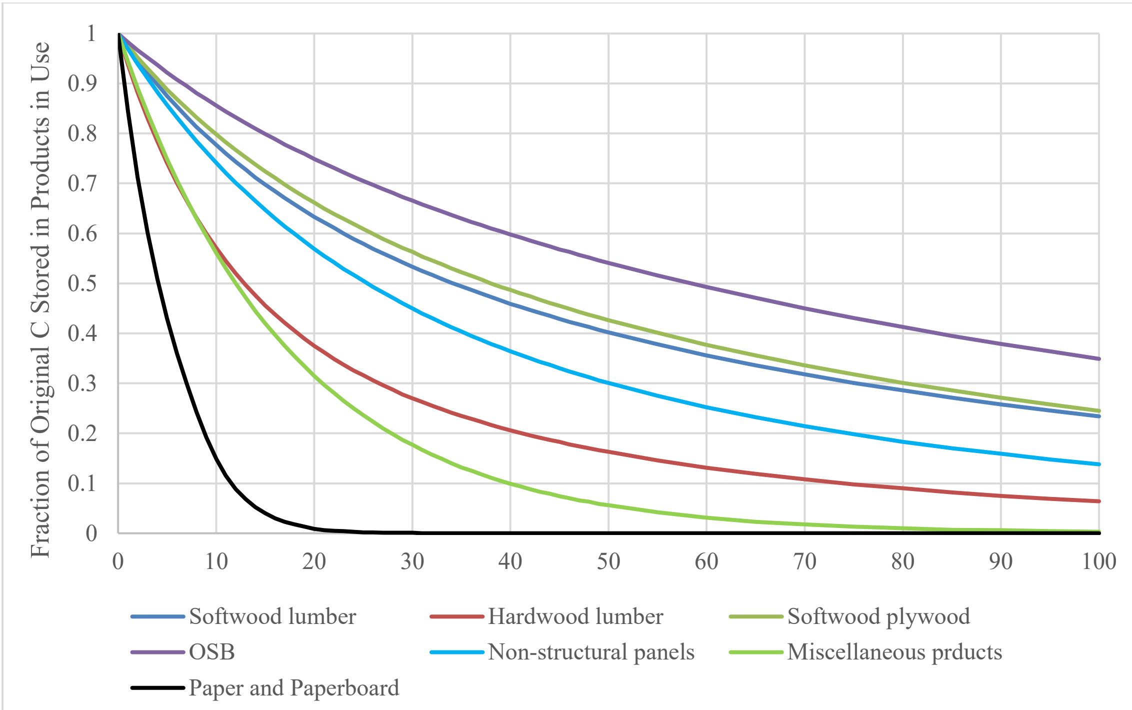


Figure 3.4. Carbon Stored in the Pool of US Products in Use [*Source*: Smith et al. (2005)]

In this report, time-in-use data from Smith et al. (2005) have been applied to information on annual production (AF&PA 2020; 2021a; Howard and Liang 2019) to determine annual additions to, and discards from, the in-use pool of products[3]. This allows calculation of annual changes in stocks of carbon in products in use manufactured in the year of interest (i.e., 1990, 2005, or 2020). Smith et al. (2005) data are not used, however, to calculate changes in stocks of landfilled carbon. This is because the Smith et al. curves are for a single point in time. While it is reasonable to assume that product-in-use curves, like those in Smith et al. (2005), will be valid over time, it is not reasonable to assume this for discarded products. This is primarily because the fraction of discards being landfilled has changed significantly over time and is expected to continue changing. These changes need to be accounted for in projections of the fate of landfilled carbon.

In this study, discards calculated using the approach described previously are allocated to recycling, combustion, and landfilling using information from US EPA (US EPA 2020e)[4]. The fate of carbon in landfilled discards is calculated using a first order decay model commonly used for such purposes (e.g., see US EPA (2022b)). From this, annual changes in stocks of landfilled carbon are calculated.

To determine effective long-term storage, annual losses of original product carbon are calculated by summing the annual changes in remaining carbon stocks in use and in landfills. The cumulative radiative forcing over 100 years is calculated for each year's losses of stored carbon. The annual

[3] Smith et al. (2005) time-in-use data for paper and paperboard products have been modified as described in Appendix A.
[4] US EPA (2020e) data do not account for paper and paperboard associated with imported products (e.g., packaging). This may affect the allocation calculations. This issue is discussed further in Section 10 of this report, in the material on emissions associated with forest products end-of-life.

values are summed to obtain the cumulative radiative forcing associated with all losses of stored carbon over 100 years. The cumulative 100-year forcing associated with losses of stored carbon is then converted to carbon equivalents and netted against the carbon added to stocks when the original product was placed into use. The result is the effective long-term carbon storage associated with that year's production. The calculations are detailed in Appendix A.

The results of these calculations are shown in Tables 3.5, 3.6, 3.7, and 3.8. Table 3.5 shows the effective long-term storage based on radiative forcing impacts, while Tables 3.6, 3.7, and 3.8 show the metric tons of original product carbon remaining stored 100 years after product manufacture. The effective long-term storage benefit (in carbon equivalents) for paper products and wood products manufactured in 2020 was equal to 16% and 75%, respectively, of the carbon in the products. Effective long-term storage values are larger than the corresponding amounts of carbon stored for 100 years because the losses of stored carbon occur gradually. Losses of stored carbon in the 99th year, for instance, cause very little radiative forcing during the 100-year post-manufacturing period because they are in the atmosphere for only 1 year of this period.

Table 3.5. Total Effective Long-Term Carbon Storage, Equivalent Net 100-Year Radiative Forcing

Year	Million Metric Tons C in Net Radiative Forcing Equivalents		
	Paper	Wood	Total
1990	13.3	31.9	45.1
2005	10.1	32.5	42.6
2020	5.9	29.0	34.9

Table 3.6. Total Carbon Remaining Stored in Products 100 Years Following Manufacturing

Year	Million Metric Tons C		
	Paper	Wood	Total
1990	11.1	28.6	39.8
2005	8.0	29.0	37.1
2020	4.3	25.9	30.3

Table 3.7. Carbon Remaining Stored in Products in Use 100 Years Following Manufacturing

Year	Million Metric Tons C		
	Paper	Wood	Total
1990	0.0	6.7	6.7
2005	0.0	7.7	7.7
2020	0.0	6.4	6.4

Table 3.8. Carbon Remaining Stored in Products in Landfills 100 Years Following Manufacturing

Year	Million Metric Tons C		
	Paper	Wood	Total
1990	11.1	22.0	33.1
2005	8.0	21.4	29.4
2020	4.3	19.6	23.9

The 100-year carbon storage benefit for 2020 production is lower than for 1990 and 2005 production. This is due to two factors. First, stored carbon in products in use declined due to lower production in 2020[5]. This also impacted carbon storage in landfills. Landfill storage was further impacted by the declining fraction of discards going to landfills. Products made more recently, therefore, will add less to landfill carbon stocks than those made in the past. The details on this phenomenon are examined in Appendix A. The sizes of the pools of stored carbon over time for products manufactured in 2020 are shown in Figure 3.5.

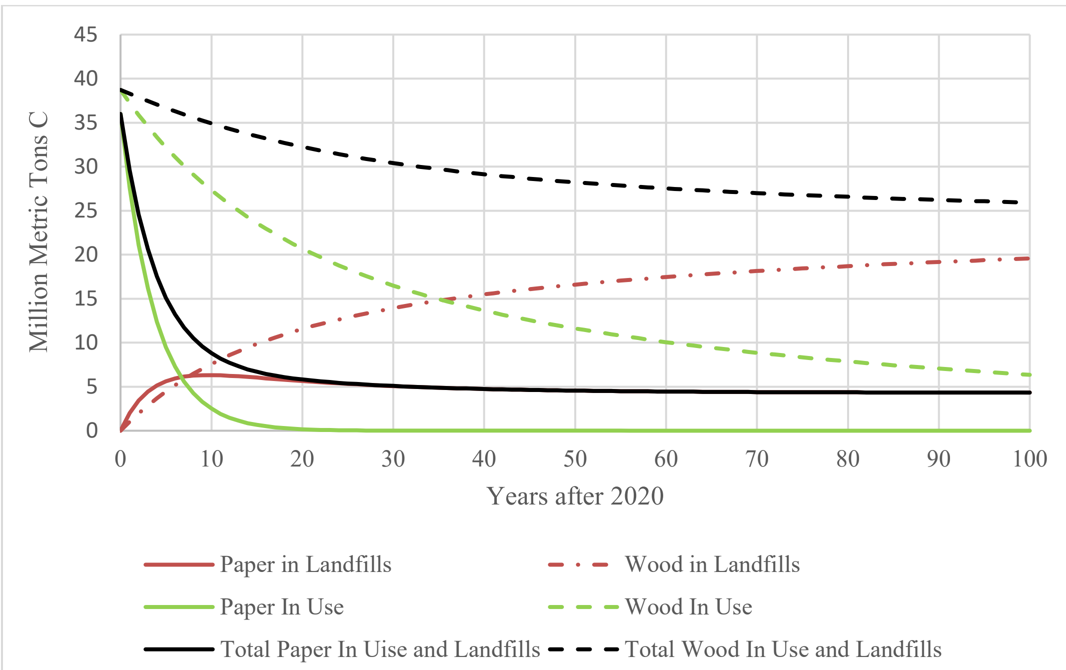


Figure 3.5. Distribution of Carbon in Use and in Landfills for US Forest Products Manufactured in 2020

4.0 ENERGY- AND PROCESS-RELATED EMISSIONS FROM PRIMARY PRODUCTION FACILITIES

The forest products industry both purchases and self-generates fuels, electricity, and steam to power its production processes. In this profile, emissions from the combustion of fuels at manufacturing and converting operations are classified as Scope 1 emissions. The emissions released from facilities producing electricity or steam purchased by these operations are accounted

[5] 2020 data were available for paper products, but the most recent values available for wood products were for 2017.

for as Scope 2 emissions. Emissions are also released in the production and transport of fuels used by the forest products industry and by its suppliers of electricity and steam. These are Scope 3 emissions. In the previous profile (NCASI 2008; Heath et al. 2010) only Scope 1 and 2 energy-related emissions were considered.

In this section, Scope 1, Scope 2, and Scope 3 emissions associated with energy use at primary production operations are calculated. Primary production facilities include pulp, paper, and paperboard mills; lumber mills; wood panel plants; and other wood products plants. Not included are facilities that convert primary products into final products. Examples of converting operations include printing plants, box plants, furniture manufacturers, and construction activities. Converting-related emissions are addressed elsewhere in this report.

4.1 Scope 1 Direct Fuel Combustion-Related GHG Emissions

Historical energy and production data for the US forest products industry have been used to calculate fuel-related GHG trends for the industry since 1990 on both an absolute basis (mass of GHG emissions) and an intensity basis (mass of GHG emissions per unit of production). Direct emissions are reported in units of CO_2eq. and calculated as the sum of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) from on-site fossil fuel sources and CH_4 and N_2O emissions from on-site biomass fuel sources. Global warming potentials (GWPs) of 25 and 298 are used for CH_4 and N_2O, respectively. Details on emission factor sources and GWPs used in calculations are provided in Appendix B.

4.1.1 Pulp and Paper Mill Emissions

4.1.1.1 Data Sources

Available government and industry data sets were synthesized for GHG emission calculations. Details regarding the classification of the US pulp and paper sector sources are provided in Appendix C.

US Energy Information Administration Manufacturing Energy Consumption Survey

The US Energy Information Administration (EIA) collects information on manufacturing energy use every 4 years as part of its manufacturing energy consumption survey (MECS) effort (EIA 2023a). While the first EIA MECS was conducted in 1985, detailed energy consumption tables are only available for 1991, 1994, 1998, 2002, 2006, 2010, 2014, and 2018. In terms of industry coverage, the 2018 EIA MECS sample of approximately 15,000 establishments was drawn from a nationally representative sample representing 97 to 98% of the manufacturing payroll. EIA MECS data are considered representative of the entire US pulp and paper industry.

American Forest and Paper Association/American Paper Institute

The American Forest and Paper Association (AF&PA) and its predecessor, the American Paper Institute (API), have regularly collected industry fuel, energy, and production information. The AF&PA/API fuel, energy, and production data set from 1972 to 2018 was used in this analysis. For the years 2000 to 2020, the fuel and energy data sets represent only AF&PA members. Therefore, these results were scaled based on a production scaling factor of total US paper, paperboard, and market pulp production from sector-wide production amounts (AF&PA 2020) divided by AF&PA member paper, paperboard, and market pulp production for each data year. Throughout this report, AF&PA and API data are referred to as “AF&PA/API” in tables and figures.

US EPA Greenhouse Gas Reporting Program

The Greenhouse Gas Reporting Program (GHGRP) (US EPA 2023a) managed by the US EPA requires reporting of GHG data and other relevant information from large GHG emission sources, fuel and industrial gas suppliers, and CO_2 injection sites in the US. Approximately 8,000 facilities are required to report emissions annually, and the reported data are made available to the public in October of each year. Data reported by the US pulp and paper industry are available annually from 2010 to 2021 (US EPA 2023a). Generally, the GHGRP requires reporting by facilities emitting more than 25,000 mt CO_2eq. per year. Approximately 85 to 90% of total GHG emissions in the US are represented by the GHGRP data sets (US EPA 2023b). GHGRP coverage for the pulp and paper sector is estimated to represent essentially 100% of industry emissions from chemical pulp and paper manufacturing and 84 to 95% of emissions from other paper and paperboard sectors.

4.1.1.2 Greenhouse Gase Emissions on an Absolute Basis

The fuel-related direct emissions from the pulp and paper sector in 1990, 2005, and 2020 are shown in Table 4.1. The trends in these emissions are shown in Figure 4.1. Direct GHG emissions peaked in 1994 for the US pulp and paper industry before decreasing, driven by switching to less GHG-intense fuels, energy efficiency improvements, and production reductions within the industry over time.

Since 1990, direct GHG emissions for the US paper, paperboard, and market pulp sector have been reduced by 51%, a result consistent with EIA MECS. Since 2005, emissions have been reduced by 40%.

Table 4.1. Direct Fuel Combustion-Related Emissions from the US Paper, Paperboard, and Market Pulp Sector

	Million Metric Tons CO_2 eq.		Based on AF&PA/API Data	
	AF&PA/API	EPA GHGRP	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990	66.9	--	1.00	
2005	55.2	--	0.83	1.00
2020	33.0	34.9	0.49	0.60

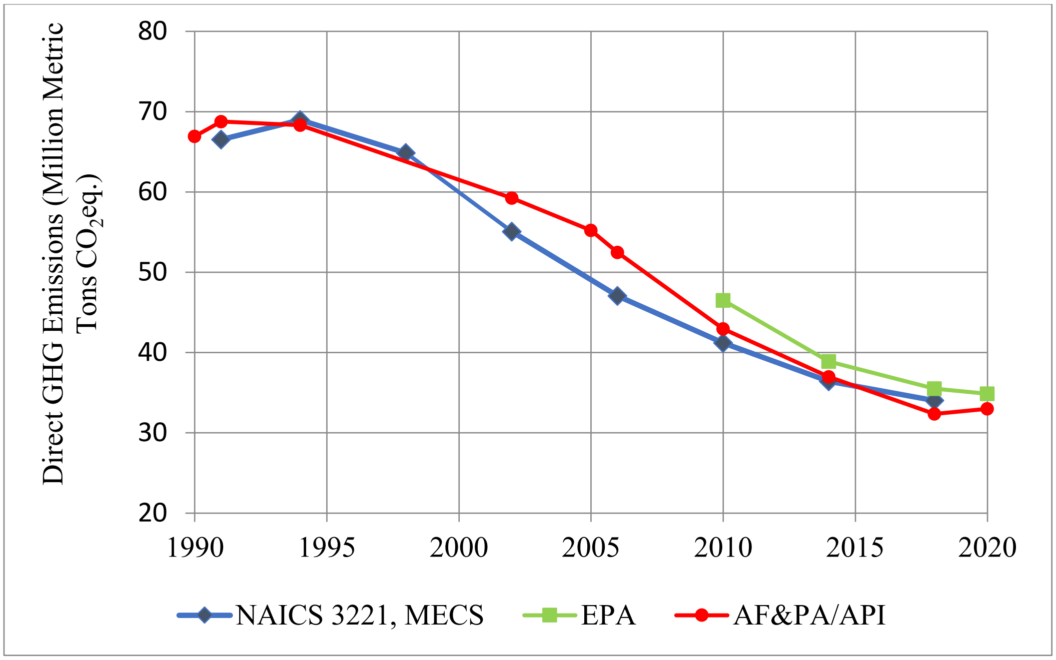


Figure 4.1. Direct GHG Emissions Over Time for the US Pulp and Paper Industry

4.1.1.3 Greenhouse Gase Emissions on an Intensity Basis

Figure 4.2 shows GHG reductions for the US pulp and paper sector on an intensity basis calculated from EIA MECS and AF&PA/API data. Since 1990, direct GHG emission intensities for the US pulp and paper sector have been reduced by 48% (0.83 metric tons CO_2eq. per metric ton production in 1990 vs. 0.43 in 2020).

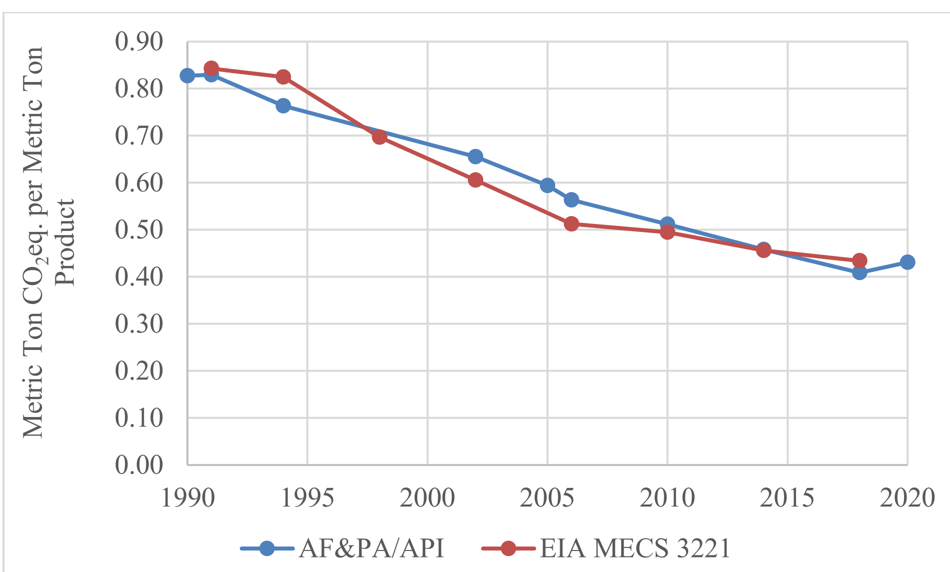


Figure 4.2. Direct US Pulp and Paper Sector GHG Emission Intensity Changes Over Time

4.1.1.4 Discussion

Increased Utilization of Biomass

The US pulp and paper industry derives most of its energy needs from biomass fuels. Biomass as a percentage of total fenceline energy (see "Fenceline Energy" text box) increased from 56% in 1990 to 65% in 2018, based on AF&PA/API data. Likewise, EIA MECS data show biomass fuel use increasing from 53% of fenceline energy needs in 1991 to 63% in 2018. Figure 4.3 shows trends in biomass energy utilization over time for the US pulp and paper industry.

Fenceline Energy

Fenceline energy consumption: The amount of fuel and energy consumed within a facility measured at the operational or ownership boundary of the facility.

EIA MECS terms to calculate fenceline energy consumption:

Fenceline energy consumption = Residual Fuel Oil* + Distillate Fuel Oil* + Natural Gas* + Hydrocarbon Gas Liquid (excluding natural gasoline)* + Coke* + Coke and Breeze* + Other* + (Purchases** + Transfers In**)

Net fenceline energy consumption: The amount of fuel and energy consumed within a facility minus the amount of fuel and energy exported from the facility, measured at the operational or ownership boundary of the facility.

EIA MECS terms to calculate net fenceline energy consumption:

Net fenceline energy consumption = Net Electricity* + Residual Fuel Oil* + Distillate Fuel Oil* + Natural Gas* + Hydrocarbon Gas Liquid (excluding natural gasoline)* + Coke* + Coke and Breeze* + Other*

Or equivalently:

Net fenceline energy consumption = Total*

Note: "Other" from Table 3.2 in EIA MECS (2023a) includes all biomass fuels such as spent liquor solids and biomass residuals. "Other" also includes *net* steam (the sum of purchases and net transfers). Published EIA MECS data do not provide equivalent information on steam as they do with electricity, so the components of steam (purchased steam, transfers in of steam, and sales and transfer out of steam) cannot be parsed in the same way as electricity. Including the contribution of sales and transfer of steam in fenceline energy consumption should make less than a 0.25% difference compared to only considering purchased steam, based upon NCASI and AF&PA survey data.

*EIA MECS (2023a) Table 3.2 Fuel Consumption 2018
https://www.eia.gov/consumption/manufacturing/data/2018/pdf/Table3_2.pdf
**EIA MECS (2023a) Table 11.1 Electricity: Components of Net Demand

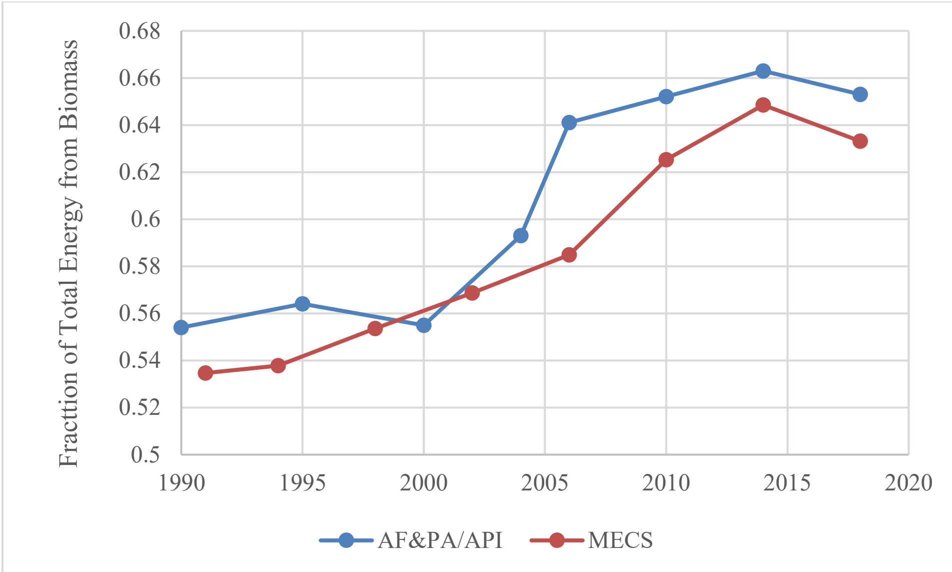


Figure 4.3. Biomass Energy as Percent of Total Fenceline Energy Use

Fuel Switching to Less Greenhouse Gas–Intense Fuels

The on-site energy mix for the US pulp and paper industry over time, based on EIA MECS data, is shown in Figure 4.4. In 1991, coal and other fossil fuels (primarily residual fuel oil) represented 13% and 7% of the on-site energy mix, respectively. In 2018, the coal contribution decreased to 3%, and the other fossil fuels' contribution decreased to 1%. Between 1991 and 2018, the natural gas energy contribution increased from 20% to 26%, and the biomass energy contribution increased from 53% to 63%. Switching from more GHG-intense fuels, such as coal and residual fuel oil, to less GHG-intense fuels, such as natural gas and biomass, has contributed to GHG reductions on both intensity and absolute bases for the US pulp and paper industry.

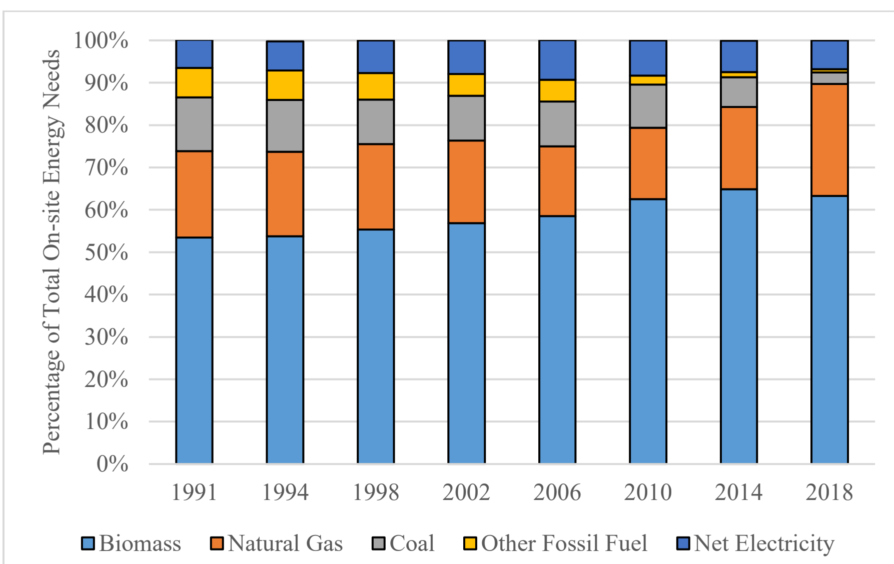


Figure 4.4. On-site Energy Mix Over Time for the US Pulp and Paper Industry Based on EIA MECS Data

Energy Intensity Improvements

The overall energy intensity of the US pulp and paper sector has decreased over time as mills have implemented energy efficiency improvements and installed more energy-efficient equipment. There have also been structural changes within the US pulp and paper sector. For instance, printing and writing paper production, which in general involves more energy-intense production processes, has declined while packaging production, which in general is less energy intensive, has increased. Figure 4.5 shows trends in total fenceline energy intensity over time. Both AF&PA/API and EIA MECS data suggest, however, that energy intensity has been relatively constant since 2005.

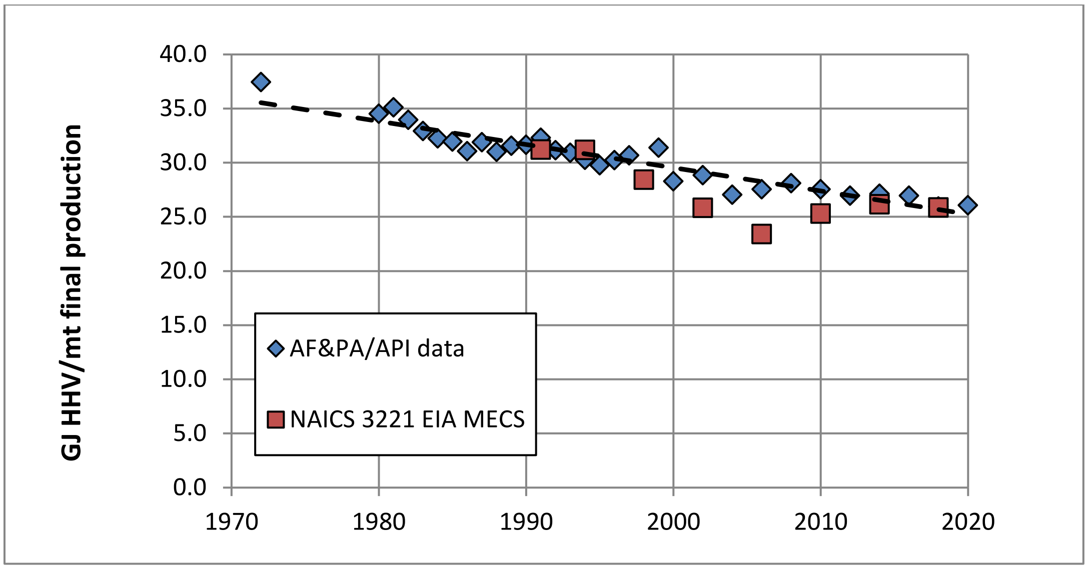


Figure 4.5. Changes in Total Fenceline Energy Intensity Over Time

Figure 4.6 shows total fenceline energy intensity over time divided between fossil fuels, net purchased electricity and steam, and biomass and other renewables. Biomass use has remained essentially constant at approximately 15 million BTU HHV/short ton of production, but the share of fossil fuel, net purchased electricity, and steam has decreased over time.

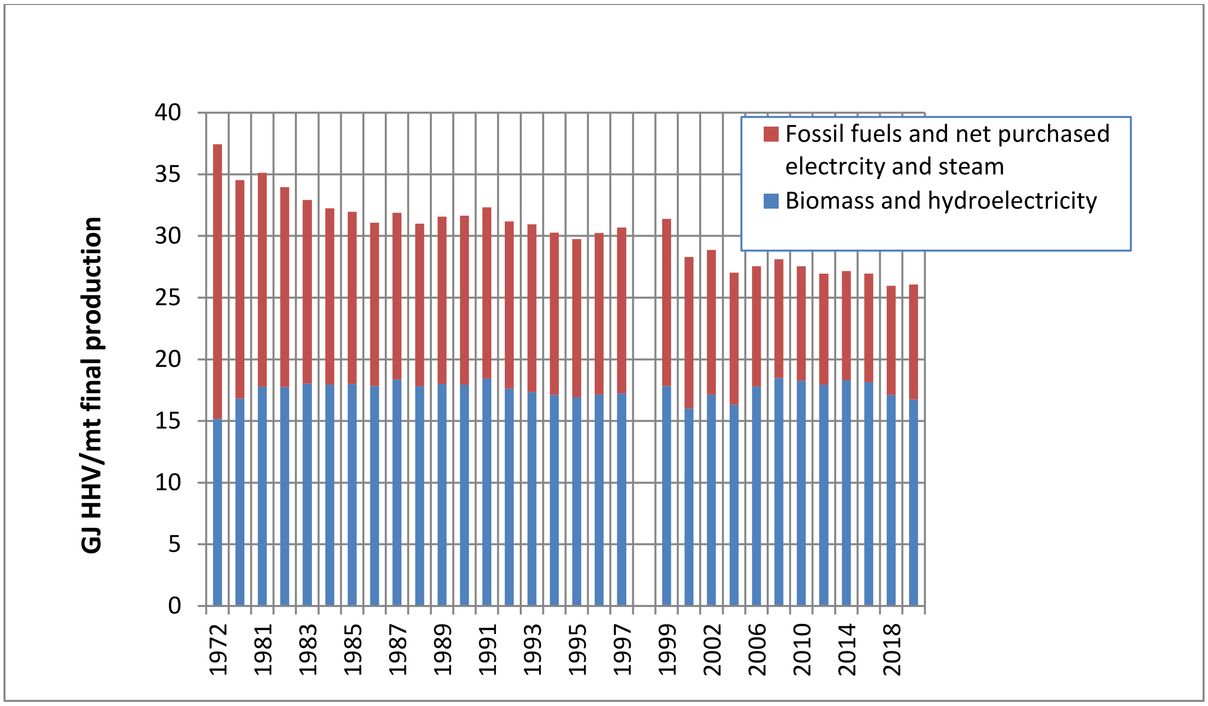


Figure 4.6. Changes in Total Fenceline Energy Intensity Over Time and Changes in Biomass and Fossil Energy Share [*Source*: AF&PA/API data]

Production Reductions

Figure 4.7 shows changes in production of paper, paperboard, and market pulp for the US pulp and paper industry over time. Production peaked in approximately 1999 and has declined since. There was an approximately 25% reduction in production between 1999 and 2020. The reduction has driven decreases in absolute GHG emissions as well as indirectly influenced GHG emission intensities. Shuttered production over time in the US has tended to be associated with older, less energy-efficient facilities, and closure of these facilities has led to overall improvements in both energy intensity and GHG intensity in the industry. In addition, increased reliance on recovered fiber as a fraction of fiber supply has contributed to the decline in energy intensity[6].

[6] AF&PA data (AF&PA 2021b, Table 9a) indicate that wood pulp consumption in the US declined by approximately 23% between 2005 and 2020, while recovered fiber consumption declined by about 8%.

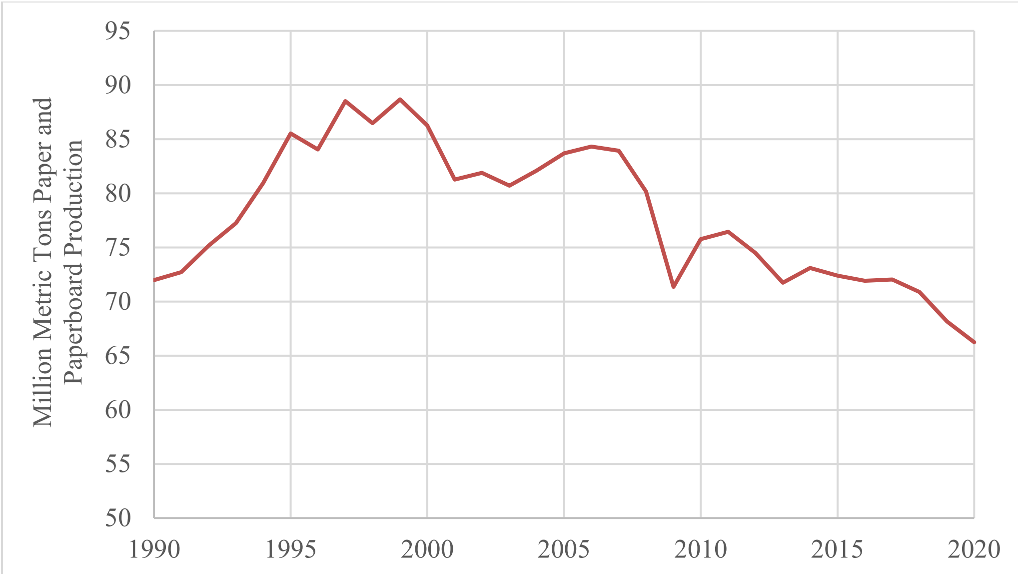


Figure 4.7. Changes in US Pulp and Paper Production Over Time [*Source*: FAOSTAT (2023)]

4.1.2 Wood Products Mills

Combustion-related emissions from wood product facilities were calculated from energy consumption data in MECS reports from 1991 through 2018 and wood products production from FAOSTAT's Forestry Production and Trade database (FAOSTAT 2023). Emissions from primary manufacturing were estimated by summing the subsector values for NAICS[7] 321 (wood products). The subsectors are NAICS 32113 (sawmills), NAICS 3212 (veneer, plywood, and engineered wood products), and NAICS 3219 (other wood products). The difference between the NAICS 321 total and the sum of the subsectors was assigned to wood products converting. In the case of wood products, the changeover to NAICS codes from Standard Industrial Classification (SIC) codes in 1997 makes it difficult to interpret the subsector data for 1991 and 1994, so only the sector totals are used for 1991 and 1994[8]. As a result, there is no residual (i.e., the difference between the sector total and the sum of subsectors) to assign to wood products converting for 1991 and 1994. Instead, these converting-related emissions are included with primary manufacturing emissions.

The US production data for wood products are shown in Figure 4.8. Output increased slowly from 1990 to 2005, dropped during the recession of 2007 to 2009, and then slowly increased from 2010 to 2018.

Emissions, calculated from EIA MECS energy data, are shown in Figure 4.9 and follow a pattern similar to that of wood products production. The on-site energy mix for the US wood products sector is given in Figure 4.10. Biomass energy sources provided approximately two-thirds of the energy needs for the wood products sector between 1991 and 2018. For this profile, the EIA MECS–based emissions for 1991, 2006, and 2018 are used to represent the years 1990, 2005, and 2020. As

[7] NAICS is the North American Industry Classification System. It replaced the Standard Industrial Classification (SIC) system in 1997.
[8] This difficulty was not encountered in the pulp and paper sector, where the alignment between NAICS and SIC codes was clearer.

noted previously, the values for primary manufacturing subsectors and the total for NAICS 321 are the same for 1990 and 1994 due to the change from SIC codes to NAICS codes in 1997. Table 4.2 shows emissions and changes over time.

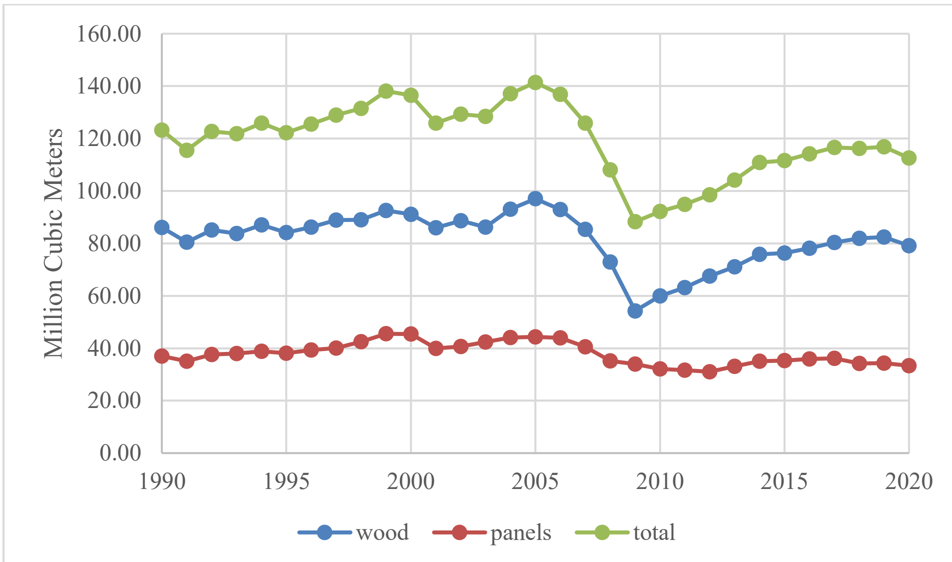


Figure 4.8. Production of US Wood Products [*Source*: FAOSTAT (2023)]

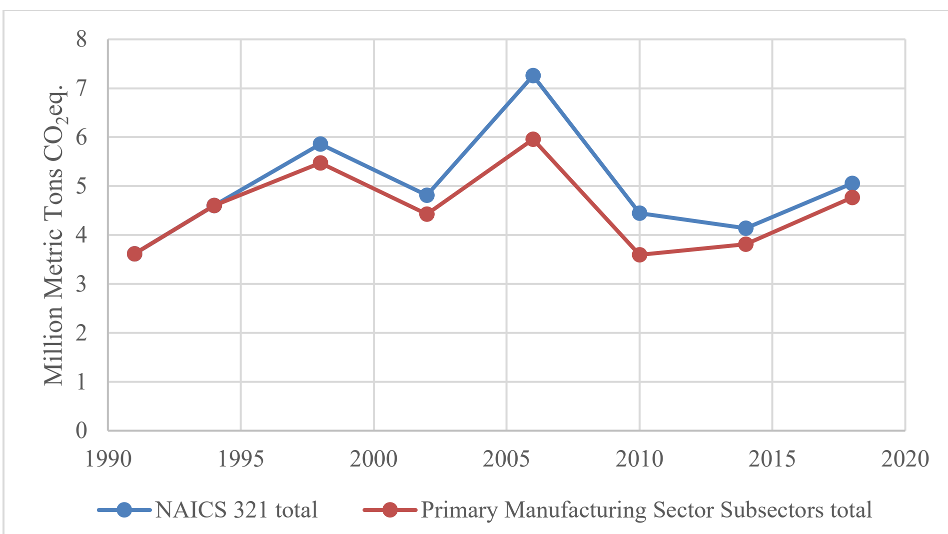


Figure 4.9. Emissions from US Wood Products Facilities [*Source*: EIA MECS data, EIA (2023a)][9]

[9] US Department of Energy withheld 2010 MECS data for distillate oil for sawmills due to confidentiality concerns. For this report, a value equal to the average of 2006 and 2014 was used.

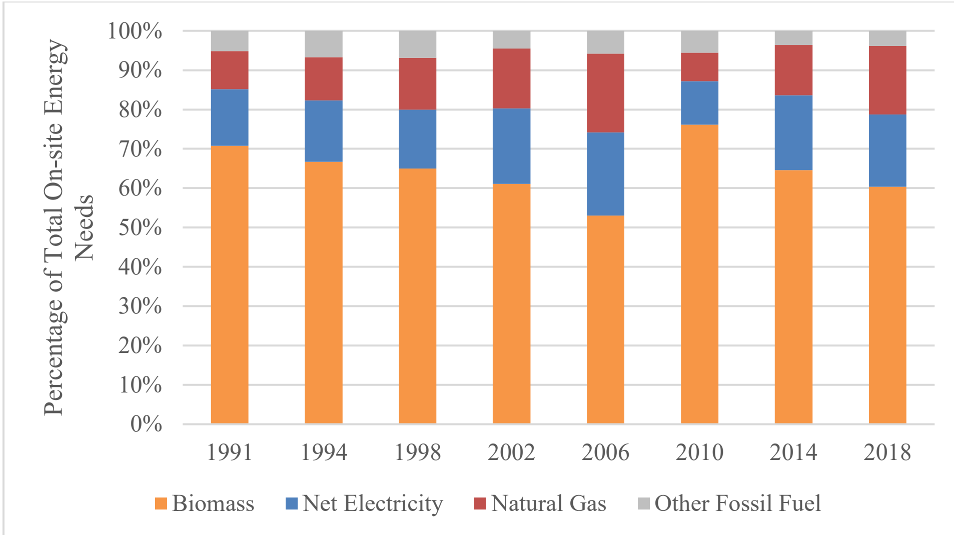


Figure 4.10. On-site Energy Mix Over Time for the US Wood Products Industry [*Source*: NCASI 2021, EIA MECS data]

Table 4.2. Combustion-Related Emissions from US Wood Products Primary Manufacturing (Total of Subsectors for NAICS 321)

	Emissions Million Metric Tons CO_2eq.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990 (EIA MECS 1991)	3.6	1.00	
2005 (EIA MECS 2006)	6.0	1.65	1.00
2020 (EIA MECS 2018)	4.8	1.32	0.80

4.1.3 Biogenic Emissions from Forest Product Industry Manufacturing

In this profile, biogenic carbon is accounted for using production accounting, the same approach used by US EPA in the annual national inventory of emissions and sinks. In production accounting, transfers of biogenic CO_2 to the atmosphere are calculated as losses in overall stocks of stored biogenic carbon (see US EPA 2022a for more discussion of production accounting). Flows of biogenic carbon to the atmosphere are not included in GHG emission totals as this would double count the quantities accounted for in the biogenic carbon stock change calculations applied to carbon in forests and forest products. Nonetheless, some reporting protocols, including the GHG Protocol Corporate Standard (GHG Protocol 2004), require that emissions of biogenic CO_2 be "reported separately." For this reason, this update includes a summary of biogenic CO_2 emissions from forest products manufacturing.

The forest products sector is unique among industrial sectors in that most energy requirements for the sector are derived from biomass energy sources. In 2018, 63% of fenceline energy needs for the US pulp and paper sector and 60% of fenceline energy needs for the US wood products sector were derived from biomass fuels. Biogenic carbon dioxide emissions have ranged between 135.7 and 144.7 million metric tons over the last 30 years (Table 4.3). The industry's biogenic carbon dioxide emissions are primarily from power boilers using biomass fuel, kraft mill recovery furnaces, and kraft mill lime kilns[10]. The biogenic emission contribution from lime kilns was approximately 9 million metric tons CO_2eq. in 2018. Figure 4.11 shows the trends in biogenic carbon dioxide emissions for US forest products manufacturing over time.

Table 4.3. Biogenic Emissions from US Pulp and Paper and Wood Product Combustion Sources

	Biogenic CO_2 Emissions (Million Metric Tons)		
Year	Pulp and Paper	Wood Products	Total
1991	118	26.7	144.7
2006	122	20.3	142.3
2018	115	20.7	135.7

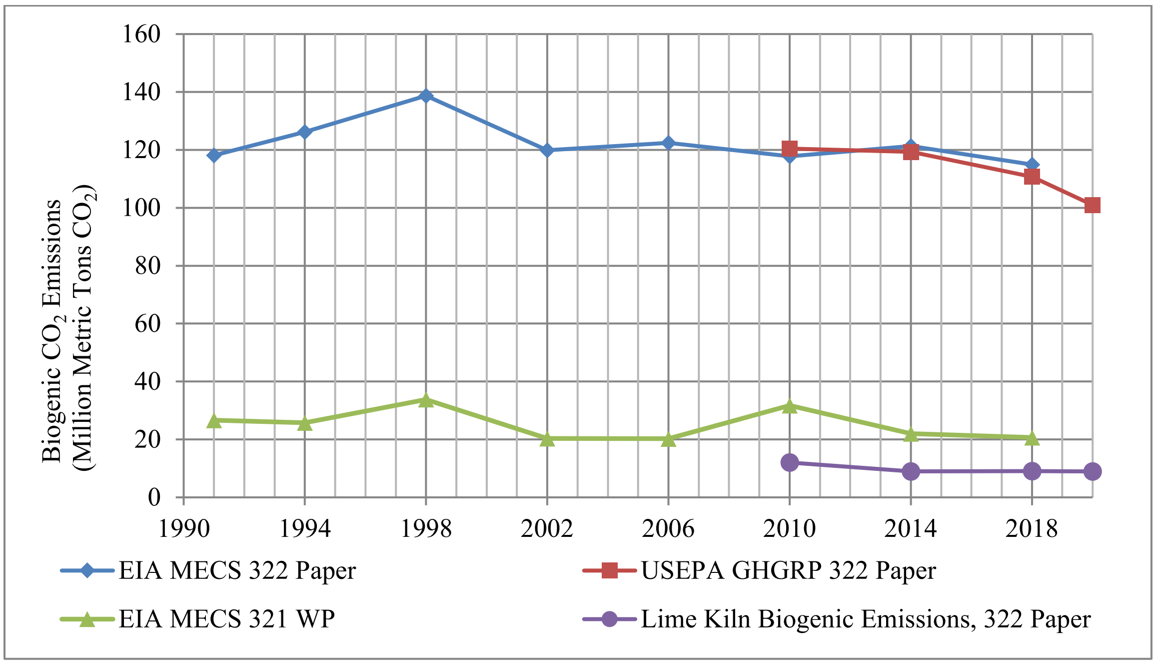


Figure 4.11. Biogenic Carbon Dioxide Emissions from US Forest Products Manufacturing [Note: the numbers in the legend are NAICS codes]

4.1.4 Converting Operations

In most cases, wood and paper products require additional manufacturing to produce final products. There are many so-called converting operations. In the case of paper and paperboard products, these include printing, packaging, cutting, folding and gluing, and many others. In the case of wood products, these include converting wood into furniture, housing, and many other products. Because

[10] Emissions from kraft lime kilns differ from CO_2 emissions from cement industry kilns because approximately 61% of CO_2 in kiln stack emissions from the pulp and paper sector are biomass-based (Miner and Upton 2002). The remaining 39% are fossil-based emissions from the fuels combusted to generate the high temperatures needed to drive the calcination reaction.

of the great variety in converting operations, any estimate of emissions associated with this part of the forest products industry value chain is subject to considerable uncertainty.

This report contains estimates of emissions from some converting operations not included in the previous profile (NCASI 2008; Heath et al. 2010). In addition, methods to estimate process energy-related emissions have been improved compared to the previous profile. As a result, there are differences between the estimates presented herein and those published in NCASI (2008) and Heath et al. (2010). Previous estimates have been revised, however, using the updated methods, allowing a consistent comparison of emissions in 1990, 2005, and 2020.

For this profile, estimates were drawn primarily from EIA MECS data. It is important to note that the EIA MECS data do not include all types of final processing operations performed on forest products. Also, for some EIA MECS sectors, materials other than forest products may be included in an NAICS code. Nonetheless, EIA MECS data provide a consistent series of data over time.

For the pulp, paper, and paperboard sector, the total for NAICS 322 was compared to the total of individual primary NAICS codes for subsectors within NAICS 322. These subsectors are NAICS 322110 (pulp mills), 322121 (paper mills except newsprint mills), 322122 (newsprint mills), and 322130 (paperboard mills). The amount by which the value for NAICS 322 was greater than the total for subcategories for primary manufacturing was assigned to converting operations. The results, shown in Figure 4.12, illustrate that, primarily because the residual values represent small differences between larger numbers, there is considerable variability in the emissions from NAICS 322 that are assigned to converting.

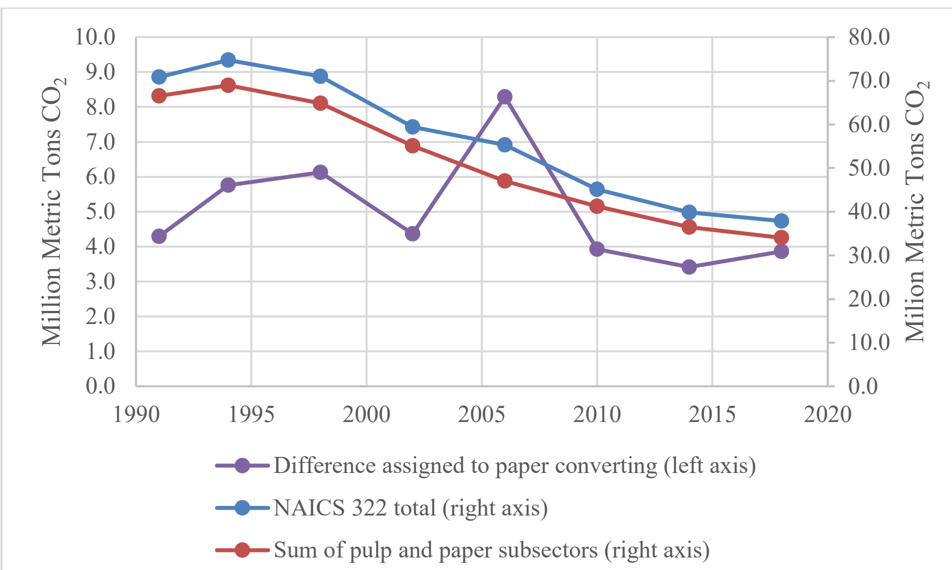


Figure 4.12. Quantity of NAICS 322 Emissions Assigned to US Paper Converting
[Note: Equals the difference between NAICS 322 and NAICS 3221, the sum of pulp and paper subsectors]

To this residual contribution from NAICS 322 were added emissions associated with NAICS 323 (printing and related support). NAICS 323 can include printing on substrates other than paper and paperboard. Examination of the description of the sector suggests that paper and paperboard products are likely dominant, but it is clear that the sector includes operations printing on non-wood-based substrates. The emissions from NAICS 323 and the converting emissions residual from NAICS 322 are shown in Figure 4.13. The total of these two, representing the total emissions assigned to paper converting in this study, are also shown.

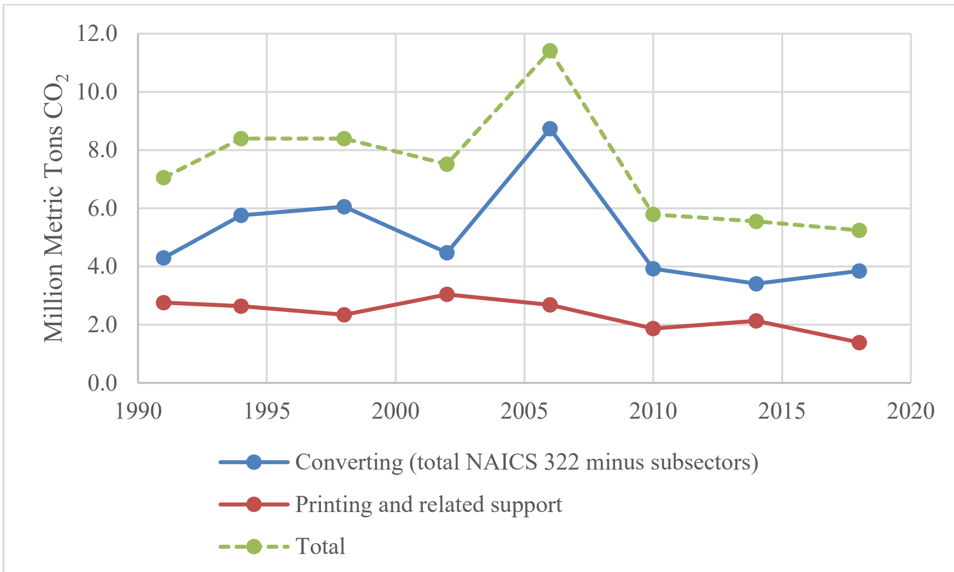


Figure 4.13. Direct Combustion-Related Emissions from US Converting Paper and Paperboard

Table 4.4 shows emissions in 1990, 2005, and 2018, assumed in this analysis to be equal to emissions in EIA MECS years 1991, 2006, and 2018, respectively.

Table 4.4. Direct Combustion-Related Emissions from US Paper and Paperboard Converting

	Emissions Million Metric Tons CO_2eq.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990 (EIA MECS 1991)	7.1	1.00	
2005 (EIA MECS 2006)	11.4	1.62	1.00
2020 (EIA MECS 2018)	5.2	0.74	0.46

Like the paper and paperboard sector, any emissions from the overall wood products sector (NAICS 321) that exceeded the total for the individual subsectors were assigned to converting. The changeover to NAICS codes in 1997, however, makes it difficult to interpret the subsector data for 1991 and 1994; therefore, subsector data were not used for these years. As a result, there is no residual (i.e., the difference between the sector total and the sum of subsectors) to assign to converting for 1991 and 1994. This can be seen in the emissions plotted in Figure 4.14. Note, however, that any converting emissions included in the wood products total is accounted for in the results for 1991 and 1994 primary manufacturing of wood products.

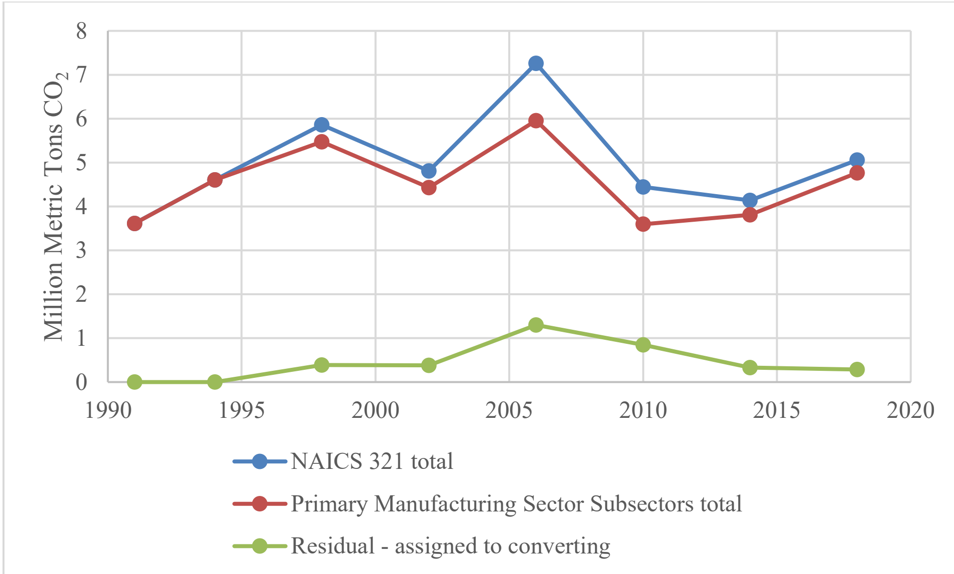


Figure 4.14. Residual from NAICS 321 Assigned to US Wood Product Converting

For the wood products sector, the only additional EIA MECS data of relevance is furniture (NAICS 337). This includes all types of furniture, wood and non-wood. EIA MECS values were allocated to wood-based furniture based on 2020 data from the American Home Furnishing Alliance. Data from the American Home Furnishing Alliance (AHFA 2022) indicate that approximately 17% of domestic furniture sales in 2020 were domestically produced wood furniture, and therefore, 17% of emissions calculated for NAICS 337 were assigned to wood converting operations[11].

An additional contribution to emissions was added to account for those associated with home construction. About 70% of softwood lumber is used in housing, and 88% of lumber produced in the US is softwood (Howard and Liang 2019). Furthermore, a significant fraction of other wood products (e.g., plywood and oriented strand board) is used in housing. There is, therefore, ample justification for including an estimate of the GHG emissions associated with the use of wood products for housing. Unfortunately, there are few robust estimates of these estimates. Puettmann et al. (2021) examined the life cycle of several mass timber buildings in the US. It was estimated that the construction phase accounted for 3%–5% of the nonrenewable fuel use in the cradle to finished house portion of the life cycle. Most of the emissions in the cradle to gate portion of the life cycle of softwood lumber are associated with manufacturing (e.g., see Puettmann et al. (2021) and Milota (2020a; 2020b)); thus, for this profile, it was assumed that emissions associated with converting wood to final products other than furniture were 4% of the average primary manufacturing emissions from wood products mills (i.e., NAICS 321) for years with EIA MECS data. The uncertainty associated with this estimate is large.

The three components of wood products direct emissions from converting operations estimated in this study are shown in Figure 4.15. The contributions of home construction and furniture production are smaller than the amounts estimated from the difference between emissions from

[11] This may understate furniture-related emissions in earlier years, given that the production of wood furniture in the US has declined since 1990.

the overall wood products sector (NAICS 321) and the sum of the listed subsectors (sawmills, veneer, plywood and engineered wood products, and other wood products).

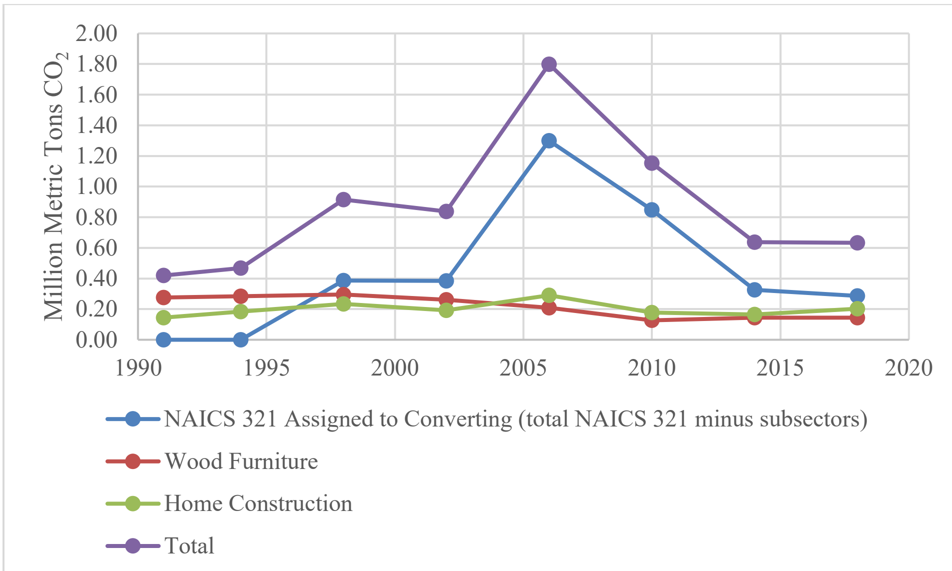


Figure 4.15. Components of Direct Fuel Combustion-Related Emissions from US Wood Product Converting Operations

The estimated direct emissions from wood product converting operations in 1990, 2005, and 2020 are shown in Table 4.5. The estimates for 1990, 2005, and 2020 are derived from EIA MECS 1991, 2006, and 2018 data, respectively. Figure 4.15 demonstrates that the changes over time are highly sensitive to the years selected and should be used with caution.

Table 4.5. Direct Combustion-Related Emissions from US Wood Product Converting

	Emissions Million metric tons CO_2eq.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990 (EIA MECS 1991)[a]	0.4	1.00	
2005 (EIA MECS 2006)	1.8	4.29	1.00
2020 (EIA MECS 2018)	0.6	1.51	0.35

[a] 1991 values are low because these converting emissions are included in the wood products primary manufacturing emissions discussed earlier.

The total converting operation emissions associated with fuel combustion are shown in Figure 4.16. The reductions are shown in Table 4.6. The uncertainties in these estimates are large, and calculated reductions are highly dependent on the dates selected.

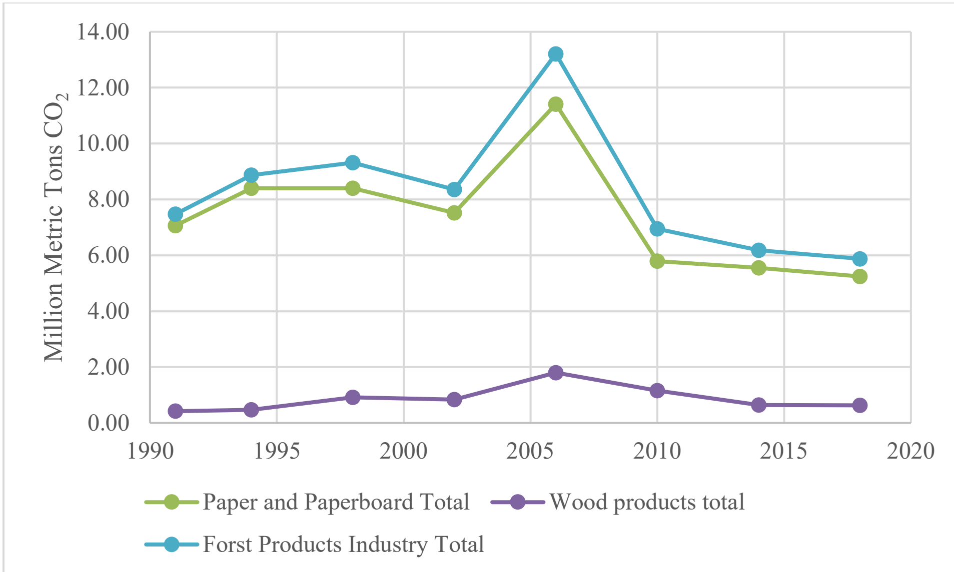


Figure 4.16. Direct GHG Emissions from Fuel Combustion Attributable to US Forest Product Converting Operations

Table 4.6. Changes in Direct Emissions from Fuel Combustion at US Forest Product Converting Operations

	Emissions Million Metric Tons CO_2eq.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990 (EIA MECS 1991)	7.5	1.00	
2005 (EIA MECS 2006)	13.2	1.77	1.00
2020 (EIA MECS 2018)	5.9	0.79	0.44

4.1.5 Total Forest Products Industry Direct Emissions from Fuel Combustion

The total combustion-related emissions from pulp, paper, paperboard, and wood products manufacturing in the US are shown in Table 4.7 and Figure 4.17. Approximately three-quarters of the 2020 emissions are associated with the pulp, paper, and paperboard primary manufacturing sector. The industry's fuel combustion–related emissions in 2020 were reduced by 44% since 1990 and 41% since 2005.

Table 4.7. Combustion-Related Emissions from US Forest Products Sector Manufacturing

Year	Million Metric Tons $CO2_e$q.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990	78.0	1.00	
2005	74.4	0.95	1.00
2020	43.6	0.56	0.59

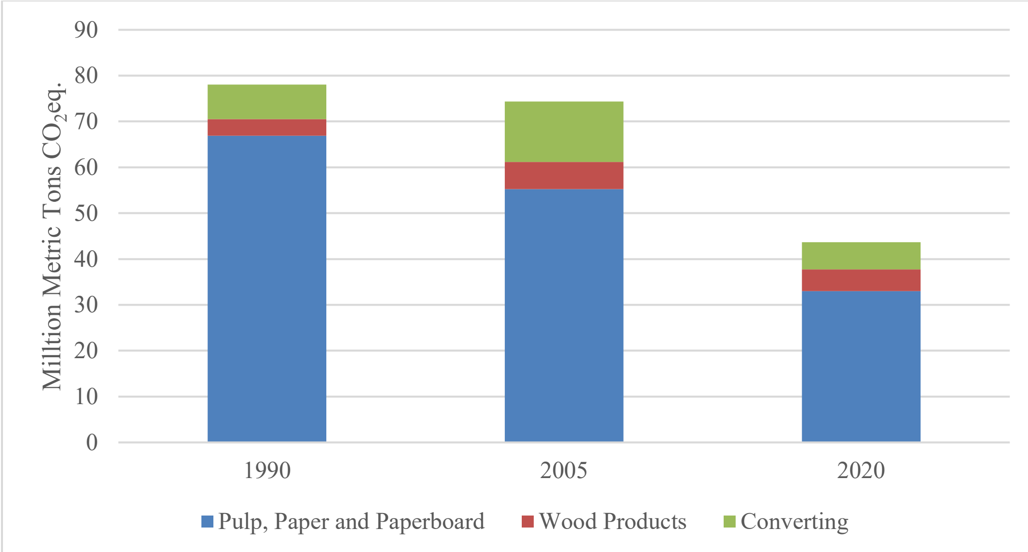


Figure 4.17. Direct Combustion-Related Emissions from US Pulp and Paper and Wood Products Manufacturing

4.2 Scope 2 Indirect Emissions Associated with Purchased Electricity and Steam

Emissions from purchased electricity are calculated by multiplying electricity purchases by the associated GHG emission factor. Details on emission factor sources and GWPs used in calculations are provided in Appendix B. Where possible, the emissions estimates in this section are shown in two ways, based on gross electricity purchases and net electricity purchases. Many inventory protocols, including the GHG Protocol, require estimates based on gross purchases (GHG Protocol 2004). Estimates based on net purchases, however, can be more reflective of the amounts of electricity consumed in manufacturing. Where data allow, estimates are also shown that include purchases of steam. Gross purchases are used as the baseline estimates for this study. The previous GHG profile study used net purchases to calculate Scope 2 emissions (NCASI 2008; Heath et al. 2010).

4.2.1 Pulp and Paper Mills

4.2.1.1 Data Sources

Data sources are the same as those used to calculate direct fuel-related emissions, discussed previously.

4.2.1.2 Greenhouse Gas Emissions on an Absolute Basis

Figure 4.18 shows emissions and reductions in absolute terms attributed to purchased electricity and steam using AF&PA/API and EIA MECS data sets. Separate data are shown for gross purchases and net purchases of electricity. Emissions attributable to purchased steam account for about 9% (range from 5 to 13%) of emissions from gross purchases of steam and electricity. Since 2008, there has been accelerated greening of the US electrical grid, and most reductions in purchased electricity GHG emissions have been achieved since then. GHG emissions from purchased electricity were relatively constant between 1990 and 2008.

Because the AF&PA/API data set includes steam purchases, and because it is the more complete data set, it has been used to calculate changes over time in this analysis. Tables 4.8 and 4.9 show that emissions reductions from 1990 to 2020 are approximately 50% for both gross- and net-electricity–based estimates. The same is true for the period of 2005 to 2020. Emissions associated with net energy purchases (including steam) were 20.0%, 16.6%, and 17.7% lower than those based on gross purchases (including steam) in 1990, 2005, and 2020, respectively.

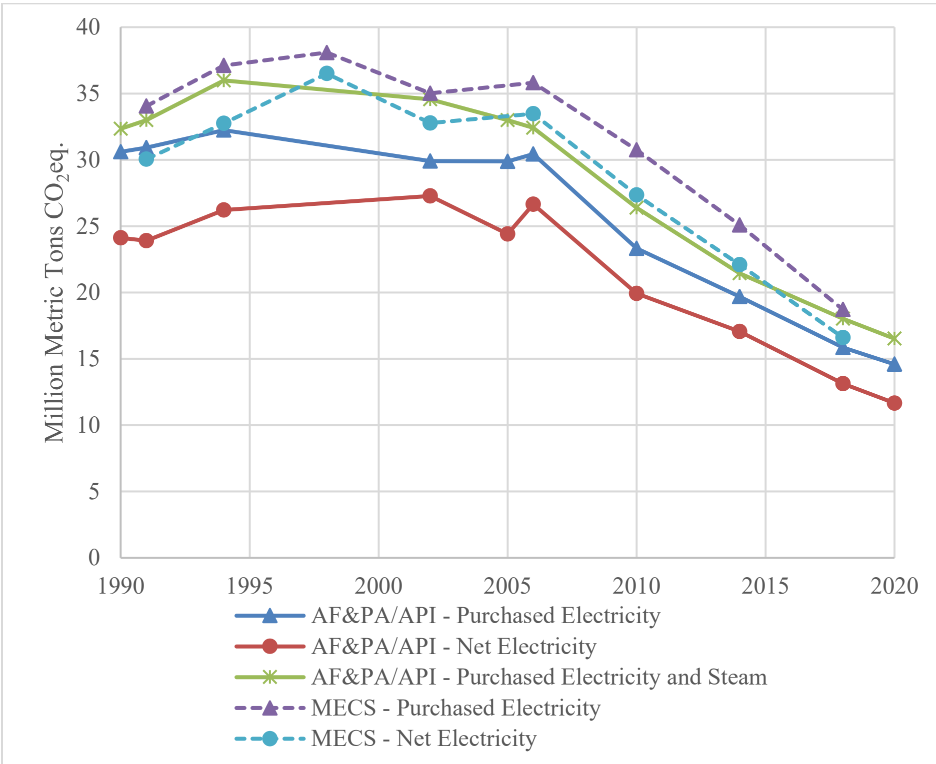


Figure 4.18. Changes in GHG Emissions Associated with Electricity and Steam Purchased by the US Pulp and Paper Sector

Table 4.8. Indirect Emissions Attributable to Gross Electricity and Steam Purchases in the US Paper, Paperboard, and Market Pulp Sector [*Source*: NCASI 2021, (AF&PA/API data)]

Year	Million metric tons CO_2eq.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990	32.3	1.00	
2005	33.0	1.02	1.00
2020	16.5	0.51	0.50

Table 4.9. Indirect Emissions Attributable to Net Electricity Purchases in the US Paper, Paperboard, and Market Pulp Sector [*Source*: AF&PA/API data, purchased steam not included]

Year	Million Metric Tons CO_2eq.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990	24.1	1.00	
2005	24.4	1.01	1.00
2020	11.7	0.48	0.48

4.2.1.3 Greenhouse Gas Emissions on an Intensity Basis

Figure 4.19 shows emission intensity reductions attributed to net purchased electricity over time. While the percent changes over time within EIA MECS data and AF&PA/API data are similar, EIA MECS emission intensities are consistently higher than indicated by AF&PA/API data. EIA MECS data likely capture more nonintegrated and recycle facilities than are represented in the AF&PA/API data set. Nonintegrated and recycle mills almost exclusively purchase electricity to meet facility electricity demand because they generate very little on-site electricity, which may be a contributing factor to the difference in net purchased electricity between the two data sets.

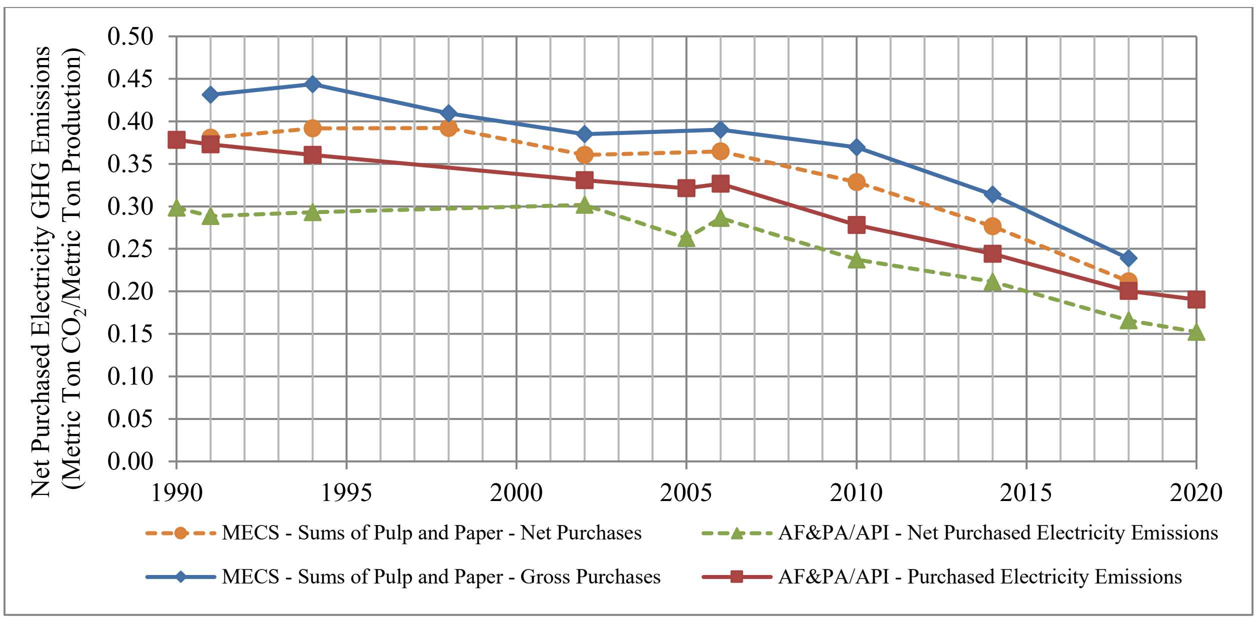


Figure 4.19. Purchased Electricity GHG Emission Intensity Changes Over Time from AF&PA/API Data and EIA MECS Data [Note: Purchased steam not included]

4.2.1.4 Discussion

Some of the factors driving reductions in the pulp and paper sector's purchased electricity-related GHG emissions are discussed here. Purchased electricity remained essentially constant between 1991 and 2018, at 7% of the energy mix. Therefore, this is not a significant factor in the reductions shown earlier in this report. Changes in the electricity grid and improvements in energy intensity, however, have contributed to reduced emissions.

Greening of the Electricity Grid

The GHG emission intensity associated with purchased electricity has been decreasing due to several factors, such as proliferation of state renewable portfolio standards, state and federal environmental regulations, and market forces. Since 1996, US EPA has maintained the Emissions & Generation Resource Integrated Database (eGRID), a comprehensive database on the environmental characteristics of almost all electric power generated in the US (US EPA 2023c). Figure 4.20 shows the decrease in US national average purchased electricity emission factors, from eGRID time series information. Since 2005, there has been a 38% reduction in the national average purchased electricity GHG emission factor.

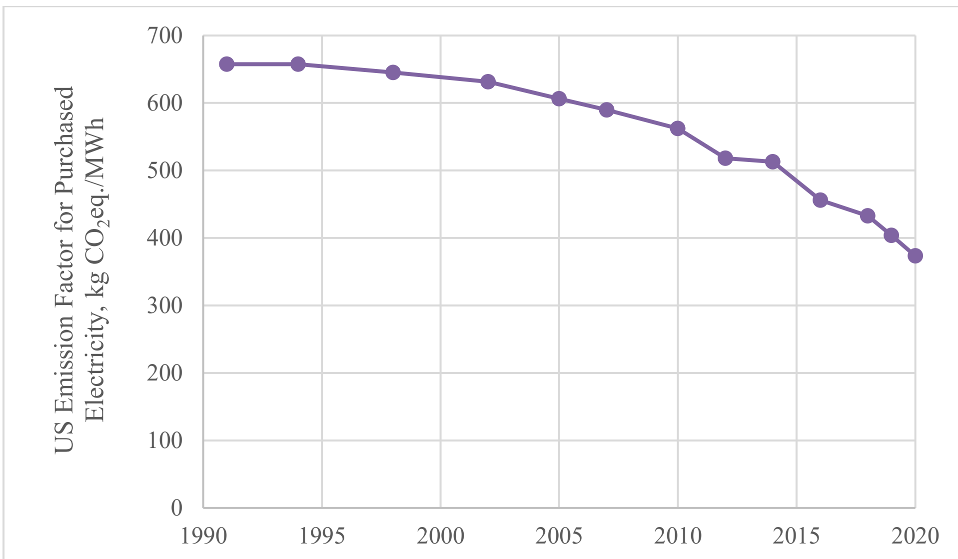


Figure 4.20. Purchased Electricity GHG Emission Factors [*Sources*: eGRID and US Department of Energy]

Energy Intensity Improvements

The overall energy intensity of the US pulp and paper sector has decreased over time as mills have implemented energy efficiency improvements and installed more energy-efficient equipment. Figure 4.6 shows total fenceline energy intensity over time divided between fossil fuels, net purchased electricity and steam, and biomass and other renewables. Fenceline energy needs from biomass have remained essentially constant, but the share of fossil fuel, net purchased electricity, and steam has decreased over time.

Production Reductions

As highlighted in the discussion of direct emissions earlier in this report, shuttered production over time in the US has tended to be associated with older, less energy-efficient facilities, and closure of these facilities has led to overall improvements in both energy intensity and GHG intensity in the industry.

4.2.2 Wood Products Mills

Data on electricity purchases by wood products mills were obtained from EIA MECS (EIA 2023a). Purchases by primary wood products producers were calculated as the sum of purchases by NAICS 321113 (sawmills), 3212 (veneer, plywood, and engineered wood products), and 3219 (other wood products)[12]. Emission factors for purchased electricity were the same as those used for the pulp and paper sector. Purchased steam was not included because (1) wood products mills seldom purchase steam and (2) EIA MECS does not provide data for purchased steam. The changeover to NAICS codes in 1997 makes it difficult to interpret wood products subsector data for these years, so only the sector totals are used. These totals may include some converting operations. As a result, there is no residual (i.e., the difference between the sector total and the sum of subsectors) to assign to converting for 1991 and 1994. For these years, converting emissions are included with primary wood products manufacturing emissions.

The results, shown in Figure 4.21 with the sector's production data (FAOSTAT 2023), indicate that until approximately 2014 purchased electricity-related emissions were closely related to production. From 2014 forward, however, the effects of reductions in the GHG intensity of the grid (shown in Figure 4.20) far outweighed the effects of increased production, resulting in significant reductions in emissions even while production was increasing. The emissions and reduction estimates for 1990 (based on EIA MECS 1991 data), 2005 (based on EIA MECS 2006 data), and 2020 (based on EIA MECS 2018 data) are shown in Table 4.10. Emissions associated with gross purchased electricity in 2020 were 43% lower than in 2005 and 27% lower than in 1990. Reductions in emissions associated with net electricity, shown in Table 4.11, are similar.

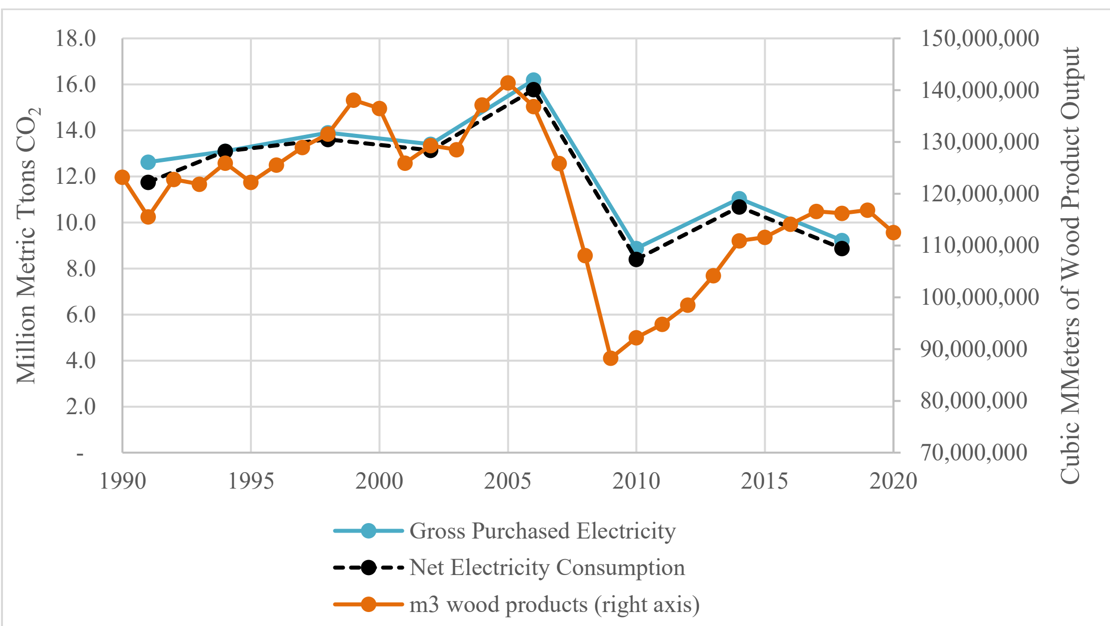


Figure 4.21. Wood Products Production and Purchased Electricity-Related Emissions

[12] The EIA MECS data for 1994 do not include values for purchased electricity at wood products facilities in order "to avoid disclosing data for individual establishments." See EIA MECS 1994, Table A25 for details.

Table 4.10. Emissions Associated with Gross Electricity Purchases by the US Wood Products Sector (NAICS 321113, 3212, and 3219)

Year	Million Metric Tons CO_2eq. per Year	Fraction of 1990 (MECS 1991)	Fraction of 2005 (MECS 2006)
1990 (EIA MECS 1991)	12.6	1.00	
2005 (EIA MECS 2006)	16.2	1.28	1.00
2020 (EIA MECS 2018)	9.2	0.73	0.57

Table 4.11. Emissions Associated with Net Electricity Consumption by the US Wood Products Sector (NAICS 321113, 3212, and 3219)

Year	Million Metric Tons CO_2eq. per Year	Fraction of 1990 (MECS 1991)	Fraction of 2005 (MECS 2006)
1990 (EIA MECS 1991)	11.8	1.00	
2005 (EIA MECS 2006)	15.8	1.34	1.00
2020 (EIA MECS 2018)	8.9	0.76	0.56

4.2.3 Converting Operations

Scope 2 emissions for converting operations were estimated using essentially the same approach as used for direct fuel combustion-related emissions (described previously). It is assumed that for converting operations, gross and net electricity purchases are equal; therefore, results are shown for gross purchases only.

The components of paper and paperboard converting emissions associated with gross purchased electricity are shown in Figure 4.22 and the estimates for 1990 (EIA MECS 1991), 2005 (EIA MECS 2006), and 2020 (EIA MECS 2018) are shown in Table 4.12. In 2020, these emissions were approximately 45% lower than in 1990 and 2005.

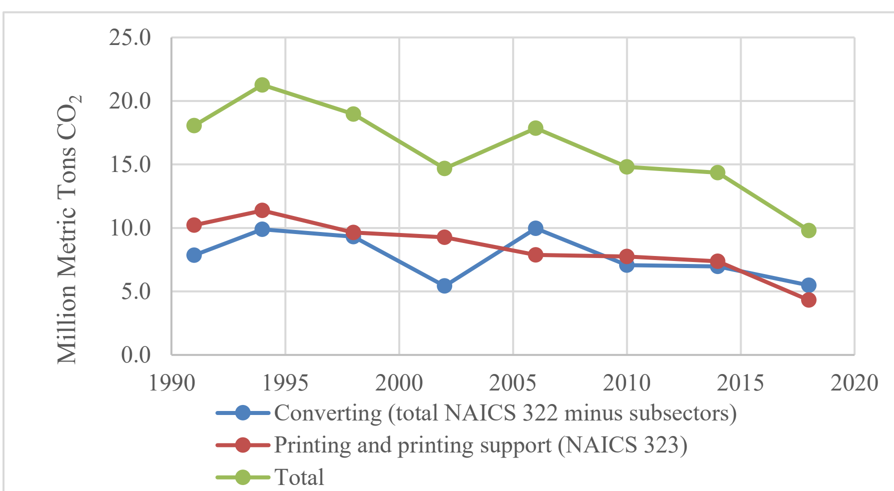


Figure 4.22. Components of US Paper and Paperboard Converting Emissions Associated with Gross Purchased Electricity

Table 4.12. Emissions Associated with Gross Electricity Purchased by US Converting Operations in the Pulp, Paper, and Paperboard Value Chain

Year	Million Metric Tons CO_2eq. per Year	Fraction of 1990	Fraction of 2005
1990 (EIA MECS 1991)	18.1	1.00	
2005 (EIA MECS 2006)	17.9	0.99	1.00
2020 (EIA MECS 2018)	9.8	0.54	0.55

The components of purchased electricity-related emissions for wood products converting are shown in Figure 4.23 and the estimates for 1990 (EIA MECS 1991), 2005 (EIA MECS 2006), and 2020 (EIA MECS 2018) are shown in Table 4.13. Again, the residual contribution attributed to converting emissions, calculated as the difference between the NAICS 321 total and the sum of the subcategories, is highly variable because it is a small difference between two large numbers. The changeover to NAICS codes in 1997 makes it difficult to interpret the wood products subsector data for 1991 and 1994; therefore, only the sector totals are used. This results in zero emissions residual to be assigned to converting for these years. These emissions are accounted for, however, in the indirect emissions from primary wood product manufacturing. The sources and methods for estimating contributions to purchased electricity-related emissions from furniture and home construction are the same as discussed earlier to estimate fuel combustion-related emissions. It should be noted that the source of the estimate for energy consumption in building construction (Puettman et al. 2021) does not specify the amounts of electricity used. It states only that "construction energy used only diesel fuel and accounted for 3–5% on the nonrenewable fuel use…" For this profile, it is assumed that this figure can be applied to estimate electricity use, likely a conservative assumption that overstates electricity requirements in building construction.

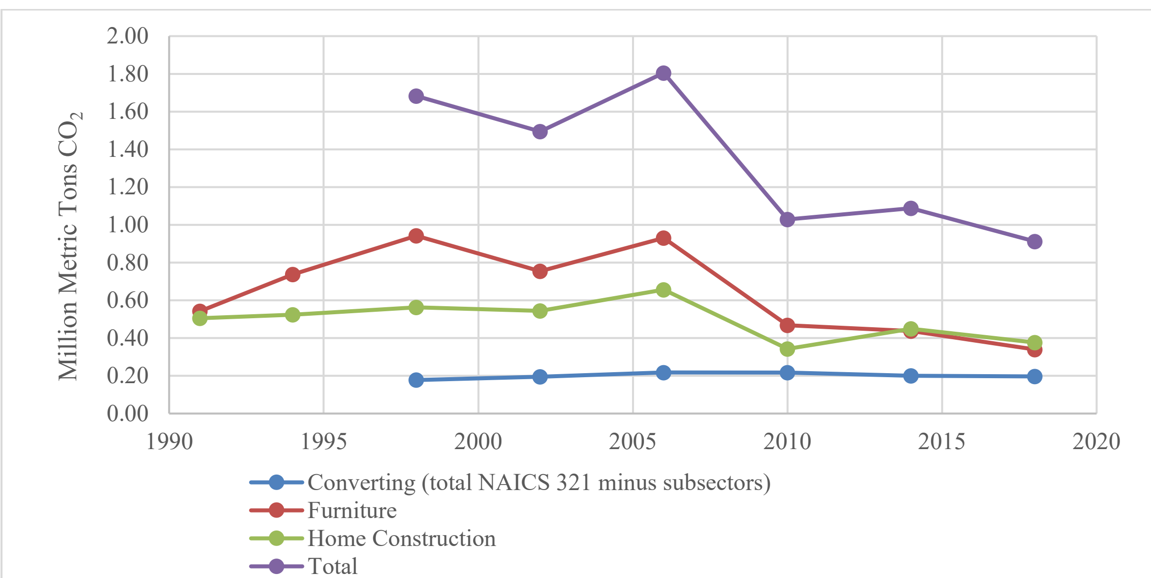


Figure 4.23. Components of Emissions Associated with Gross Purchased of Electricity by US Wood Product Converting Operations

Table 4.13. Emissions Associated with Gross Purchases of Electricity by Converting Operations in the US Wood Products Value Chain

Year	Million Metric Tons CO_2eq. per Year	Fraction of 1990	Fraction of 2005
1990 (EIA MECS 1991)	1.0	1.00	
2005 (EIA MECS 2006)	1.8	1.72	1.00
2020 (EIA MECS 2018)	0.9	0.87	0.51

Summing converting emissions for the paper sector and wood products sector results in the totals for converting shown in Figure 4.24 and Table 4.14. On average, 93% (ranging from 91 to 95%) of the converting emissions associated with purchased electricity are in the pulp and paperboard sector.

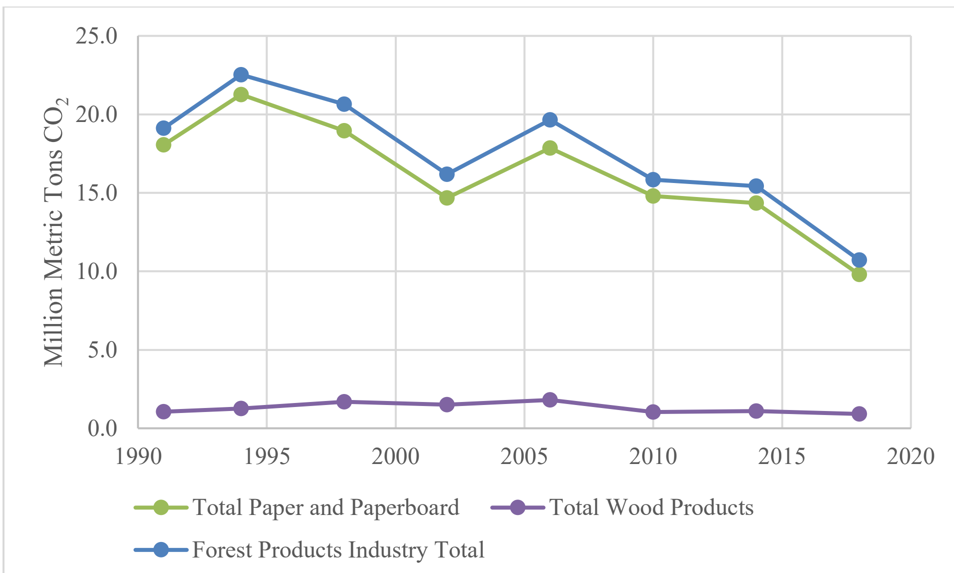


Figure 4.24. Emissions Associated with Gross Electricity Purchases at US Forest Products Industry Converting Operations

Table 4.14. Reductions in Emissions Associated with Gross Electricity Purchases at Converting Operations in the US Forest Products Industry

Year	Million Metric Tons CO_2eq. per year	Fraction of 1990	Fraction of 2005
1990 (EIA MECS 1991)	19.1	1.00	
2005 (EIA MECS 2006)	19.7	1.03	1.00
2020 (EIA MECS 2018)	10.7	0.56	0.54

4.2.4 *Total Forest Products Industry*

The indirect emissions associated with gross electricity and steam purchases by the US forest products industry are shown in Figure 4.25. In 2020, the pulp, paper, and paperboard sector accounted for 45% of these emissions. Wood products accounted for 25%. The converting sector accounted for 30%, almost all of which was associated with converting paper and paperboard.

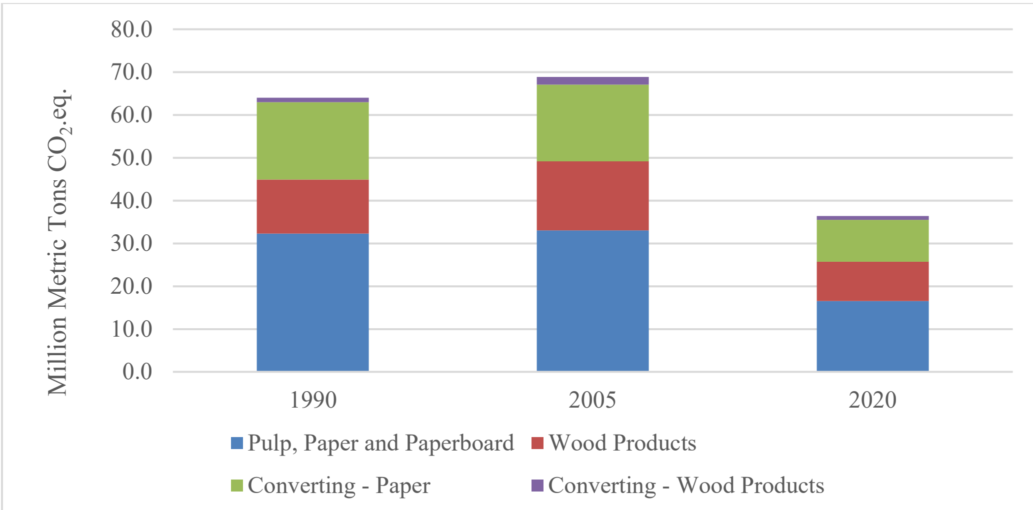


Figure 4.25. Scope 2 Emissions from the US Forest Products Industry in 1990, 2005, and 2020

Reductions over time are quantified in Table 4.15. These emissions have been reduced by 43% since 1990 and 47% since 2005. Much of the reduction has been driven by reductions in the GHG intensity of purchased electricity which, in 2020, was 38% lower than in 2005.

Table 4.15. Scope 2 Emissions Associated with Gross Electricity and Steam Purchases by the US Forest Products Industry

Year	Million Metric Tons CO_2eq.						
	Pulp, Paper, and Paperboard	Wood Products	Converting – Paper	Converting – Wood Products	Total	Fraction of 1990	Fraction of 2005
1990	32.3	12.6	18.1	1.0	64.0	1.00	
2005	33.0	16.2	17.9	1.8	68.8	1.08	1.00
2020	16.5	9.2	9.8	0.9	36.4	0.57	0.53

4.3 Scope 3 Indirect Emissions Associated with the Production and Transport of Fossil Fuels Used by the US Forest Products Industry

GHGs are emitted in the production and transport of fuels used in the forest products industry. Upstream emission factors for the production and transport of fuels used by the industry were obtained from NCASI's Scope 3 GHG Screening Tool, version 1.1a (NCASI 2023a). These factors indicate that upstream GHG emissions associated with natural gas, oil, and coal are 19.8%, 16.1%, and 6.1% of the combustion emissions, respectively. Using these and EIA MECS data to determine the mix of fuels in 1991, 2006, and 2018, a fuel-weighted ratio of upstream to combustions emissions was developed for each year and sector. These ratios are shown in Table 4.16. In some cases, these ratios have increased over time due to an increased reliance on natural gas, for which upstream emissions are higher than for other fossil fuels.

Table 4.16. Ratio of Upstream to Direct Emissions Associated with Fossil Fuels Used in the US Forest Products Industry

Ratio of Upstream (Production and Transport) to Direct Emissions			
	1991	2006	2018
Pulp and paper	0.13	0.13	0.18
Pulp and paper converting	0.19	0.20	0.20
Wood products	0.19	0.19	0.19
Wood products converting	0.19	0.19	0.19

The ratios for 1991, 2006, and 2018 were then applied to the estimates of combustion-related emissions for 1990, 2005, and 2020, respectively, with the results shown in Table 4.17.

Table 4.17. Emissions Associated with the Production and Transport of Fossil Fuels Used by the US Forest Products Industry

	Upstream Million Metric Tons CO_2eq.		
	1990	2005	2020
Pulp and paper	9.0	7.4	5.9
Pulp and paper converting	1.4	2.2	1.0
Wood products	0.7	1.1	0.9
Wood products converting	0.1	0.3	0.1
Total	11.1	11.1	7.9

4.4 Scope 3 Indirect Emissions Associated with Purchased Electricity and Steam

Emissions in 1990, 2005, and 2020 associated with upstream processes involved in producing purchased electricity were estimated using data from the National Renewable Energy Laboratory (NREL) on the ongoing upstream GHG emissions attributable to different types of power (NREL 2021) and information on the composition of the grid, as reported by eGRID (US EPA 2023c).

Although the NREL fact sheet (NREL 2021) suggests that its factors include transport emissions, the underlying literature source (Whitaker et al. 2012) indicates that transport is not included in the factors. Whitaker et al. (2012) also indicate that where transport is included, it typically adds 3 to 5% to life cycle coal-based electricity emissions. The NREL fact sheet indicates that these life cycle emissions are about 1,000 gmCO_2e/kWh, and therefore, the factor used for coal and oil in the NREL study was increased by 40 gmCO_2e/kWh (i.e., from 10 to 50 gCmO_2e/kWh). Emissions in 1990 were estimated using eGRID data for 1996, the earliest data in eGRID. The NREL fact sheet does not include a factor for biomass, thus the effects of biomass were investigated by varying the factor for biomass from zero up to the value for coal. This range did not impact the overall Scope 3 factor within two significant figures. The Scope 3 factors from NREL are shown in Table 4.18, while the composition of the grid is shown in Table 4.19. The resulting Scope 3 factors based on the composition of the grid are shown in Table 4.20.

Table 4.18. Ongoing Scope 3 Emissions Associated with Producing Electricity [*Source*: NREL (2021)]

Type of Generation	Ongoing Upstream Emissions, gm CO_2eq./kwh
Natural gas	71
Coal	10 + 40 = 50
Nuclear	12
Hydro	1.9
Wind	0.74
Solar	0.74
Geothermal	6.9
Oil	No value; therefore, assumed same as coal

Table 4.19. Composition of Supply to the US Electricity Grid [*Source*: eGRID, US EPA (2023c)]

	Composition of Supply to US Grid		
Type of Generation	1996[a]	2005	2020
Coal	52.5%	49.6%	19.3%
Natural gas	12.3%	18.8%	40.5%
Oil	4.5%	3.5%	0.9%
Hydro	9.6%	6.7%	7.1%
Nuclear	19.0%	19.3%	19.7%
Wind	0.1%	0.4%	8.4%
Biomass	1.5%	1.3%	1.4%
Geothermal	0.5%	0.4%	0.4%
Solar	0.0%	0.0%	2.2%

[a] 1996 data are the earliest available in eGRID and were used for 1990.

Table 4.20. Scope 3 Emission Factors for Purchased Electricity

Year	Kg CO_2eq. per Purchased MWh US Average	Ratio of Upstream Emission Factor to Combustion Emission Factor
1990	38	0.058
2005	41	0.068
2020	41	0.110

Scope 3 emissions associated with power purchased by the industry can be calculated using the factors in Table 4.20 and information on emissions associated with purchased electricity. The results are shown in Table 4.21. The upstream Scope 3 emissions associated with the production of purchased electricity and steam are a small fraction of the industry's emissions profile, being only 3.7, 4.7, and 4.0 million metric tons CO_2eq. in 1990, 2005, and 2020, respectively.

Table 4.21. Scope 3 Emissions Associated with Electricity and Steam Purchased by the US Forest Products Industry

Year	Million Metric Tons CO_2eq.				Fraction of 1990	Fraction of 2005
	Pulp, Paper, and Paperboard	Wood Products	Converting – Forest Products Industry	Total		
1990	1.9	0.7	1.1	3.7	1.00	
2005	2.2	1.1	1.3	4.7	1.26	1.00
2020	1.8	1.0	1.2	4.0	1.08	0.86

5.0 EMISSIONS ASSOCIATED WITH PRODUCING AND HARVESTING WOOD

GHG emissions are released in the production and harvesting of roundwood, for instance from equipment used for planting and harvesting. Chemicals may be used for weed and pest control and fertilization, and fire may also be used as a management tool. Factors that include these emissions have been developed for NCASI's Scope 3 GHG Screening Tool, version 1.1a (NCASI 2023a) and have been used in this study.

Factors are available for production and harvesting of hardwood and softwood from the north and south US. The fraction of northern and southern, hardwood and softwood, were calculated from Table 39 in Oswalt et al. (2019). Wood from other regions, which accounts for approximately 20% of US harvest, was ignored in calculating a weighted average factor for the US. The regional factors, the fraction of total harvest in each category, and the weighted overall factor, are shown in Table 5.1.

Table 5.1. Development of a Weighted National Emission Factor for US Wood Production and Harvesting

Wood Type	Emission Factor, kg CO_2eq./kg Dry Wood	Fraction of Harvest
Northern hardwood logs	0.0123	0.24
Northern softwood logs	0.0080	0.06
Southern hardwood logs	0.0123	0.15
Southern softwood logs	0.0106	0.56
Weighted emission factor	0.0111	1.00

The weighted factor was assumed to apply equally to current and past wood production and harvesting. Emissions were therefore calculated by multiplying the weighted factor in Table 5.1 by the quantity of roundwood production in 1990, 2005, and 2020. Roundwood production data were obtained from FAOSTAT (FAOSTAT 2023). It was assumed that a cubic meter of green wood weighs 0.9 metric tons and all wood is 50% water. The resulting emissions in 1990, 2005, and 2020 are shown in Table 5.2. The reductions in emissions reflect reductions in domestic roundwood production. Note that these do not include transport-related emissions, which are addressed elsewhere in this report.

Table 5.2. Emissions Associated with Producing and Harvesting Wood

Year	Million Metric Tons CO_2eq.	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990	2.5	1.00	
2005	2.3	0.92	1.00
2020	2.1	0.84	0.92

In its annual inventory, US EPA estimates emissions associated with nitrogen fertilizer use in forestry (US EPA 2020a). US EPA's estimates for 1990, 2005, and 2020 are 0.01, 0.05, and 0.05 million metric tons CO_2eq. (US EPA 2020 inventory). It is reasonable that these would be less than those estimated herein, as the estimates in Table 5.2 include not only emissions associated with fertilizer use, but also, most notably, emissions associated with producing and using forest chemicals and fossil fuels.

6.0 GREENHOUSE GAS EMISSIONS ASSOCIATED WITH PRODUCING NONFIBER, NONFUEL INPUTS

6.1 Approach

To estimate upstream emissions associated with nonfiber, nonfuel inputs, NCASI divided the industry into the product types shown in Table 6.1. Upstream emissions associated with nonfiber inputs to manufacturing (not including fuels) were estimated using factors contained in the Forest Industry Carbon Assessment Tool (FICAT), a model developed by NCASI for the International Finance Corporation of the World Bank. Information on FICAT and the factors it contains are available from NCASI (NCASI 2023b). The FICAT factors are shown in Table 6.1. Production levels for pulp, paper, and paperboard were obtained from AF&PA statistical reports (AF&PA 2020; 2021a). Data on wood product output was obtained from Howard and Liang (2019). The most recent wood products data available were for 2017; therefore, these were used to represent 2020.

6.2 Results and Discussion

The emissions estimates are shown in Table 6.2. By assessing these emissions by product type, it is possible to identify two factors, other than total industry output, that have affected trends in emissions. First, AF&PA statistics indicate that between 2005 and 2020, the production of newsprint and printing/writing grades of paper dropped by almost 70% (AF&PA 2021a). The factors in Table 6.1 indicate that these grades tend to be associated with higher upstream nonfiber, nonfuel emissions than many other grades of paper and paperboard. This has caused the overall production-weighted average factor for the paper, paperboard, and market pulp sector to decline, being 94.6, 92.8, and 83.4 kg CO_2/metric ton production in 1990, 2005, and 2020, respectively.

The previous profile (NCASI 2008; Heath et al. 2010), which focused on emissions in 1990 and 2005, used a factor of 100 kg CO_2/metric ton for the entire sector, which, based on the values shown earlier in this report, appears to have been a reasonable value for that period. Between 2005 and 2017, however, the structural change in the industry reduced the average factor for the paper, paperboard, and market pulp sector by about 10%. Total production decreased by approximately 20% over this period. The combined effect of these factors over this period was a 26% reduction in emissions attributable to nonfiber, nonfuel inputs to paper, paperboard, and market pulp mills. The effect of the structural changes in the paper industry on the forest products industry's overall emissions is evident in Figure 6.1.

Table 6.1. Product Types and Associated Emission Factors

Product Type	FICAT Emission Factor for Nonfiber, Nonfuel Materials used in Manufacturing (NCASI 2023b) (kg CO_2/Metric Ton Product)
Paper, Paperboard, and Market Pulp	
Newsprint	75
Coated freesheet	300
Coated mechanical	200
Uncoated mechanical	60
Uncoated freesheet	100
Packaging and special industrial	20
Linerboard – Virgin	20
Linerboard – Recycled	20
Semi-Chem Medium	30
Medium – Recycled	40
Solid bleached paperboard	40
Recycled and other board	200
Tissue	50
Market pulp	200
Wood Products	
Creosote-treated wood	700
Untreated lumber	0
Other preservative-treated lumber	50
Panels, veneer, and misc. products	200

Table 6.2. Emissions Associated with Nonfiber, Nonfuel Inputs to US Forest Products Manufacturing

Sector	Million Metric Tons CO_2		
Paper, Paperboard, and Market Pulp			
	1990	2005	2020
Newsprint	1.8	1.5	0.1
Coated freesheet	0.6	1.0	0.4
Coated mechanical	0.2	0.3	0.1
Uncoated mechanical	0.2	0.2	0.0
Uncoated freesheet	0.2	0.2	0.1
Packaging and special industrial	0.1	0.1	0.1
Linerboard – Virgin	0.3	0.4	0.4
Linerboard – Recycled	0.01	0.1	0.2
Semi-Chem Medium	0.2	0.2	0.2
Medium – Recycled	0.1	0.1	0.3
Solid bleached paperboard	0.8	1.0	0.9
Recycled and other board	0.5	0.4	0.4
Tissue	1.1	1.3	1.4
Market pulp	1.4	1.6	1.7
Total Paper, Paperboard, and Market pulp	7.4	8.4	6.2
Wood Products			
	1990	2005	2017
Creosote-treated products	0.7	0.7	0.7
Other preservative-treated wood	Negligible		
Untreated lumber	0.3	0.3	0.3
Panels, veneer, and other wood products	4.4	5.4	4.2
Total Wood Products	5.4	6.5	5.2
Total Forest Products Industry			
	1990	2005	2017/2020
Total Forest Products Industry	12.8	14.9	11.5

There have also been changes in the wood products sector. The weighted average emission factors in 1990, 2005, and 2017 were 73, 81, and 81 kg CO_2/metric ton, respectively. The increase from 1990 to 2005 was primarily attributable to the growing share of wood products production represented by panels. The major factor affecting total emissions from 2005 to 2017, however, was wood product output, which decreased by about 20% over that period, explaining most of the 19% decline in emissions attributable to nonfiber, nonfuel inputs to the wood products sector.

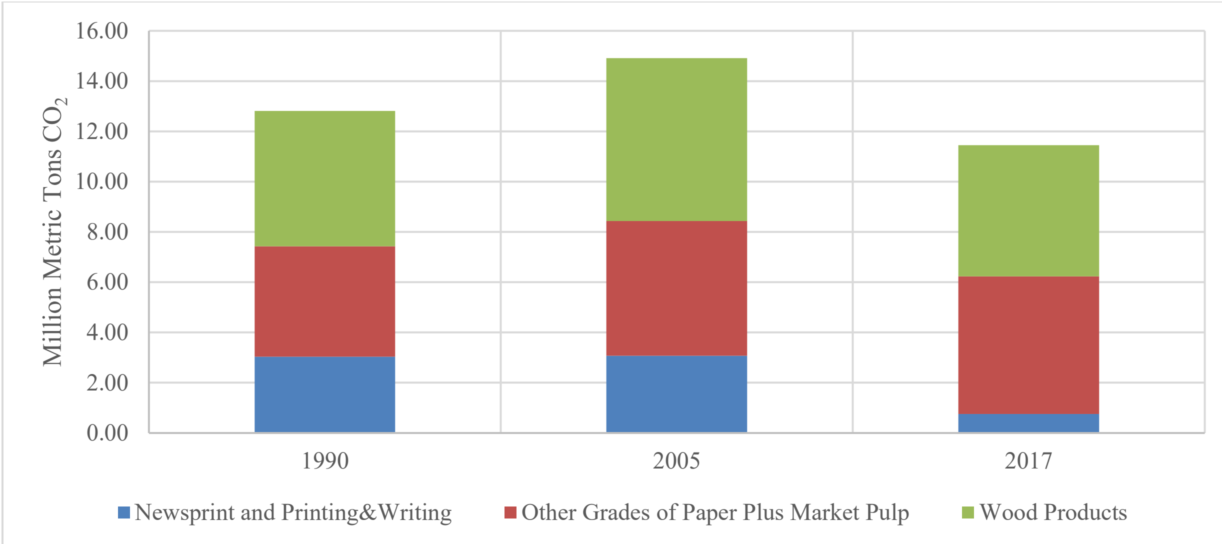


Figure 6.1. Illustration of the Effects of Structural Changes on US Forest Products Industry Emissions Attributable to Nonfiber, Nonfuel Inputs to Manufacturing

7.0 TRANSPORT-RELATED EMISSIONS

The forest products industry's value chain requires transport of fibrous and nonfibrous raw materials, products, fuels, and wastes. This section contains calculations for the emissions associated with these transport operations except for the transport of fossil fuels used by manufacturing and converting operations (which is included in the factors used to calculate Scope 3 emissions for fuels earlier in this report). In the context of this report, these emissions are those associated with fossil fuel consumption by transport vehicles and are reported as Scope 3 whether the vehicles are owned by a forest products company or not.

7.1 Methods: Roundwood Transport

The forest industry value chain starts in the forest, where roundwood is harvested. Although most of the transport-related calculations in this section are based on information from the Census Bureau Commodity Flow Survey (CFS; US Census Bureau 2023), this is not possible for roundwood transport because the CFS does not cover the "forestry and logging" industry (additional information is available in Appendix D). Instead, emissions associated with roundwood transport are calculated using the assumptions and values shown in Table 7.1.

Table 7.1. Assumptions and Values Used to Calculate Roundwood Transport Emissions

Variable	Value	Basis
Roundwood transported (includes firewood)	1990, 445 million metric tons 2005, 408 million metric tons 2020, 376 million metric tons	FAOSTAT 2023, converted from cubic meters using a factor of 0.87 metric ton per cubic meter (from FAO 2020, Table 2.3.1, average of conifer and non-conifer)
Haul mode	Diesel truck	Assumed
Haul distance	100 km one-way (round trip emissions assumed to be 1.75 times one-way emissions to account for empty return to the forest)	Oneil and Puettmann 2017; Milota 2020; Puettmann et al. 2020c; Bergman and Bowe 2008; Barrett et al. 2017; Conrad 2018; 2023
Fuel economy	2.56 km/l (6.2 mpg) in 2020. Lower to 2.4 km/l for 2005 and 1990	Davis and Boundy, 2022, Table 2.16, converted using 129,000 Btu per gallon diesel
Average load	20 metric tons (22 short tons)	LaGore 2020; Barrett et al. 2017; Conrad 2018; 2021
Diesel emission factor	2.7 kg CO_2/liter (10.21 kg CO_2 per gallon)	US EPA 2014c, CO_2 only, does not include upstream emissions associated with fuel production
Emissions per km	1.02 kg CO_2/km in 2020, 1.12 kg CO_2/km in 2005 and 1990	Calculated from values above
Emissions per metric ton-km based on 20 metric ton load and one-way haul distance	0.0512, 0.0562 and 0.0560 kg CO_2 per metric ton-km for 1990, 2005, and 2020 respectively	Calculated from values above

7.2 Methods: Product Transport

Every 5 years (years ending in 2 or 7) a congressionally mandated survey, the CFS, is distributed under the auspices the US Department of Transportation and the US Department of Commerce. The survey collects data on commodity shipments from a sample of establishments in specific industries (identified by NAICS code). Commodities are identified by Standard Classification of Transported Goods (SCTG) codes. The data in CFS reports are used in this report for estimating transport-related emissions for wood and paper products, both primary products and converted products. Detailed information on the CFS and its use in this report is available in Appendix D.

The products of interest to the US forest products are assigned SCTG codes 26, 27, 28, and 29 and are described in Table 7.2. A complete list of SCTG codes is shown in Appendix D.

Table 7.2. SCTG Codes for Primary and Converted Wood and Paper Products

SCTG Code	Description
26	Wood Products 26100 Wood chips or particles 26211 Lumber, treated 26212 Lumber, untreated 26221 Wood continuously shaped along any of its edges or faces 26222 Shingles and shakes 26310 Veneer sheets and sheets for plywood 26320 Particle board, fiberboard, and similar board of wood or other ligneous materials 26330 Plywood, veneered panels, and similar laminated wood including door skins 26401 Windows, doors, and frames and thresholds 26409 Other builders' joinery and carpentry of wood, not elsewhere classified [illegible]
27	Pulp, Newsprint, Paper, and Paperboard 27110 Mechanical wood pulp 27120 Non-dissolving grades of soda or sulfate chemical wood pulp 27191 Dissolving grades of chemical wood pulp 27199 Other pulp of fibrous cellulosic materials, not elsewhere classified (including recycled pulp) 27200 Newsprint in large rolls or sheets 27311 Uncoated paper for writing, printing, or other graphic purposes, in large rolls or sheets 27312 Toilet or facial tissue stock, towel or napkin stock, and similar paper stock used for household or sanitary purposes, in large rolls or sheets 27319 Other uncoated paper in large rolls or sheets, not elsewhere classified 27320 Uncoated paperboard in large rolls or sheets 27410 Paper, coated, impregnated, treated, or worked, in large rolls or sheets

(Continued on next page.)

Table 7.2. Continued

SCTG Code	Description
28	Paper or Paperboard Articles 28010 Toilet paper, facial tissues, towels, tampons, sanitary napkins, disposable diapers, and similar articles of paper for household, sanitary, or hospital use, and paper articles of apparel 28021 Sacks and bags of paper, paperboard, cellulose wadding, or cellulose fiber webs 28029 Other packing containers of paper, paperboard, cellulose wadding, or webs of cellulose fibers, not elsewhere classified 28091 Wallpaper and similar wall coverings 28092 Envelopes, letter cards, plain postcards and correspondence cards, and boxed sets of paper stationery 28099 Other paper or paperboard articles, not elsewhere classified (except blank books, office pads, and forms, see 2999x)
29	Printed Products 29100 Printed books, brochures, leaflets, and similar printed products (except advertising materials including catalogs, see 29300; atlases and music books, see 29999) 29210 Newspapers 29220 Journals and periodicals 29300 Advertising material, commercial or trade catalogs, and similar printed products, including flyers 29910 Printed or illustrated postcards, messages, or announcements, and printed cards bearing personal greetings 29991 Manifold business-forms and interleaved carbon-sets 29999 Other printed products, not elsewhere classified, including blank books, binders, and albums

The reported 2017 metric ton-km values for the aforementioned SCTG codes are shown in Table 7.3. Emissions are calculated by multiplying the metric ton-km values by emission factors derived from multiple sources. The emission factors used for heavy-duty trucks are shown in Table 7.1. Emission factors for rail were obtained from the US Department of Transportation based on reported fuel efficiencies of 332, 414, and 487 short ton-miles per gallon of fuel consumed in 1990, 2005, and 2020, respectively (US DOT 2024). Using emission factors for diesel fuel (see Table 7.1), these values can be converted to emission factors of 0.0211, 0.0169, and 0.0144 kg CO_2 per metric ton-km, respectively. Assuming that all transport of products was by truck or rail, these factors for trucking and rail transport were weighted (based on metric ton-km for each mode of transport) to derive an overall emission factor for each commodity (US Census Bureau 2023). The weighted factors are shown in Table 7.4.

Table 7.3 Metric ton-km Statistics for Wood and Paper Products [*Source*: US Census Bureau 2023]

NAICS Code	Wood Products SCTG 26	Pulp, Newsprint, Paper, and Paperboard SCTG 27	Paper or Paperboard Articles SCTG 28	Printed Products SCTG 29
	Metric ton-km (Millions)			
321 Wood product manufacturing	98,496	155	77 [b]	77 [b]
322 Paper manufacturing	404	88,175	25,637	802
323 Printing and related support	0 [c]	774	407	10,976
423 Merchant wholesalers, durable goods	23,133	5,082 [a]	2,584	397
424 Merchant wholesalers, nondurable goods	0 [c]	5,082	6,846	1,657
Other	7,547	22,457	2,852	
Total	129,580	121,725	38,405	16,233

[a] Assumed to be the same as NAICS code 424
[b] Assumed one-half of value for SCTG 27
[c] Assumed to be the same as paperboard articles SCTG 28

Table 7.4 Emission Factors for a Mix of Truck and Rail Transport for Forest Products

SCTG Commodity Code – Description	Emission Factor for a Mix of Truck and Rail Transport ($kg\ CO_2$ per metric ton-km) 1990	2005	2020
SCTG 26 – Wood products	0.0515	0.0511	0.0464
SCTG 27 – Pulp, newsprint, paper, and paperboard	0.0501	0.0495	0.0449
SCTG 28 – Paper or paperboard articles	0.0546	0.0546	0.0497
SCTG 29 – Printed products	0.0560	0.0562	0.0512

Emissions associated with transporting pellets were estimated separately because they are not included in the list of commodities in SCTG 26, wood products (see Table 7.2). Pellet production in 2005 was estimated to be 1 million metric tons (from Figure 1 in Abt et al. (2014)) while pellet production in 2019 (used to represent 2020) was 8.837 million metric tons (Alderman 2022). Pellet production was assumed to be zero in 1990. Approximately 70% of pellets were exported in 2019, primarily to Europe (Alderman 2022). This was assumed to be true for 1990, 2005, and 2020. Based on this information, all pellets were assumed to be hauled by a combination of truck and rail, and then 70% were assumed to be shipped by ocean bulk carriers to Europe. Over land haul distance was set at 563 km (350 mi), the average haul distance per shipment for wood products (SCTG 26) in the CFS (US Census Bureau 2023). The emission factors used for over land shipments of pellets were calculated using the same information as used for wood products (including the ratio of truck to rail transport) and shown in Table 7.4. Additional emissions associated with transport by bulk ocean carrier were calculated for pellets exported to Europe. The distance was assumed to be 6,500 km. The International Marine Organization has published emission factors for bulk ocean freight of 0.0041 kg CO_2 per metric ton-km (6.9 g CO_2 per short ton-nautical mile) in 2008 and 0.0066 kg CO_2 per metric ton-km (11.08 g CO_2 per short ton-nautical mile) in 2018 (IMO 2020). The 2018 value was used to represent 2020 while the 2008 value was used for 1990 and 2005.

7.3 Methods: Transport of Coproducts Produced by Wood Products Mills

Wood products mills produce significant quantities of chips, sawdust, etc. that are used as raw materials by other facilities. These materials, however, are included in the list of wood products in SCTG 26 in Table 7.2, and the emissions associated with their transport are therefore included in the total for wood products.

7.4 Methods: Transport of Nonfiber, Nonfuel Inputs

The production of forest products requires a range of inputs besides wood fiber. The production and transport of these materials result in GHG emissions. Emissions associated with the production of nonfiber, nonfuel inputs are estimated elsewhere in this report. In this section, transport-related emissions attributable to these inputs are examined.

Quantities of nonfiber, nonfuel inputs were estimated from life cycle assessment (LCA) studies, identified in Table 7.5. For paper and paperboard, studies conducted by AF&PA (in collaboration with other groups) were used. For wood products, studies published by CORRIM were used. For wood products other than lumber, the value for panels was used. Conversions were necessary in some cases to derive as-shipped weights. The results and data sources are shown in Table 7.5.

Table 7.5 Factors to Calculate Quantities of Nonfiber, Nonfuel Inputs to Manufacturing US Primary Forest Products

Product	Value	Units	Source
Containerboard	0.053	metric tons per metric ton product	AF&PA and CPA LCA of containerboard (NCASI 2017)
Uncoated freesheet	0.277	metric tons per metric ton product	AF&PA LCA of printing and writing papers (NCASI 2010)
Coated freesheet	0.375	metric tons per metric ton product	AF&PA LCA of printing and writing papers (NCASI 2010)
Uncoated mechanical	0.081	metric tons per metric ton product	AF&PA LCA of printing and writing papers (NCASI 2010)
Coated mechanical	0.362	metric tons per metric ton product	AF&PA LCA of printing and writing papers (NCASI 2010)
Lumber	0.437	kg per cubic meter product	Average of CORRIM studies of Pacific Northwest and Southeast lumber production (Milota 2020a; 2020b)
Plywood	18.17	kg per cubic meter product	Average of CORRIM studies of Pacific Northwest and Southeast plywood production and Southeast oriented strand board production (Puettmann et al. 2020a; 2020b; 2020c)

The grades of pulp, paper, and paperboard were assigned to the appropriate factors in Table 7.5 and multiplied by the quantities produced in 1990, 2005, and 2020, using statistics from AF&PA (AF&PA 2020; 2021a) to calculate the quantities of nonfiber, nonfuel inputs. The same approach was used for wood products. All non-lumber products were assigned the factors for panels. Production data were obtained from FAOSTAT (FAOSTAT 2023).

For inputs to the pulp, paper, and paperboard sector, transport distances of nonfiber, nonfuel inputs were assumed to be represented by an average of commodity codes SCTG 12 (gravel and crushed stone) shipped to all users, SCTG 20 (basic chemicals) shipped to NAICS 322 (paper and paperboard mills,) and SCTG 31 (nonmetallic mineral products) also shipped to NAICS 322 (paper and paperboard mills). In the 2017 CFS, the average of these was 207 km. For wood products, transport distances for nonfiber, nonfuel inputs were represented by SCTG 20 (basic chemicals) shipped to manufacturing facilities (NAICS 31–33). In the 2017 CFS this distance was 365 km. All transport was assumed to be by truck with a cargo load of 20 metric tons, allowing the use of emission factors in Table 7.1.

7.5 Methods: Transport of Used Products

After use, products are transported to recovery and/or disposal operations. Data on the quantities requiring transport were obtained from US EPA (US EPA 2020e) and are shown in Table 7.6. The haul distance used was 362 km (225 mi), the value reported for 2017 in the CFS for SCTG 4112, waste and scrap of paper and paperboard. Trucks were assumed to be the mode of transport and the truck emission factors for 20 metric ton loads in Table 7.1 were used.

Table 7.6. Generation Rates, Before Recovery, of Used Paper, Paperboard, and Wood Products in the US [*Source*: US EPA (2020e)]

	Million Metric Tons Generated		
	1990	2005	2018
Paper and paperboard	66,118,182	77,127,273	61,263,636
Wood	11,100,000	13,445,455	16,445,455
Total	77,218,182	90,572,727	77,709,091

7.6 Transport Emission Results and Discussion

The transport-related emissions, except those related to fossil fuel transport (which are addressed in Section 4.3 of this report) are shown in Table 7.7. These emissions were 24.1, 24.4, and 20.1 million metric tons CO_2eq. in 1990, 2005, and 2020 respectively. Approximately 73% of these emissions are associated with transport of paper and wood products. This reflects all products along the value chain, so it accounts for multiple sequential shipments of the same fiber as it is transformed from raw material into final products at the retail level. About 17% of transport-related emissions are attributable to the transport of roundwood. Another 7% of emissions are from the transport of used products to recycling or disposal.

Table 7.7. Emissions Attributable to Transport in the US Forest Products Industry Value Chain

	Million Metric Tons CO_2		
	1990	2005	2020
Raw Materials			
All roundwood, including firewood	4.37	4.02	3.36
Recovered fiber (shown as used products below in table)			
Paper and paperboard nonfiber, nonfuel inputs	0.17	0.18	0.12
Wood product nonfiber, nonfuel inputs	0.01	0.02	0.01
Products			
SCTG 26 Wood products	9.07	7.52	6.01
SCTG 27 Paper and paperboard	5.20	5.94	5.47
SCTG 28 Paper articles	1.49	2.31	1.91
SCTG 29 Printed materials	1.58	1.84	0.83
Wood-based pellets	0	0.06	0.40
Used Products	1.57	1.84	1.44
Total	23.5	23.7	19.5
Fraction of 1990 total emissions	1.00	1.01	0.83
Fraction of 2005 total emissions		1.00	0.82

The estimates in Table 7.3 for 1990 and 2005 are significantly different from those in Heath et al. (2010) and NCASI (2008). The differences in estimates of emissions from roundwood transport are primarily due to Heath et al. (2010) and NCASI (2008) having used CFS data for these estimates. An examination of the CFS roundwood transport data revealed, however, that they are not appropriate for characterizing roundwood transport in the US forest industry value chain as they do not include transport by forestry and logging companies. The new estimates, shown in Table 7.7, are therefore a better representation of industry practices than contained in Heath et al. (2010) and NCASI (2008).

Another significant difference between the estimates in Table 7.7 for 1990 and 2005 and those in Heath et al. (2010) and NCASI (2008) is for product shipment. The previous estimates did not attempt to account for multiple, sequential shipments along the value chain. In this study, the data from the 2017 CFS have been used for all commodity codes describing forest products. The CFS data include multiple, sequential shipments. The estimates in Table 7.7, therefore, are a better representation of product transport in the industry's value chain than the estimates in Heath et al. (2010) and NCASI (2008). Other differences are due to the new estimates being based on updated methods and data.

Transport-related emissions in 2020 were 17% lower than in 1990 and 18% lower than in 2005. The emissions reductions were due to a combination of reduced industry production and improvements in fuel efficiencies for trucking and rail transport.

8.0 EMISSIONS ASSOCIATED WITH PRODUCT USE

Few forest products emit GHGs during use, or cause GHGs to be emitted during use. Paper and paperboard do not require energy to use so are not responsible for emissions during the use phase. Wood products are used in structures that must be heated and cooled, but these emissions are normally attributed to the heating and cooling systems or the structure as a whole rather than to the materials used to assemble the structures. Forest-based fuels, however, release methane and nitrous oxide when burned, and these are included in Scope 3 inventories. Biogenic CO_2 emissions are not included as emissions because these flows to the atmosphere are captured in the production accounting used to calculate changes in biogenic carbon stocks, discussed earlier in this report.

8.1 Methods

The amounts of wood energy used in the US up to 2017 are reported by the US Forest Service (Howard and Liang 2019). The US Forest Service assigns domestic wood energy consumption to residential, commercial, industrial, and electricity utility sectors. The consumption in the industrial sector is due almost entirely to the use of wood-derived materials for fuel at forest products manufacturing facilities. Emissions from the use of these materials are accounted for elsewhere in this profile and are excluded from the estimates here.

In addition to domestic consumption, wood pellet exports have become increasingly important. For this report, pellet export data since 2012 were obtained from the US Department of Agriculture Global Agricultural Trade System (GATS) (US DA 2023). It was assumed that these exports were zero in 1990 and increased linearly until the first GATS data in 2012. Figure 8.1 shows the amount of wood energy burned in nonindustrial sectors and exported in wood pellets from 1990 to 2017. The residential sector consumes about one-half of nonindustrial wood energy. Recent growth sectors have been electric utilities and pellet exports. Exported pellets are primarily used to produce electricity in the importing countries.

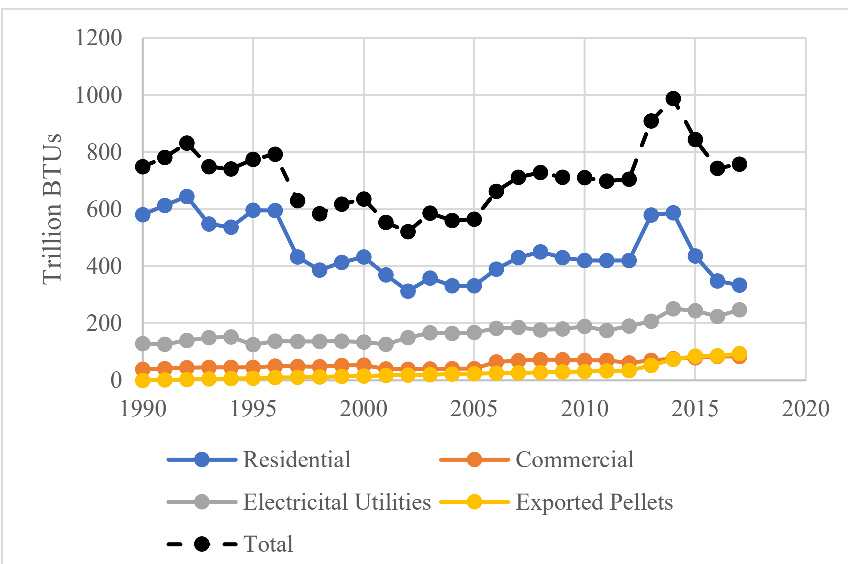


Figure 8.1. Consumption of US-Sourced, Wood-Based Fuels by Nonindustrial Sectors

European Integrated Pollution Prevention and Control Bureau (IPPC) emission factors (Gomez et al. 2006) were used to estimate methane and nitrous oxide emissions from all non-industry sectors except the residential sector. For the residential sector, emissions are highly variable depending on combustion conditions. For this report, IPCC's methane emission factor for noncatalytic wood stoves was used (497 kg CH_4/Tg energy input). For all other sectors, IPCC uses a value of 11 kg CH_4/Tg energy input. IPCC has no value for nitrous oxide from residential consumption of wood for energy; therefore, IPCC's factor for other sectors was used (7 kg N_2O/Tg energy input).

8.2 Results

The nonindustrial GHG emissions associated with the use of wood-derived fuels produced in the US are shown in Figure 8.2. These emissions are dominated by methane emissions from residential use of wood for energy. Residential firewood accounted for 84% of nonindustrial wood energy emissions in 2017. Changes since 1990 are quantified in Table 8.1. The changes in residential use of fuel wood are related, in part, to fuel oil prices, as can be seen by comparing Figure 8.2 with Figure 8.3, which shows changes in fuel oil prices since 1990 in both nominal and real (adjusted for inflation) terms (EIA 2023b).

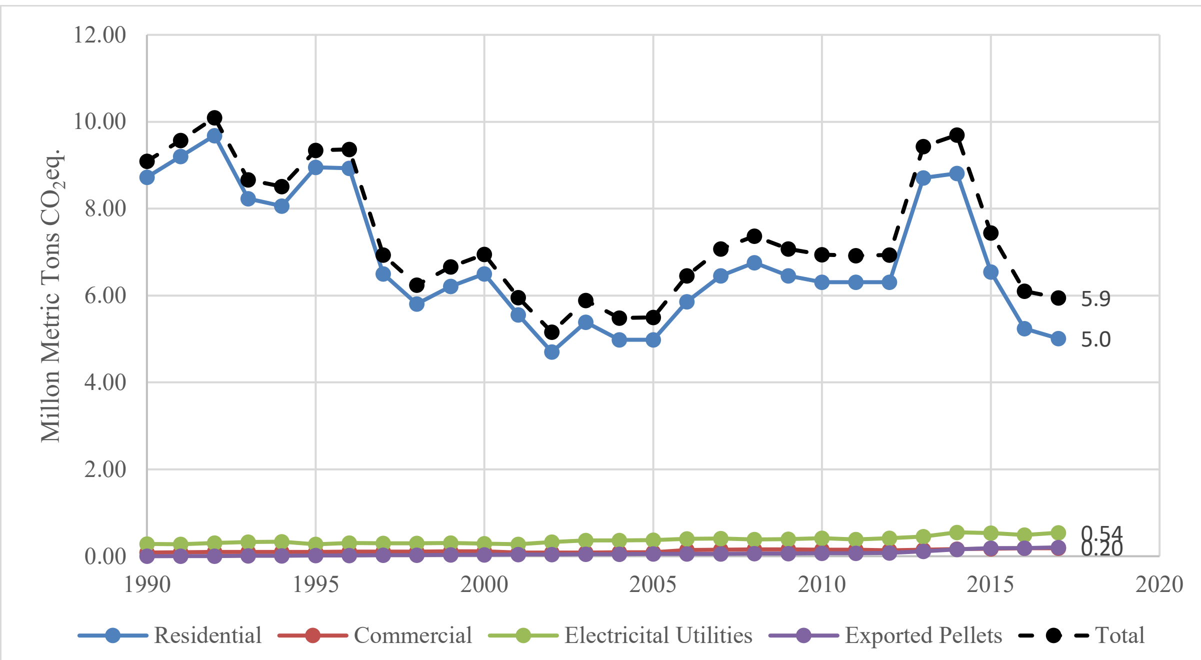


Figure 8.2. GHG Emissions from US-Produced Wood for Energy in Nonindustrial Sectors

Table 8.1. Changes in Emissions from US Wood-Derived Fuels Since 1990

Year	Million Metric Tons CO_2eq.	Fraction of 1990	Fraction of 2005
1990	9.1	1.00	
2005	5.5	0.60	1.00
2020[a]	5.9	0.65	1.08

[a] 2017 data, the most recent available, are used to reflect 2020 emissions.

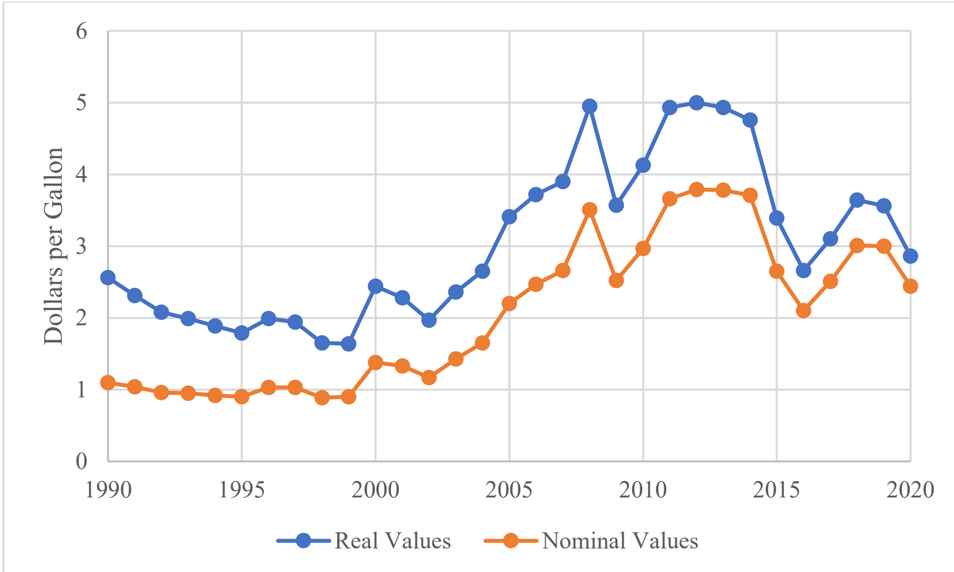


Figure 8.3. US Heating Oil Prices

9.0 EMISSIONS FROM MANAGEMENT AND DISPOSAL OF INDUSTRY WASTES

The forest products industry produces wastewater and solid wastes. Most wastewater generated by the industry is treated on-site in aerobic treatment systems where methane is unlikely to be generated in significant quantities. In some cases, however, these systems may contain anaerobic zones and, therefore, have the potential to produce methane. In addition, solid wastes from the industry are often disposed of in landfills, where methane can be produced. In this section, these waste management–related emissions are estimated. The data do not allow an accurate differentiation between waste management operations owned by the industry compared to those owned by other entities. For this profile, therefore, it is assumed that all waste management emissions are Scope 1.

9.1 Emissions from Landfills Receiving Mill Wastes

For the 2008 profile (NCASI 2008; Heath et al. 2010), the quantities of pulp and paper mill wastewater treatment residuals (WWTR) and wood product mill solid wastes were estimated from periodic survey data. Survey data were also used to estimate the fraction of waste landfilled. The data were used to derive an industry-wide estimate of the quantities of solid waste landfilled each year from 1970 through 2005. The decomposition of landfilled wastes was modeled using a first order decay equation, along with appropriate k and L_o values. From this, 1990 and 2005 forest products industry landfill methane emissions were calculated. Separate estimates were made for pulp and paper mills and wood products plants.

Since then, more options have become available for making these estimates, particularly for pulp and paper mills landfill emissions. In this updated profile, the original approach has been applied (in an updated form), using updated data and parameters. In addition, for WWTR, several other approaches are also used. Finally, because US EPA's GHGRP includes several wastes not included in the earlier profile, this updated profile is expanded to include these additional wastes.

9.1.1 Pulp and Paper Mills

NCASI has estimated methane emissions attributable to four different wastes that may be disposed of in mill landfills: WWTR, boiler ash, recovery area wastes, and other wastes. In earlier profiles, only WWTR were considered, but US EPA's GHGRP includes calculation parameters for all four types of wastes. Therefore, all are included here. In addition, the GHGRP includes parameter values for combined wastes from pulp and paper mills, thus separate estimates are presented assuming that all landfilled wastes are co-disposed.

9.1.1.1 Wastewater Treatment Residuals

NCASI has identified three methods to estimate methane emissions from landfills receiving WWTR from pulp and paper mills:

1. Adapt estimates developed by US EPA for its annual inventory (described in US EPA 2022b);
2. Extrapolate emissions reported by facilities under US EPA's GHGRP;
3. Use the same approach as used in the earlier profile (as specified in the GHGRP) but incorporate updated data and parameters.

US EPA's Annual Inventory

US EPA develops estimates of methane emissions from landfills in the pulp/paper and food/beverage sectors (US EPA 2022b). US EPA states that it used a factor of 0.05 metric tons of landfilled waste per metric ton of pulp and paper production to estimate annual disposal amounts and presents estimates of total waste generation and waste landfilled (US EPA 2022b)[13]. Degradation was modeled using a first order decay model with a value for k of 0.06 yr^{-1} and degradable organic carbon (DOC) of 0.15 (L_o of 49 m^3 methane/metric ton waste), wet basis[14]. A comparable approach, using different parameter values, was used for the food/beverage sector. US EPA does not show the emissions results separately for the two sectors, but the amounts of waste disposed are shown separately. NCASI has estimated the pulp and paper sector's contribution to these emissions by multiplying the total emissions from the two sectors by the ratio of pulp and paper waste amounts to total waste amounts from the two sectors. Values from 2005 to 2026 were interpolated. The results are shown in Table 9.1. Using this approach, US EPA's annual inventory calculations suggest that methane emissions attributable to WWTR increased from 289,000 metric tons in 1990 to 363,000 metric tons in 2009 before declining to 355,000 metric tons in 2020.

[13] Table A-219 in the annexes to US EPA's 1990–2020 (US EPA 2022b) inventory indicates that the pulp and paper sector produced 121 million metric tons of waste in 2020. Using a factor of 0.05 would suggest that pulp and paper production equaled 2.4 billion metric tons, which is more than an order of magnitude too large. The table, however, also shows US EPA's estimate of the metric tons landfilled, which in 2020 equaled 5.9 million metric tons (assumed wet). This is much higher than NCASI estimates (see discussion later in this section) but far more reasonable, given industry production and other estimates.

[14] The text in US EPA's 1990–2020 GHG inventory, Annex Page A-455 (US EPA 2022b) implies that a dissolved organic carbon (DOC) of 0.15 corresponds to a Lo of 49 m^3/wet metric ton. However, Lo is a measure of the ultimate methane production potential, which is related to the decomposable fraction of biogenic carbon (DOCf) rather than to the total amount of biogenic carbon (DOC). DOC and Lo can be used together to calculate DOCf. A quick calculation, for instance, can show that a Lo of 49 with a DOC of 0.15 is equivalent to a DOCf of 0.328 (assuming a density of methane of 0.67 kg per m^3).

Table 9.1. Adaptation of US EPA Annual Inventory Calculations to Estimate Methane Emissions Attributable to WWTR in Landfills

Year	Emissions from the Pulp-Paper and Beverage-Food Sectors Combined, Metric Tons Methane	Fraction of Waste from the Pulp-Paper Sector	Estimated Emissions from the Pulp and Paper Sector, Metric Tons Methane
1990	436,000	0.67	289,000
2005	574,000	0.63	363,000
2020	674,000	0.53	355,000

Greenhouse Gas Reporting Program

Since 2011, many pulp and paper facilities have been required to report annual GHG emissions to US EPA under the GHGRP (US EPA 2023b). In reporting year 2021, 103 pulp and paper facilities reported to the program, 90 of which reported emissions for industrial landfills (based on the FLIGHT tool available at US EPA 2023a). The calculations involve the same first order decay approach described earlier. US EPA publishes default values for the parameters needed with this approach to calculate annual emissions from current and past discards of waste in landfills.

The annual emissions reported under the GHGRP for landfill emissions at pulp and paper mills (NAICS codes 322110, 322121, 322122, and 322130) are shown in Table 9.2. These are approximately one-half of those estimated earlier in this report (derived from the national inventory). In addition to the quantity of emissions being different, the decline in emissions from 2011 to 2020 is also much larger than indicated by the US EPA national inventory calculations shown in Table 9.1. The differences in emissions estimates are examined in more detail later in this report.

Table 9.2. Methane Emissions from Pulp and Paper Mill Landfills Reported in 2020 Under the GHGRP

Year	Metric Tons Methane
2011	232,000
2012	224,000
2013	204,000
2014	200,000
2015	200,000
2016	188,000
2017	172,000
2018	168,000
2019	168,000
2020	164,000
2021	160,000

NCASI Estimates Based on First Order Decay Model

In the 2008 profile, NCASI used published and unpublished survey data and a first order decay model to estimate emissions from landfills receiving WWTR. For this update, the same approach has been applied using updated data and research. As noted previously, this modeling approach is the same as that published by IPCC for national inventories (see Pipatti et al. 2006) and that used by US EPA for estimating industrial landfill emissions in its national inventories (see US EPA 2022b) and in the US EPA's GHGRP (US EPA 2023a).

NCASI has data allowing estimates of WWTR generation rates per metric ton of production back to 1990. This is the starting date used in the GHGRP for landfill methane calculations. These have been multiplied by the corresponding industry production to estimate annual quantities of the residuals, as called for in the GHGRP. Survey data are also available indicating the fraction of residuals that are landfilled[15]. The sources of this information are described herein.

US EPA's GHGRP and US EPA's annual inventory calculations do not use the same default DOC and k values for WWTR. The values used in the two programs are shown in Table 9.3.

Table 9.3. Pulp and Paper Mill Wastewater Treatment Residual Parameter Values for First Order Decay Model

Parameter	Units	GHGRP Subpart TT (US EPA 2023a)	US EPA 2020 GHG Inventory Annexes, p. A-455 (US EPA 2022b)
DOC	kg organic C/kg wet waste	0.12	0.15
DOCf	Fraction of DOC that will degrade in landfill	0.5 (equivalent to a Lo of 59.7 m^3 methane per wet metric ton waste)	0.328 (calculated from Lo of 49 m^3 methane per wet metric ton waste)
k	yr^{-1}	0.02 (dry climate) 0.04 (moderate climate) 0.06 (wet climate)	0.06
MCF	Fraction of degradable carbon that is subject to anaerobic conditions	1	1
F	Fraction by volume of CH_4 in landfill gas, as generated	0.5	0.5
OX	Fraction of CH_4 that is oxidized before it is released to the atmosphere	0.1 (values ranging from 0.0 to 0.35 are listed with the choice depending on site-specific circumstances)	0.1

DOC = Degradable Organic Carbon; DOCf = Fraction of DOC that is degradable under anaerobic conditions; k = First order decay constant; MCF = Methane Correction Factor; F = Fraction of DOCf converted to methane; OX = Fraction of methane that is oxidized as it passes through the landfill cover

[15]The fraction of WWTR being landfilled has declined from over 80% in the 1970s to under 40% in 2020. The balance of WWTR is used beneficially, for instance, as soil amendment or as a source of energy. In this profile, only landfill emissions are estimated because the other management methods produce far fewer GHGs.

The amounts of WWTR generated and landfilled were estimated from NCASI survey data to construct a series of annual values starting in 1960 and ending in 2020. This period was used as it is the period used in the GHGRP. For the period before 1988, mean generation rates reported in Miner and Unwin (1991) were used. Data for subsequent years were from periodic industry surveys. The five estimates from these sources ranged from 40.5 kg per metric ton in 1975 to 47.55 kg per metric ton in 2016. The average of the values, 44 dry kg per metric ton of production, was used to reflect all years. The information from these sources is in dry weight, but the DOC values used by US EPA are on a wet waste basis (see Appendix E for more detail on the implications of using a wet basis vs. a dry basis when interpreting DOC values). Therefore, the generation rates were converted to a wet basis using a solids content of 31%. This value represents the facility-weighted average reported in NCASI (2019), Table 3.1[16]. Generation rates were multiplied by annual production of paper, paperboard, and market pulp to derive estimates of metric tons of WWTR each year from 1960 to 2020.

The fraction of wastewater treatment plant residuals that is landfilled has changed significantly over time. Data from the sources discussed previously indicate that the fraction landfilled was 86%, 70%, 51%, and 36% in 1979, 1988, 1995, and 2016, respectively (Miner and Unwin 1991; NCASI 1999a; NCASI 2019). To estimate the annual amounts of landfilled residuals, the 1979 value (86%) was used for 1960 to 1979, the 2016 value (36%) was used for 2016 to 2020, and the values for 1980 to 2017 were interpolated using the 1979, 1988, 1995, and 2016 data points. Methane releases for each year from 1990 were estimated using the calculation approach in Subpart TT of the GHGRP (US EPA 2023a). The moderate climate value for k was used. For comparison, the parameter values used in US EPA's inventory calculations were also used. The results are shown in Table 9.4.

Table 9.4. Methane Releases Attributed to Landfilling Pulp and Paper Mill WWTR Calculated Using the First Order Decay Model with Different Sets of US EPA Defaults (see Table 9.3)

	Methane Emissions, Metric Tons	
Year	First Order Decay Model Using GHGRP Subpart TT Parameter Values (US EPA 2023a)[a]	First Order Decay Model Using US EPA GHG Inventory Parameter Values (US EPA 2022b)
1990	175,000	179,000
1995	190,000	188,000
2000	199,000	192,000
2005	202,000	189,000
2010	200,000	181,000
2015	194,000	170,000
2020	186,000	158,000

[a] Elsewhere in this report, where a single estimate is required, this approach is used to provide the estimate.

Comparison of Approaches

[16] The number of mills with each type of waste and dewatering device was multiplied by the percent solids for that combination. The total was summed and divided by the number of mills. Dryers were not included.

The results of the approaches described earlier are compared in Figure 9.1. There are large differences between some of the estimates, the reasons for which are explored herein.

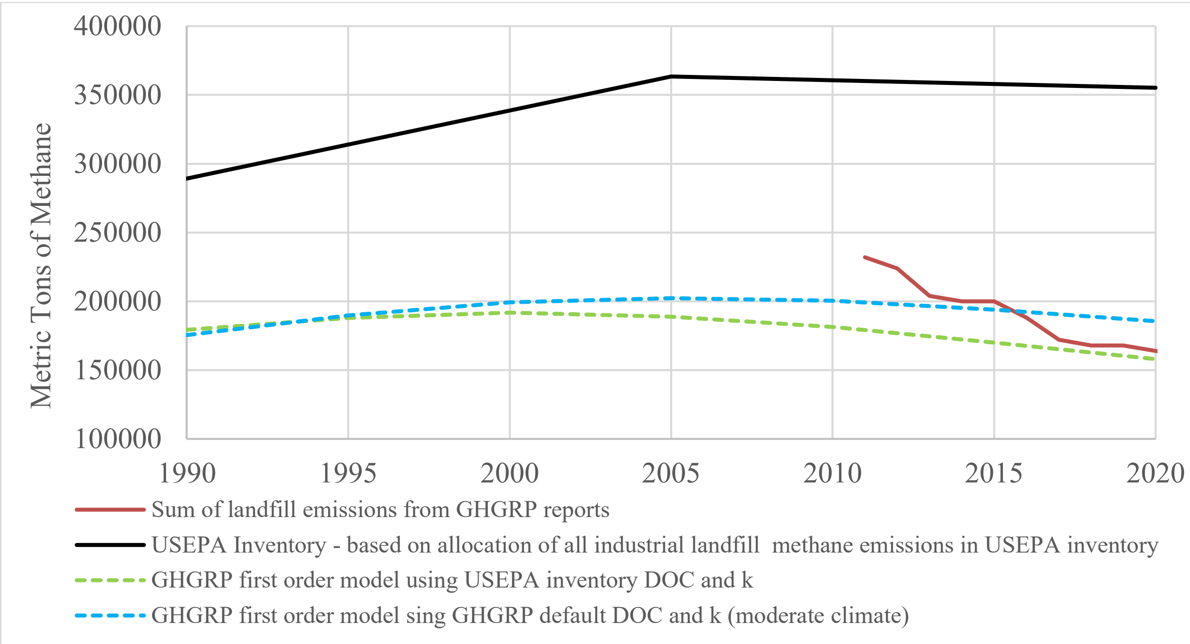


Figure 9.1. Methane Emissions Attributable to Landfilling Pulp and Paper Mill WWTR: A Comparison of Methods

Data from the GHGRP are available only since 2011. Over the 2011 to 2020 period, they are similar to NCASI's estimates but suggest much more rapid reductions. There are several factors to consider in interpreting these data.

The companies reporting under the GHGRP are not required to use default values for k, DOC, DOCf, or OX if they have more appropriate values. Indeed, examination of several GHGRP reports confirms that mills are not always using default values but rather values the companies find more appropriate. It is unsurprising, therefore, that the sum of these reports would not be the same as an industry estimate based on default values.

It is also important to consider that the GHGRP only includes materials placed in landfills owned by the reporting company. Landfills owned by third parties are not included in the reports from pulp and paper mills and are not easily identified in the GHGRP data. The three other approaches shown in Figure 9.1 are intended to include all landfills, regardless of ownership.

Another important consideration in interpreting GHGRP data is the approach used to estimate historical deposits into landfills. Most mills reporting under the GHGRP do not have data dating to 1960 (the starting year for GHGRP calculations) and, therefore, have had to use GHGRP-designated methods to estimate many prior years of landfill deposits. Examination of several mill reports suggests that the GHGRP methods can yield estimates for previous years that are larger than current deposits, sometimes by a significant amount. This may be producing overestimates of emissions in the earlier years of the reporting program, which would increase apparent reductions in emissions since the reporting began. For all these reasons, GHGRP reporting data on landfill methane emissions are not used in this profile for estimating these emissions.

Estimates produced by allocating industrial landfill emissions reported in US EPA's annual inventories are consistently much higher than those in the GHGRP as well as those estimated by NCASI when using the same parameter values as US EPA uses in GHG inventory calculations. Some of the difference is likely related to differences in estimates of the amounts of WWTR landfilled over time. US EPA's estimates are based on wet weight whereas NCASI uses dry weight. Therefore, a comparison of waste quantities requires knowing the solids content of the waste being landfilled. Based on survey data described earlier in this report, NCASI has used 31% solids for WWTR in its calculations. The data in Figure 9.2 are shown for US EPA estimates of landfilled quantities if WWTR are at 30%, 40%, and 50% solids. The figure shows that even at 30% solids, NCASI's data indicate lower quantities in landfills since 2005. At higher solids contents, the difference becomes larger. In addition, the figure shows that while NCASI data indicate large reductions in amounts landfilled over time, US EPA's estimates have remained relatively constant. These differences would contribute to NCASI-modeled estimates of methane emissions being lower than those shown in US EPA's inventory, especially in recent inventory years.

Another difference is that for its national inventory, US EPA begins the calculations in 1940 instead of 1960. NCASI has extended its calculations back to 1940 by lowering landfilled quantities according to changes in real gross domestic product, the method used by US EPA. Using this approach, starting in 1940 increases 1990 emissions by only about 5%, with the difference decreasing over time, leading to only a 1% difference in 2020. Thus, this difference does not explain the disparity between NCASI's and US EPA's estimates.

Even if all industrial waste identified in US EPA's calculations is assigned to the pulp and paper industry, NCASI's estimates, using US EPA methods and parameter values, remain 20 to 25% below those published by US EPA for methane from all industrial landfills. The reasons for the differences between NCASI's and US EPA's estimates, therefore, remain unclear.

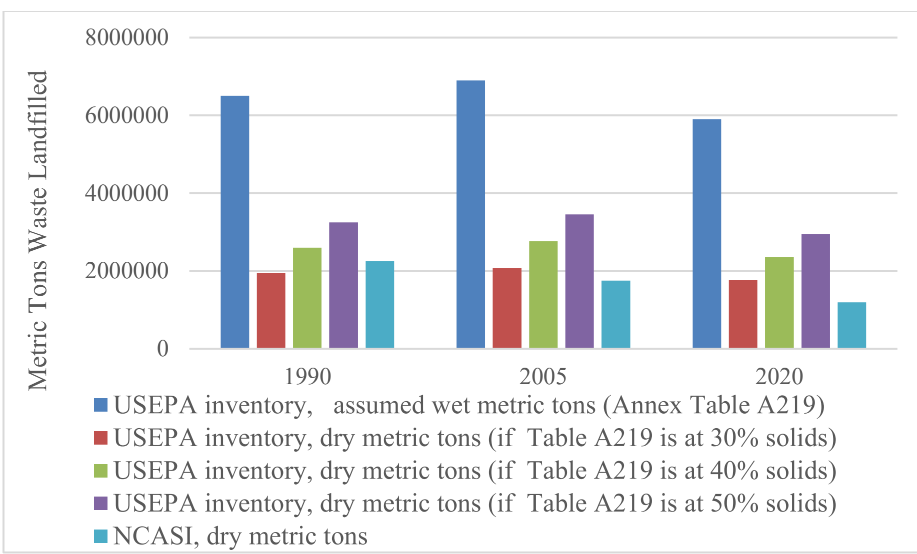


Figure 9.2. WWTR Quantities Landfilled Estimated by US EPA and NCASI

The 2020 reductions from 1990 and 2005 derived from these different calculation approaches are shown in Table 9.5. NCASI's modeling using GHGRP defaults for moderate climate (used in this study as a base case when a single estimate is required) indicates that 2020 methane emissions attributable to landfill disposal of WWTR were 6% higher than 1990 emissions and 8% lower than 2005 values.

Table 9.5. Methane Emissions in 2020 Attributable to WWTR and Change Since 1990

Estimation Method	2020 Emissions, Metric Ton Methane	2020 as Fraction of 1990	2020 as Fraction of 2005
Allocate industrial landfill emissions in US EPA's 2020 inventory report (1991 values used for 1990)	355,000	1.23	0.98
NCASI modeling using the same parameter values as used in US EPA's 2020 inventory calculations (DOC = 0.15, DOCf = 0.328, k = 0.06)	158,000	0.88	0.84
NCASI modeling using the same parameter values as used in the GHGRP program (moderate climate) (DOC = 0.12, DOCf = 0.5, k = 0.04)[a]	186,000	1.06	0.92

[a] In this update, when a single estimate is required, these are the results used.

9.1.1.2 Boiler Ash

Boiler ash was not included in earlier profiles due to its low content of organic matter and, in the case of ash derived from wood and bark (hereafter referred to as simply wood ash), due to its high pH. The pH of wood ash is seldom below 10 (e.g., see Campbell 1990; NCASI 1999b), while methane production is not favored at such a high pH (e.g., see Malyan et al. 2016). Given that companies are required to report ash landfills under the GHGRP, however, methane attributable to ash (wood and coal) disposal is estimated in this profile.

Information on the quantities of ash produced by pulp and paper mills has only been collected by NCASI since the mid-1990s. These survey data indicate that from the mid-1990s until approximately 2010, ash generation rates remained relatively constant, averaging 56.55 kg per metric ton. Starting in approximately 2010, however, ash production began dropping significantly so that in 2020 the generation rate was 25.3 kg per metric ton. This trend is primarily the result of industry's reduced reliance on coal for energy. The US Department of Energy (DOE) MECS survey, for instance, indicates that the pulp and paper industry (NAICS 322) reduced coal consumption from 8 million metric tons in 2010 to 2.7 million metric tons in 2018 (EIA 2023a).

For the current profile, it was assumed that from 1960 until 2010, ash was produced at a rate of 56.55 kg per metric and from then until 2020, the rate declined according to NCASI's semiannual survey data until it reached 25.3 kg per metric ton in 2020. NCASI published information indicates that the fraction landfilled was 84%, 72%, and 75% in 1988, 1995, and 2016, respectively (NCASI 1992; 1999a; 2019). Based on the survey data, however, it is impossible to know the extent to which these values include ash that is comingled with WWTR. For the calculations here, it is assumed that ash is managed separately from WWTR. To the extent this assumption is incorrect, it would result in double counting emissions from ash, making it a conservative approach. The landfilling rates between 1988 and 2016 were interpolated. The value for 1988 was extended back to 1960, while the 2016 value was extended forward to 2020. Generation rates were multiplied by annual production of paper, paperboard, and market pulp (AF&PA 2020; 2021a) to derive estimates of metric tons of ash each year from 1960 to 2020. The default parameter values used in the GHGRP to model methane emissions from boiler ash landfills are shown in Table 9.6.

Table 9.6. Pulp and Paper Mill Boiler Ash Parameter Values for First Order Decay Model

Parameter	Units	GHGRP, Subpart TT
DOC	kg organic C/kg wet waste	0.06
DOCf	Fraction of DOC that will degrade in landfill	0.5
k	yr^{-1}	0.02 (dry climate) 0.03 (moderate climate) 0.04 (wet climate)
MCF	Fraction of degradable carbon that is subject to anaerobic conditions	1
F	Fraction by volume of CH_4 in landfill gas, as generated	0.5
OX	Fraction of CH_4 that is oxidized before it is released to the atmosphere	0.1 (values ranging from 0.0 to 0.35 are listed with the choice depending on site-specific circumstances)

The DOC values in the GHGRP Subpart TT rules are on a wet weight basis. Generation rates, however, are on a dry weight basis, requiring a factor to convert to as-disposed wet weight. Data are not available on the solids content of boiler ash disposed in pulp and paper industry landfills. Ash management methods, however, have been documented by NCASI (NCASI 1999a). NCASI collected 1995 data from 145 mills producing boiler ash. Forty-two of the mills indicated having a dewatering method for ash. Most of these were gravity drainage systems, such as drainage pads and settling ponds. Six respondents reported using mechanical dewatering devices. This information was used to infer that 71% (i.e., 103/145) of ash was handled and disposed of in a 100% dry state, 25% (i.e., 36/145) was dewatered using gravity systems like settling ponds, and 4% (i.e., 6/145) was dewatered mechanically. For this analysis, mechanical dewatering is assumed to produce a dewatered product at 90% solids (wet weight basis). A solids content of 60% (wet basis) was used to represent a reasonable value for dredged ash, representing the remaining wet ash (e.g., see EPRI 1979). The weighted average solids content of ash disposed by the industry is therefore calculated to be 90% (wet basis) in 1995. The survey was repeated in 2016 (NCASI 2019), but the results on ash management were not published. Using the same approach, however, the unpublished data for 2016 yield a weighted average solids content of disposed ash of 98%. The higher solids content reflects reduced reliance on wet scrubbing systems for particulate control on power boilers. For these calculations, it was assumed that in 1960 all ash was dewatered in ponds to 60%. The solids content used to convert the GHGRP wet basis DOC to a dry basis started at 60% in 1960, increased linearly to 90% in 1995, and again increased linearly to 98% in 2016, where it is estimated to have remained.

Using these values in the landfill model with GHGRP parameter defaults for DOC (wet basis) and k yields the results shown in Table 9.7.

Table 9.7. Methane Emissions from Boiler Ash Landfilled at Pulp, Paper, and Paperboard Mills

Year	NCASI Calculations Using GHGRP Equations – GHGRP Default DOC and k (Moderate Climate)		Fraction of 1990 value	Fraction of 2005 value
	Metric Tons CH_4	Million Metric Tons CO_2eq.[a]		
1990	40,300	1.0	1.00	
1995	45,300	1.1	1.12	
2000	49,800	1.2	1.24	
2005	53,200	1.3	1.32	1.00
2010	55,600	1.4	1.38	1.05
2015	55,700	1.4	1.38	1.05
2020	52,700	1.3	1.31	0.99

[a] GWP of 25.

9.1.1.3 Recovery Area Waste

The recovery area of kraft pulp mills can produce several waste streams, often collectively called causticizing area waters. The three main wastes are lime mud, slaker grit, and green liquor dregs. These have not been included in past attempts to estimate methane emissions from mill landfills due to their relatively small quantities, low organic content, and typically elevated pH, which is generally not conducive to methane production. Nonetheless, the GHGRP contains parameter values for estimating landfill methane emissions attributable to these materials (US EPA 2023a).

NCASI survey data allow estimates of the quantities of solid wastes, other than WWTR and ash, at several points back to 1995. In addition, NCASI 1999a and NCASI 2019 indicate that recovery area wastes represented 29% and 34% of these “other” wastes in 1995 and 2016, respectively. Using these two data sets, NCASI estimates that from 1960 to 1995, the industry generated about 21.05 kg per metric ton of recovery area wastes. This decreased to 14.25 kg per metric ton in 2002 and then varied between 14 and 16 kg per metric ton until 2016. From 2016 to 2020, generation rates increased from 16.3 to 18.35 kg per metric ton. Dry weight was converted to wet weight using a solids content of 65%, the average reported for these materials in NCASI (2019). A weighted average percent landfilled of 87.9% for dregs, lime mud, and slaker grit was developed from data in NCASI (1999a). This was the only data point available; therefore, it was used for the entire period of 1960 to 2020. The parameter values used to model methane attributable to landfilled recovery area wastes are shown in Table 9.8.

Table 9.8. Pulp and Paper Mill Recovery Area Waste Parameter Values for First Order Decay Model

Parameter	Units	GHGRP, Subpart TT
DOC	kg organic C/kg wet waste	0.025
DOCf	Fraction of DOC that will degrade in landfill	0.5
k	yr^{-1}	0.02 (dry climate) 0.03 (moderate climate) 0.04 (wet climate)
MCF	Fraction of degradable carbon that is subject to anaerobic conditions	1
F	Fraction by volume of CH_4 in landfill gas, as generated	0.5
OX	Fraction of CH_4 that is oxidized before it is released to the atmosphere	0.1 (values ranging from 0.0 to 0.35 are listed with the choice depending on site-specific circumstances)

The results are shown in Table 9.9. These estimates indicate that the amounts of methane attributable to recovery area wastes are far lower than those attributable to other wastes.

Table 9.9. Methane Emissions Attributable to Landfilled Recovery Area Wastes from Pulp and Paper Mills

Year	Metric Tons Methane Emitted – NCASI Calculations Using GHGRP Equations – GHGRP Default DOC and k (Moderate Climate)	Fraction of 1990 value	Fraction of 2005 value
1990	7,700	1.00	
2005	10,800	1.41	1.00
2020	11,800	1.53	1.09

9.1.1.4 Other Waste from Pulp and Paper Mills

In addition to WWTR, ash, and recovery wastes, many mills dispose of other materials in landfills. These other materials can include wood yard waste, pulp mill and paper mill rejects, and miscellaneous refuse. The types and quantities vary among mills, as do the methods used for disposal. The amounts of other waste were derived using the same set of data as described for recovery area wastes, using the assumption that all "other waste" reported in NCASI surveys that is not from the recovery area could be considered "other pulp and paper wastes" as described in GHGRP rules (US EPA 2023a, Subpart TT Table TT-1). For years up to 1995, this resulted in a value of 51.5 dry kg per metric ton. This declined to 29.5 to 31 kg per metric ton between 2004 and 2015, when it began increasing, reaching 35.5 kg per metric ton in 2020. Dry weights were converted to wet weights using a solids content of 53%, derived from Table 3.3 of NCASI (2019)[17]. The fraction landfilled (0.342) was derived from data in Figure 11 and Table 8 in NCASI (1999a) for nonrecovery area wastes. The model parameter values used to estimate landfill methane emissions attributable to these other solid wastes are shown in Table 9.10. The estimated emissions are shown in Table 9.11.

Table 9.10. Pulp and Paper Mill "Other" Waste Parameter Values for First Order Decay Model

Parameter	Units	GHGRP, Subpart TT
DOC	kg organic C/kg wet waste	0.2
DOCf	Fraction of DOC that will degrade in landfill	0.5
k	yr^{-1}	0.02 (dry climate) 0.03 (moderate climate) 0.04 (wet climate)
MCF	Fraction of degradable carbon that is subject to anaerobic conditions	1
F	Fraction by volume of CH_4 in landfill gas, as generated	0.5
OX	Fraction of CH_4 that is oxidized before it is released to the atmosphere	0.1 (values ranging from 0.0 to 0.35 are listed with the choice depending on site-specific circumstances)

Table 9.11. Methane Emissions Attributable to Landfilled "Other" Wastes from Pulp and Paper Mills

Year	Metric Tons Methane Emitted – NCASI Calculations Using GHGRP Equations – GHGRP Default DOC and k (Moderate Climate)	Fraction of 1990 value	Fraction of 2005 value
1990	71,800	1.00	
2005	98,800	1.38	1.00
2020	100,300	1.40	1.01

9.1.1.5 Assume All Landfilled Waste, Except WWTR, is Combined

US EPA's GHGRP includes parameter values for pulp and paper combined wastes, not including WWTR (US EPA 2023a). The methods used to estimate quantities of the individual materials are described earlier in this report. These were summed (not including WWTR) to perform calculations using this additional category in the GHGRP. The parameter values are shown in Table 9.12 and the

[17] The is the simple average of the reported values in Table 3.3 of NCASI (2019) for virgin fiber pulping rejects, secondary fiber pulping rejects (deinking and other), uncombusted woodyard residuals, process water treatment solid residuals, and general mill refuse.

results in Table 9.13. When ash was assumed to be landfilled at 60% solids, the weighted average solids content for the combined wastes was also assumed to be at 60%.

The estimated emissions from assumed combined landfilling of ash, recovery area wastes, and “other” wastes are almost double those calculated when the three emissions are estimated separately, as shown in Table 9.14. An analysis of the calculations indicates that the difference is primarily due to the large difference between the DOC value for ash managed separately (DOC = 0.06) compared to the value applied if it is combined with other non-WWTR wastes (DOC = 0.15). As noted previously, most of the ash from the forest products industry is low in organic content and high in pH, suggesting that the amounts of methane produced from it would be low. Based on this, it is NCASI’s judgment that the higher estimate of methane emissions associated with US EPA’s parameter values for combined wastes is less reliable than the lower value obtained by examining the wastes separately.

Table 9.12. First Order Decay Model Parameters for non-WWTR Combined Wastes

Parameter	Units	GHGRP, Subpart TT
DOC	kg organic C/kg wet waste	0.15
DOCf	Fraction of DOC that will degrade in landfill	0.5
k	yr^{-1}	0.02 (dry climate) 0.03 (moderate climate) 0.04 (wet climate)
MCF	Fraction of degradable carbon that is subject to anaerobic conditions	1
F	Fraction by volume of CH_4 in landfill gas, as generated	0.5
OX	Fraction of CH_4 that is oxidized before it is released to the atmosphere	0.1 (values ranging from 0.0 to 0.35 are listed with the choice depending on site-specific circumstances)

Table 9.13. Methane Emissions for Non-WWTR Combined Wastes from Pulp and Paper Mills

Year	Metric Tons Methane Emitted	Fraction of 1990 value	Fraction of 2005 value
	NCASI Calculations Using GHGRP Equations – 60.1% Solids – GHGRP Default DOC and k (Moderate Climate) for Combined Wastes		
1990	226,000	1.00	
1995	267,000	1.18	
2000	301,000	1.34	
2005	322,000	1.43	1.00
2010	337,000	1.49	1.05
2015	343,000	1.52	1.06
2020	337,000	1.49	1.05

Table 9.14. Comparison of Combined and Individual Landfilling of Non-WWTR Solid Wastes at Pulp and Paper Mills

Year	Metric Tons Methane	
	Co-disposed, Not Including WWTR	Individually Disposed, Not Including WWTR
1990	226,000	120,000
2005	322,000	163,000
2020	337,000	165,000

9.1.1.6 Total Landfill Methane Emissions from Pulp and Paper Mill Wastes

The earlier analysis yields estimates of methane emissions attributable to landfilled solid wastes from pulp and paper mills, shown in Table 9.15.

Table 9.15. Total Methane Emissions Attributable to Landfilled Solid Wastes from Pulp and Paper Mills

Year	Metric Tons Methane Emissions					
	WWTR	Ash	Recovery Area	Other Wastes	Total of Individually Disposed Wastes	WWTR Emissions Plus Combined Non-WWTR Wastes, Co-disposed
1990	175,000	40,000	7,700	72,000	295,000	401,000
2005	202,000	53,000	11,000	99,000	365,000	524,000
2020	186,000	53,000	12,000	100,000	350,000	523,000

The contributions of different wastes to these emissions in 2020 are shown in Figure 9.3. WWTR contributes most to emissions, at about 53% of the total. Boiler ash contributes 15% and other waste contributes 29%. Methane emissions attributable to recovery area wastes are only about 3%.

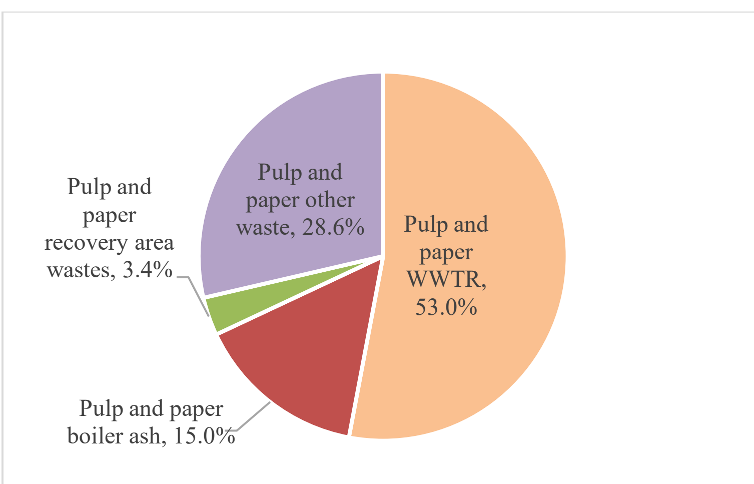


Figure 9.3. Contributions of Different Wastes to Pulp and Paper Mill 2020 Landfill Methane Emissions

Emissions in 2020 are compared to those in 1990 and 2005 in Table 9.16. Emissions in 2020 were 19% higher than 1990 emissions, based on the sum of individual waste estimates, and 4% below 2005 emissions. If non-WWTR wastes are modeled as co-disposed, 2005 and 2020 emissions have increased more than indicated in calculations that keep the wastes separate.

Table 9.16. 1990 to 2020 Change in Methane Emissions Attributable to Landfilled Solid Wastes from Pulp and Paper Mills

	Fraction of 1990 Emissions		Fraction of 2005 Emissions	
Year	Sum of Individual Wastes Separately Disposed	WWTR Emissions Plus Combined Disposal of All Other Wastes	Sum of Individual Wastes Separately Disposed	WWTR Emissions Plus Combined Disposal of All Other Wastes
1990	1.00	1.00		
2005	1.24	1.31	1.00	1.00
2020	1.19	1.30	0.96	1.00

9.1.2 Wood Product Facilities – Lumber

Wood products manufacturers often use wood-derived fuels that produce boiler ash. In addition, other wastes can be associated with wood handling and manufacturing operations. For this profile, therefore, methane emissions attributable to landfilling of ash and non-ash wastes were estimated. Non-ash wastes from wood products mills can include wood yard waste and manufacturing waste not suitable for use as fuel, as a raw material in the mill, or elsewhere.

The products and processes used at wood product plants vary. The available waste-related data, however, do not allow detailed differentiation between types of mills. Therefore, for this analysis, the wood products industry was divided into only two sectors: lumber and panels. The lumber category includes all softwood and hardwood lumber while the panel category includes all panels and engineered wood products. The high-level aggregation of mill types obscures important differences that may need to be considered in assessments dealing with issues beyond those addressed in this report (e.g., substitution effects).

9.1.2.1 Boiler Ash

As noted previously, boiler ash was not included in earlier profiles due to its low content of organic matter and, in the case of ash derived from wood ash, its high pH. Nonetheless, because US EPA has published parameter values for estimating methane emissions from ash, methane attributable to ash disposal is estimated in this profile.

There are few survey data available on the amounts and types of solid waste from wood products mills. For this profile, NCASI calculated ash quantities based on data on biomass energy consumption reported in EIA MECS surveys (EIA 2023a). Biomass energy consumption per unit of production was calculated using production data from FAOSTAT (FAOSTAT 2023). An analysis of biomass energy consumption per unit of production over time did not reveal a statistically significant trend; therefore, the average biomass energy use per unit of production was used for all years (based on EIA MECS data from 1994 to 2018). The average was calculated to be 1.3 million BTU HHV (or 1370 MJ) per m^3 of lumber. This was converted to fuel quantity using a factor of 17.48 million BTU HHV per short ton of biomass fuel (US EPA 2016). Ash production was calculated by multiplying biomass quantity by an ash content of 2% (value derived in Someshwar 1996). These factors resulted in a calculated ash generation rate of 1.35 dry kg per m^3 of lumber. This value was used for all years. For the pulp and paper sector (discussed earlier in this section), NCASI relied on published information indicating that the fraction landfilled was 84%, 72%, and 75% in 1988, 1995, and 2016 respectively (NCASI 1992; 1999a; 2019). Due to the lack of robust data for lumber mills, these pulp and paper mill data were used to describe the disposition of boiler ash. The landfilling rates between 1988 and 2016 were interpolated using these values. The value for 1988 was extended back to 1960, while the 2016 value was extended forward to 2020.

The DOC values in GHGRP Subpart TT rules (US EPA 2023a) are on a wet weight basis. Generation rates, however, are on a dry basis, requiring a factor to convert them to as-disposed wet weight. Data are not available, however, on the solids content of boiler ash disposed of in wood product mill landfills. For pulp and paper mill ash disposal estimates, data were available, allowing solids content to be estimated over time. These data are not available for wood products mills, however. Therefore, the average solids content for pulp and paper mill ash disposal from 1990 to 2020, equal to 83%, was used. The current default DOC values for ash are most likely based on as-disposed solids content (see Appendix E). Therefore, the base case for this assessment, used when a single estimate is needed, is based on an assumed solids content of 83% in as-disposed ash.

The parameter values used for modeling methane emissions attributable to landfilled ash from lumber mills were the same as those used for pulp and paper mills (see Table 9.6). Results are shown in Table 9.17. The emissions are much lower than those attributable to ash disposal at pulp and paper mills because the amounts of ash disposed are much lower.

Table 9.17. Methane Emissions Attributable to Landfilled Boiler Ash from Lumber Mills

Year	Metric Tons Methane Emitted	Fraction of 1990 Value – Reflects Only Changes in Lumber Production	Fraction of 2005 Value – Reflects Only Changes in Lumber Production
1990	1.100	1.00	
2005	1,400	1.31	1.00
2020	1,500	1.41	1.08

9.1.2.2 Other Wastes

CORRIM studies[18] were used to develop a factor for estimating the amounts of non-ash solid waste landfilled at lumber mills. Data were taken from LCA reports on softwood lumber from the Pacific Northwest and Southeast US (Milota 2020a; 2020b). For all processes in the mill, the reported quantities of "dirt and rocks," "general trash," "woody material," and "other organics" were added together and then summed for all processes. The result was rounded to an "other waste" generation rate of 0.009 kg/m^3 of lumber, which was used for all years. This factor was multiplied by

[18] See CORRIM.org for more information.

the production of lumber, downloaded from FAOSTAT (FAOSTAT 2023), for every year from 1960 to 2020. As a result, the year-to-year changes in production of “other waste” are entirely reflective of changes in production.

The DOC values in GHGRP Subpart TT (US EPA 2023a) are on a wet weight basis; therefore, the dry weights estimated herein were converted to wet weights using a solids content of 85% based on the fact that the GHGRP DOC is taken from IPCC (Pipatti et al. 2006), where the associated water content is 15%.

The GHGRP Subpart TT defaults for non-ash waste from wood and wood products mills are shown in Table 9.18. Note that the DOC for non-ash wastes is more than double the corresponding value for miscellaneous wastes from pulp and paper mills (0.2).

Table 9.18. Wood Product Non-Ash Waste Parameter Values for First Order Decay Model

Parameter	Units	GHGRP, Subpart TT
DOC	kg organic C/kg wet waste	0.43
DOCf	Fraction of DOC that will degrade in landfill	0.5
k	yr^{-1}	0.02 (dry climate) 0.03 (moderate climate) 0.04 (wet climate)
MCF	Fraction of degradable carbon that is subject to anaerobic conditions	1
F	Fraction by volume of CH_4 in landfill gas, as generated	0.5
OX	Fraction of CH_4 that is oxidized before it is released to the atmosphere	0.1 (values ranging from 0.0 to 0.35 are listed with the choice depending on site-specific circumstances)

The estimated emissions associated with landfilled non-ash waste from lumber mills, derived using the GHGRP Subpart TT default DOC for “wood and wood product [wastes] (other than industrial sludges)” are shown in Table 9.19. It is likely that these estimates are too high given the nature of the material involved. The major components of this waste are likely to be wood-derived materials that are not usable as raw material or fuel due to their contamination with soil and other debris. The default DOC of 0.43 at 85% solids equates to a carbon content of 50% of dry weight (which is reasonable if most of the material is woody). However, the default GHGRP value for the fraction of carbon degradable in the landfill is 50%, which is not reasonable given current research. US EPA’s Waste Reduction Model (WARM), for instance, considers 88% of the carbon in lumber and 77% of the carbon in branches to be nondegradable in anaerobic landfills (US EPA 2020b).

If one uses an assumption that 80% of the carbon is nondegradable (DOCf = 0.2) instead of the GHGRP Subpart TT value (DOCf = 0.5), estimated methane production decreases by 60%. The effect of this is approximately equivalent to using the DOC for miscellaneous wastes from pulp and paper mills (0.2) and a DOCf of 0.5. The alternative estimates, using a DOC of 0.43 (GHGRP Subpart TT default for non-ash waste at wood products mills) and a DOCf of 0.2 are shown in Table 9.20. These values are used in this profile in places where a single estimate is required.

Table 9.19. Methane Emissions Attributable to Landfilled Non-Ash Solid Waste from Lumber Mills Using GHGRP Subpart TT Default Values[a]

Year	Metric Tons Methane Emitted	Fraction of 1990 Value	Fraction of 2005 Value
1990	60	1.00	
2005	80	1.40	1.00
2020	90	1.54	1.10

[a] Changes reflect only changes in production of lumber.

Table 9.20. Methane Emissions Attributable to Landfilled Non-Ash Solid Waste from Lumber Mills – Using GHGRP Subpart TT Default Values Except for DOCf (Using 0.2 Instead of 0.5)

Year	Metric Tons Methane Emitted	Fraction of 1990 Value[a]	Fraction of 2005 Value[a]
1990	24	1.00	
2005	33	1.40	1.00
2020	36	1.54	1.10

[a] Changes reflect only changes in production of lumber.

9.1.3 Wood Product Facilities – Panel Plants

9.1.3.1 Boiler Ash

The approach to estimating methane attributable to landfilled boiler ash at panel plants was the same as described earlier for lumber mills. The amounts of landfilled ash were estimated from fuel consumption reported in EIA MECS reports. Ash generation rate was estimated to average about 2.39 kg dry ash per m^3 of panel (considering all years with EIA MECS data). Because of limited data and little evidence that there was a trend over time, this value was used for all years. This factor was multiplied by the production of panels, downloaded from FAOSTAT, every year from 1960 to 2020 (FAOSTAT 2023). As a result, the year-to-year changes in emissions are entirely reflective of changes in production. The fraction of ash landfilled was assumed to be the same as that for lumber mills, described previously.

US EPA's current default DOC values for boiler ash are based on as-disposed solids content. Therefore, for the calculations in this update, DOC was converted to a dry basis using the same assumptions for solids content as used for lumber mills (i.e., 83% solids).

The parameter values used for modeling methane emissions attributable to landfilled ash from panel plants were the same as those used for pulp and paper mills (see Table 9.6). Results are shown in Table 9.21. The emissions are much lower than those attributable to ash disposal at pulp and paper mills and lumber mills because the amounts of ash disposed are much lower.

Table 9.21. Methane Emissions Attributable to Landfilled Boiler Ash from Panel Plants

Year	Metric Tons Methane Emitted	Fraction of 1990 Value[a]	Fraction of 2005 Value[a]
1990	54	1.00	
2005	87	1.62	1.00
2020	107	2.00	1.23

[a] Changes reflect only changes in production of panels.

9.1.3.2 Other Wastes

CORRIM studies were used to develop a factor for estimating the amounts of non-ash solid waste landfilled at panel plants. Data taken from LCA reports on plywood from the Pacific Northwest and Southeast US were used (Puettmann et al. 2020a; 2020b). The data are limited; therefore, NCASI used a simple average of "solid waste – wood waste" from these two reports. This produced a generation rate of 20 dry kg waste per cubic meter of plywood, which was used for all years. This factor was multiplied by the production of panels, downloaded from FAOSTAT, every year from 1960 to 2020 (FAOSTAT 2023). As a result, the year-to-year changes in disposal quantity and emissions are entirely reflective of changes in production.

The DOC values in GHGRP Subpart TT are on a wet weight basis; therefore, the dry weights estimated herein were converted to wet weights using a solids content of 85% based on the fact that the Subpart TT DOC is taken from IPCC (Pipatti et al. 2006), in which the associated water content is 15%.

The GHGRP Subpart TT defaults for non-ash waste from wood and wood products mills are shown in Table 9.18 (US EPA 2023a). Note that the DOC for non-ash wastes is more than double the corresponding value for miscellaneous wastes from pulp and paper mills (0.2).

The estimated emissions associated with landfilled non-ash waste from panel plants, derived using the GHGRP Subpart TT default DOC for "wood and wood product [wastes] (other than industrial sludges)" are shown in Table 9.22. As noted earlier, it is likely that these estimates are too high given the nature of the material involved. If the calculations are performed using an assumption that 80% of the carbon is nondegradable (DOCf = 0.2) instead of the Subpart TT value (DOCf = 0.5), estimated methane production decreases by 60%. This is approximately equivalent to using the DOC for miscellaneous wastes from pulp and paper mills (0.2) and a DOCf of 0.5. The alternative estimates, using a DOC of 0.43 (Subpart TT default for non-ash waste at wood products mills) and a DOCf of 0.2 are shown in Table 9.23. These values are used as the base case estimates in this profile. These values suggest that non-ash wastes from panel plants (and other mills that are not lumber mills) contribute approximately 90% of emissions from landfilled wastes in the wood products sector.

Table 9.22. Methane Emissions Attributable to Landfilled Non-Ash Solid Waste from Panel Plants Using GHGRP Subpart TT Default Parameter Values

Year	Metric Tons Methane Emitted	Fraction of 1990 Value[a]	Fraction of 2005 Value[a]
1990	38,000	1.00	
2005	66,000	1.77	1.00
2020	83,000	2.21	1.25

[a] Changes reflect only changes in panel production.

Table 9.23. Methane Emissions Attributable to Landfilled Non-Ash Solid Waste from Panel Plants – Using GHGRP Subpart TT Default Values Except for DOCf (Using 0.2 Instead of 0.5)

Year	Metric Tons Methane Emitted	Fraction of 1990 Value[a]	Fraction of 2005 Value[a]
1990	15,100	1.00	
2005	26,600	1.77	1.00
2020	33,200	2.21	1.25

[a] Changes reflect only changes in panel production.

9.1.4 Summary of Landfill Methane Emissions from US Forest Products Industry Wastes

Methane emissions and CO_2eq. for 1990, 2005, and 2020 are shown in Table 9.24. The changes since 1990 and 2005 are summarized in Table 9.25. A breakdown of industry landfill methane emissions is shown in Figure 9.4. Over 90% of the industry's landfill methane emissions are attributable to pulp and paper mill wastes. WWTR contributes about one-half of methane emissions from pulp and paper mill wastes.

Table 9.24. Methane and CO_2 Equivalent Emissions from Forest Products Industry Landfills

Landfilled Waste	Metric Tons Methane			Million Metric Tons CO_2eq. [a]		
	1990	2005	2020	1990	2005	2020
Pulp and paper WWTR	175,000	202,000	186,000	4.4	5.1	4.6
Pulp and paper boiler ash	40,000	53,000	53,000	1.0	1.3	1.3
Pulp and paper recovery area wastes	7,700	11,000	12,000	0.2	0.3	0.3
Pulp and paper other waste	72,000	99,000	100,000	1.8	2.5	2.5
Total pulp and paper	295,000	365,000	350,000	7.4	9.1	8.8
	1990	2005	2020	1990	2005	2020
Lumber mill boiler ash	1,100	1,400	1,500	0.03	0.03	0.04
Lumber mill non-ash waste (DOCf = 0.2)	20	30	40	0.0006	0.0008	0.0009
Panel plant boiler ash	500	900	1,100	0.01	0.02	0.03
Panel plant non-ash waste (DOCf = 0.2)	15,000	27,000	33,000	0.4	0.7	0.8
Total wood products	17,000	29,000	36,000	0.4	0.7	0.9
	1990	2005	2020	1990	2005	2020
Total forest products industry	312,000	394,000	386,000	7.8	9.8	9.7

[a] GWP of 25.

Table 9.25. Changes in CH_4 Emissions Attributable to Landfilled Wastes in the Forest Products Industry

	2020 Metric Tons Methane Emissions	Fraction of 1990	Fraction of 2005
Pulp and Paper Mills			
Wastewater treatment residuals	186,000	1.06	0.92
Boiler ash	53,000	1.31	0.99
Recovery area wastes	12,000	1.53	1.09
Other wastes	100,000	1.40	1.01
Pulp and paper total of individual wastes[a]	350,000	1.19	0.96
Lumber Mills			
Boiler ash	1,500	1.41	1.08
Other wastes[b]	40	1.54	1.10
Panel Plants			
Boiler ash	1,100	2.00	1.23
Other wastes[b]	33,000	2.21	1.25
Wood products sector total	36,000	2.15	1.24
Total Forest Products Industry	386,000	1.24	0.98

[a] GHGRP Subpart TT contains first order decay parameter values for individual wastes and for all non-WWTR wastes combined. The value for individual wastes, shown here, is used as the base case for this profile.
[b] Using a DOCf of 0.2 (based on literature) instead of GHGRP default of 0.5.

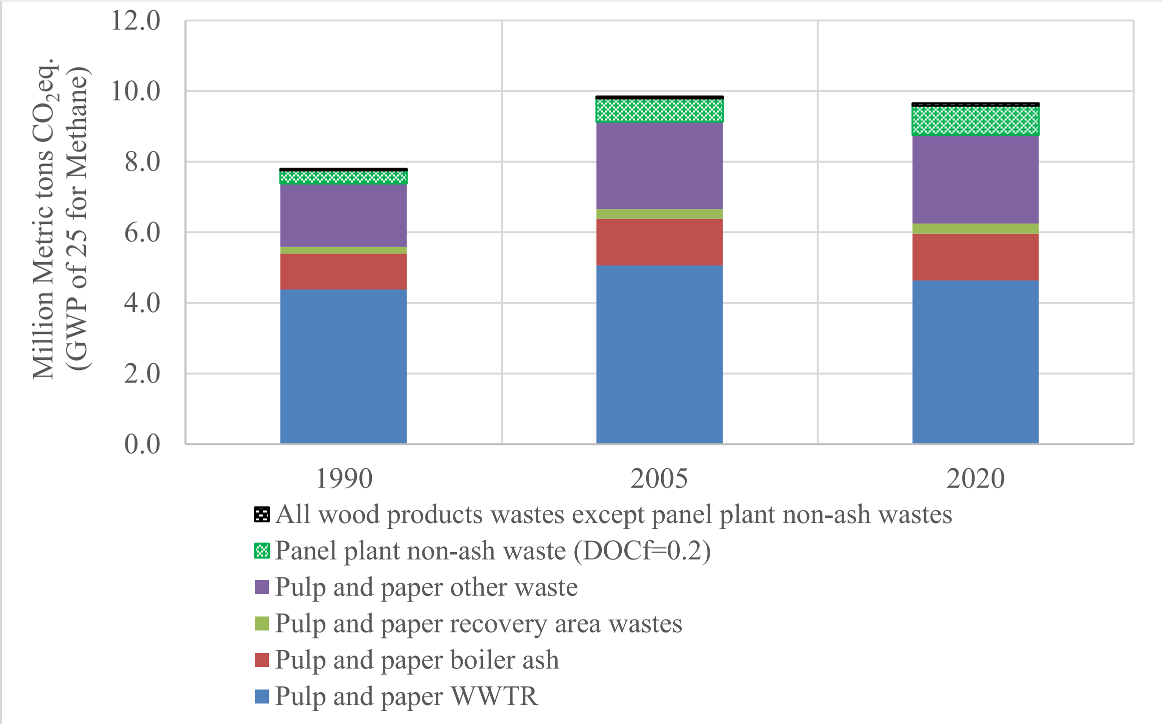


Figure 9.4. Sources of Methane Emissions Attributable to Landfilled Waste from the Forest Products Industry

9.1.5 Comparison to Results of 2005 Profile Study

This updated profile estimates that landfilled waste from the forest products industry was associated with the emission of 386,000 metric tons of methane in 2020. This is equal to 9.7 million metric tons CO_2 (GWP = 25). The updated estimate for 2005 emissions is 9.8 million metric tons CO_2eq. The previous profile estimated 2005 emissions from landfills to be 2.60 million metric tons CO_2eq., after converting from a GWP of 21 to a GWP of 25 (NCASI 2008; Heath et al. 2010).

There are several reasons for these large differences. First, the earlier profile considered only WWTR from pulp and paper mills and landfilled material from wood products plants, with WWTR contributing almost all of the emissions. In this updated profile, more types of wastes are considered, reflecting the scope of wastes with parameter values in the GHGRP. Nonetheless, the updated estimate for WWTR from pulp and paper mills in 2005 is 4.6 million metric tons of CO_2eq., which is still much larger than 2.6.

Much of the remaining differences can be attributed to the use of different values for k and DOC than were used in the 2005 profile. In particular, very different values were used for WWTR DOC. In this study, the default DOC for WWTR is 0.12. Using this DOC, the solids content for WWTR applied in this study (31% solids), and the default DOCf in the GHGRP (0.5), one can show that the equivalent Lo is 193 m^3 per dry metric ton of waste[19]. This compares to the Lo of 80.5 m^3/metric ton

[19] Lo is a commonly used alternative to DOC. It is expressed in units of volume of methane per unit of waste. Lo was used in the 2005 study. See Appendix D for more information.

dry waste used in the 2005 study, the best value available at the time. This difference largely explains why the updated profile produces WWTR methane estimates that are more than twice those calculated in the 2005 profile.

There are other differences between the calculations in 2005 and 2020, which suggest, in total, that it is not possible to directly compare the landfill methane emissions calculated in the earlier profile to those calculated in this update.

9.2 Emissions from Wastewater Treatment

In the previous profile (NCASI 2008; Heath et al. 2010), emissions from treatment of wastewater from pulp, paper, and paperboard mills were estimated using a small number of field studies. The emissions were estimated to be 0.4 million metric tons CO_2eq. in both 1990 and 2005. In addition to providing a basis for the estimates, the field studies revealed large variability in these emissions, both within and between sites.

In its 2022 annual inventory of GHG emissions and sinks, US EPA estimated 2020 GHG emissions from the treatment of pulp, paper, and paperboard mill wastewaters (US EPA 2022a). US EPA's estimates are model-based and are also subject to considerable uncertainty. The estimates produced by US EPA are approximately twice those developed using NCASI's field studies. Although higher than NCASI's earlier estimates, US EPA's estimates confirm the finding that these emissions are relatively small. Given the comparable results and the readily available nature of US EPA's inventory results, US EPA's estimates are used as the basis for this updated inventory.

Emissions associated with the treatment of wastewater from wood products mills and from converting operations are not included as these are very small compared to those from pulp, paper, and paperboard manufacturing.

9.2.1 Methods

US EPA estimates wastewater treatment emissions using IPCC models that calculate methane release based on information regarding the amount of organic matter subject to anaerobic conditions and factors that convert a fraction of this material into methane. US EPA also estimates nitrous oxide emissions associated with treating industry wastewater (US EPA 2022a). To develop estimates for this profile, NCASI assumed that the most important parameter in US EPA's calculations that changed significantly since 1990 was the amount of organic matter in untreated wastewater. Accordingly, estimates of 1990 and 2005 emissions were produced by scaling US EPA's 2020 estimate based on influent loads. Data in NCASI 2019 indicate that influent loads decreased from about 25 kg BOD/metric ton production in 1990 to 15 kg/metric ton in 2020. Production data to convert these to mass loads were taken from US EPA (2022b).

It Is important to note that US EPA's estimates for N_2O emissions include emissions from wastewater treatment plants as well as emissions that occur in the environment attributable to the residual nitrogen in industry effluents. In addition, NCASI's analysis suggests that, because US EPA does not consider several industry-specific factors, its estimates of N_2O emissions from wastewater treatment plants may be high.

9.2.2 Results

Estimated emissions of methane and nitrous oxide (in CO_2eq.) associated with treatment of wastewater from pulp, paper, and paperboard mills are shown in Table 9.26. These emissions have declined steadily over time, amounting to 1.6, 1.4, and 0.9 million metric tons CO_2eq. in 1990, 2005, and 2020, respectively. The declines are due to reductions in wastewater loads, per unit of production, and reduced production.

Table 9.26. Greenhouse Gas Emissions (CH_4 and N_2O) Associated with Treatment of Wastewaters from Pulp, Paper, and Paperboard Mills

Year	Methane and Nitrous Oxide, Million Metric Tons CO_2eq.[a]	Fraction of 1990 Emissions	Fraction of 2005 Emissions
1990	1.6	1.00	
2005	1.4	0.92	1.00
2020	0.9	0.57	0.62

[a] GWPs of 25 and 298 for methane and nitrous oxide, respectively.

10.0 EMISSIONS ASSOCIATED WITH FOREST PRODUCTS END-OF-LIFE

At end-of-life, forest products are recycled, landfilled, combusted, or diverted to a beneficial use (e.g., composting). Emissions associated with recycling occur in transporting recovered fiber and using it to produce new products. These emissions are accounted for elsewhere in this study. End-of-life emissions from combustion (primarily municipal solid waste [MSW] waste-to-energy) are not included because US EPA estimates that emissions of N_2O and CH_4 associated with MSW combustion[20], in total, are less than 0.5 million metric tons of CO_2eq. per year, and only a fraction of this can be attributed to forest products (US EPA 2022a).

EPA's annual inventory indicates that total emissions from composting in 2020 were 4.3 million metric tons CO_2eq. per year (US EPA 2022a). The wastes creating these emissions are, according to US EPA, "primarily yard trimmings (grass, leaves, and tree and brush trimmings) and food scraps from the residential and commercial sectors (such as grocery stores; restaurants; and school, business, and factory cafeterias)" (US EPA 2022a). There is no reason, therefore, to suspect that paper and wood products are a significant source of GHGs from composting, so these potential emissions are not considered in this updated profile.

In this update, the methane emissions attributable to discarded industry products in landfills are calculated two ways. First, we estimate the annual methane emissions associated with products placed in landfills in the current and past years. This is the approach used in national inventories. In addition, we project future methane emissions associated with paper and wood products manufactured in specific years (i.e., 1990, 2005, and 2020).

10.1 Annual Emissions Associated with Current and Past Disposal of Forest Products

Forest products can degrade when exposed to anaerobic conditions that exist in most MSW landfills, resulting in production of methane and carbon dioxide. Because CO_2 is biogenic in origin, it is accounted for in estimating stock changes in forests and products. It is not considered in GHG totals (Pipatti et al. 2006; US EPA 2022a), but the methane is considered in GHG totals because its GWP is over 20 times that of carbon dioxide.

[20] Carbon dioxide produced by combustion of forest products in MSW is not included because it is part of the calculation of changes stocks of carbon in forest products presented earlier in this report.

Methane released from MSW landfills is a result of decomposition of a range of materials, including food and yard waste. As a result, estimates of total methane released from MSW landfills are not appropriate for characterizing the forest products industry's GHG profile. Instead, it is necessary to estimate the amounts of methane that result from decomposition of forest products only.

In the 2008 profile report, this was done by the US Forest Service using the Woodcarb II model (Heath et al. 2010). Woodcarb II estimates changes in domestically produced wood products in use, discards from use, deposits in dumps and landfills, and decay in dumps and landfills.

For this update to the profile, we have used a different method. US EPA's annual inventory values for total MSW landfill methane emissions have been used as a starting point, and the fraction attributable to forest products has then been calculated. This new approach has at least one significant advantage. Instead of being based only on models, US EPA's estimates of MSW landfill methane emissions also incorporate direct measurements from landfill gas recovery systems. These are considered "Tier 3 (highest quality data) under the 2006 IPCC Guidelines" (Section 3.14 in US EPA 2020b).

The primary disadvantage of this new method is that it does not use the production approach, meaning that the emissions are not specifically attributable to wood harvested in the US (US EPA 2022a). Instead, this new approach calculates methane emissions attributable to all forest products disposed of in US landfills, regardless of where the wood was harvested.

It is difficult to precisely determine the expected difference between the two approaches. An emissions estimate based, at least in part, on direct measurements would be expected to be more accurate than one based only on modeling. There are significant uncertainties, however, in estimating the fraction of these emissions attributable to domestic production of forest products. First, the forest products in landfills come from both domestic and non-domestic production. Second, methane emissions from landfills are attributable to all decomposable materials placed in landfills, of which forest products are only a fraction.

The potential impacts of non-domestic forest products in landfills can be examined in several ways. One is by comparing imports and exports of forest products. Over the past several decades, with the exception of lumber, imports and exports of forest products have been comparable, as shown in Figure 10.1. Assuming the end-of-life fates of imported and exported products are the same, the impacts of exported forest products would be expected to be approximately offset by the impacts of imported forest products, which are included.

In the case of lumber, a large fraction of the material deposited in landfills is nondegradable and does not contribute to methane emissions. Wang et al. (2011), for instance, found that only 0 to 19.9% of the carbon in a range of wood products was degradable under anaerobic conditions. For wood products derived from softwood, like most lumber, the range was 0 to 1.8%. In addition, over the period of record, imports of lumber have been larger than exports, so to the extent that these products contribute to methane emissions from landfills, considering all methane, regardless of the source of the wood, will tend to overestimate the amounts attributable to lumber produced from wood harvested in the US, resulting in a conservative estimate.

More difficult to assess is the potential impact of (1) paper, paperboard, and wood associated with containers used to package or ship imported products and (2) other imported consumer products that are comprised, in part, of paper, paperboard, or wood.

The difficulty caused by these materials is exemplified by comparing US EPA's various estimates of MSW quantities. In its most recent "Advancing Sustainable Materials Management" report, US EPA estimates that in 2018, 133 million metric tons of material were landfilled in the US (US EPA 2020e). In the annual inventory of emissions and sinks, however, US EPA estimates that 213 million metric tons of MSW were landfilled in 2018 (US EPA 2022b), an estimate 1.6 times that in the "Advancing Sustainable Materials Management" report. US EPA provides the following explanation for the difference[21]:

Figure 10.1. Domestic Production, Imports and Exports of Forest Products [*Source*: Howard and Liang (2019)]

The EREF-extrapolated estimate of total MSW landfilled for 2017 and 2018 [i.e., the value in US EPA (2022b)] is based on a bottom-up approach using information at the facility-level to estimate MSW for the sector as a whole, while the Facts and Figures

[21] The factors contributing to the difference have also been examined by Staley and Kantner (2018).

> report [i.e., US EPA (2020e)] uses a top-down (materials flow mass balance) approach to estimate the same quantity. The materials flow methodology is generally based on production data for each material at the state- (recycling, composting) or national- (waste generation) level. (US EPA 2022b)

The potential impact of imported products is specifically acknowledged in US EPA's description of methods used in the Facts and Figures report (US EPA 2020e), where US EPA identifies the need to adjust for packaging of imported goods but finds that "current data for adjustments for packaging of imported goods are not available."

While it is not possible to precisely estimate the impact of forest products used for packaging or shipping, or otherwise associated with imported consumer goods, the nature of the effect may be examined by comparing imports and exports of consumer goods. According to the Bureau of Economic Analysis in the Department of Commerce, imports of consumer goods are typically much larger than exports, as shown in Figure 10.2 (BEA 2024). The amounts of paper and paperboard packaging disposed of in the US are, therefore, larger than they would be if they only included material of domestic origin. Using data from the Bureau of Economic Analysis (BEA 2024), NCASI estimates that, excluding vehicles, food, and energy, net imports of consumer goods to the US account for about 15 to 20% of personal consumption expenditures for these goods. This is large enough to suggest that packaging for imports of consumer goods may affect estimates of the quantity and origin of paper and paperboard in MSW.

Because some landfilled forest products are of non-domestic origin, measurements and models of gas production from all landfilled MSW are larger than they would be if only material of domestic origin was being landfilled. Hence, estimates based on the total methane attributable to landfilled paper and paperboard may overstate the contribution of the US forest products industry.

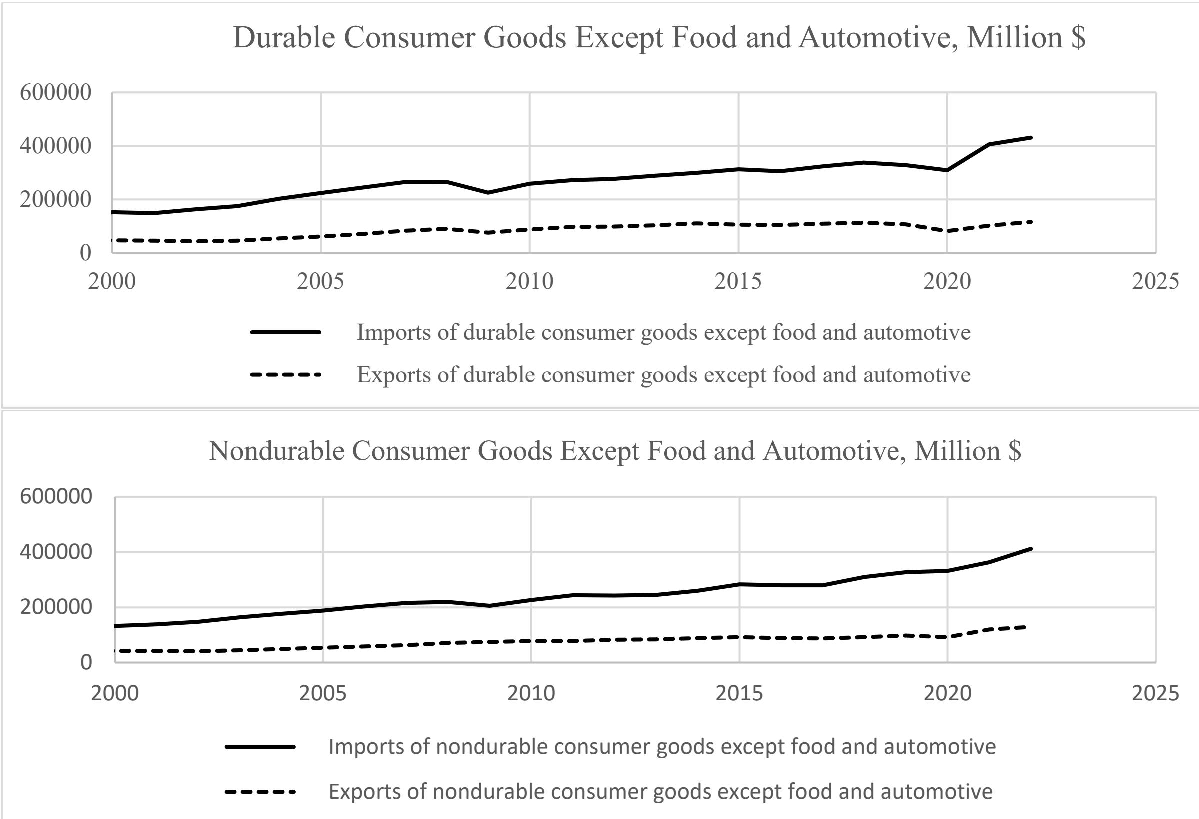


Figure 10.2. Imports and Exports of Consumer Goods Except Food and Automotive [*Source*: BEA (2024)][22]

While it is difficult to adjust methane emission measurements to account for non-domestic forest products, it is possible to adjust the emissions to isolate the role of forest products compared to other degradable materials. To estimate the fraction of methane attributable to forest products, we separately modeled the decay of all decomposable fractions of MSW deposited in landfills since 1960. The data on deposits were taken from US EPA (US EPA 2020e)[23]. Parameter values for decay rates and nondegradable carbon were taken from the documentation for version 15 of US EPA's WARM (US EPA 2020b).

The analysis of year-to-year decomposition and methane production, described in Appendix F and shown in Figure 10.3, reveals that the contribution of forest products to MSW methane releases has decreased over time. The primary reason is that paper and paperboard's contribution to MSW has

[22] Nondurable consumer goods, excluding food and auto, include the following: apparel, footwear, and household goods; medicinal, dental, and pharmaceutical products; toiletries and cosmetics; and other nondurable goods.
Durable consumer goods, excluding food and auto, include the following: televisions, video receivers, and other video equipment; radio and stereo equipment, including recorded media; toys and sporting goods, including bicycles; household and kitchen appliances and other household goods; jewelry and collectibles; gem diamonds and other gemstones; and other durable goods.
[23] US EPA (2020e) data are likely affected by the issues in accounting for imported products noted previously. How this affects the data on paper and paperboard compared to other components of MSW has not been examined. The data set is used because it is the only consistent time series on MSW composition available.

diminished. In 1990, paper and paperboard represented 30% of MSW landfill input. By 2018, this had dropped to 11.8% (data in Appendix F). As a fraction of degradable materials going to landfill, the contribution from paper and paperboards decreased from 44% in 1990 to 23% in 2018. Also evident in Figure 10.3 is the effect of efforts to divert yard waste from MSW landfills.

These estimates can be compared to those recently developed by US EPA for methane from food waste. US EPA estimates that food waste was responsible for 58% of landfill methane releases in 2020 (US EPA 2023d), which compares to an estimate for 2018 in this study of 60%.

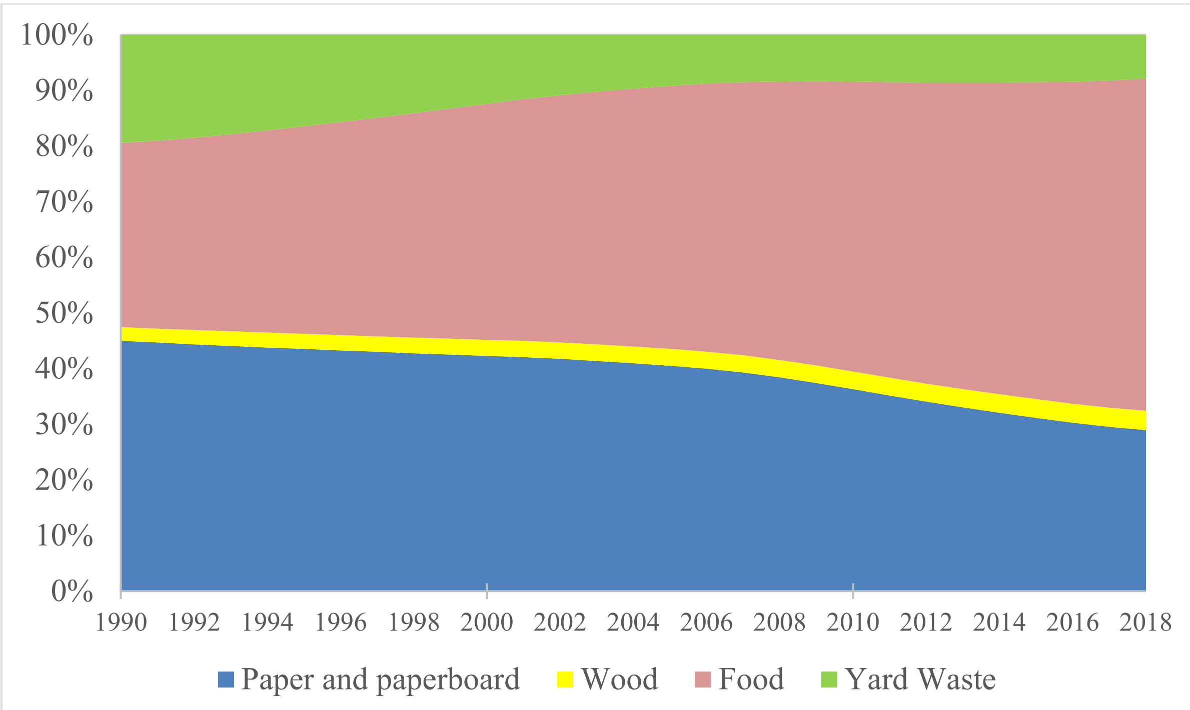


Figure 10.3. Sources of MSW Methane: 1990–2018 (Data in Appendix F)

The analysis of the role of forest products on methane generation is complicated by incomplete information on the amounts of construction and demolition (C&D) debris going to MSW landfills vs. the amounts going to dedicated construction and debris landfills. US EPA estimates that in 2018, about 37.1 million metric tons of wood waste in C&D debris was generated in the US, and 26.9 million metric tons were disposed in C&D landfills (US EPA 2020c). This does not include the 11.1 million metric tons of wood disposed in MSW landfills in 2018.

In considering methane emissions from C&D landfills, it is important to understand that the conditions in C&D debris landfills do not promote methane production. The fact that C&D landfills produce far less methane than MSW landfills is due to several factors. One important factor is that readily degradable materials are not placed in C&D landfills. Another important factor, however, is that C&D waste often contains considerable amounts of sulfates, primarily from gypsum board (US EPA 2020d), which puts sulfate-reducing bacteria at a competitive advantage to methane-producing bacteria (e.g., see Stefanie et al. 1994; Forester Media 2004). Indeed, US EPA observed that, in C&D landfills, other than odor-producing gases, "substantial quantities of [gases], such as CH_4, are not produced, which is a reflection of the types of waste normally deposited in C&D landfills" (US EPA 2014a). In addition, in the documentation for version 15 of US EPA's WARM, US EPA notes that in the case of landfilled gypsum board, "the sulfate in wallboard is estimated to reduce methane generation; thus, the methane yield from gypsum board is likely to be negligible and is therefore adjusted to 0%" (US EPA 2020b). These observations support the assumption, for purposes of this report, that methane emissions from wood products in C&D landfills are negligible and can be ignored.

As a result of increased recovery and diversion of degradable wastes, and intensified efforts to capture methane from MSW landfills, total methane releases from MSW landfills have been decreasing, as shown in Figure 10.4.

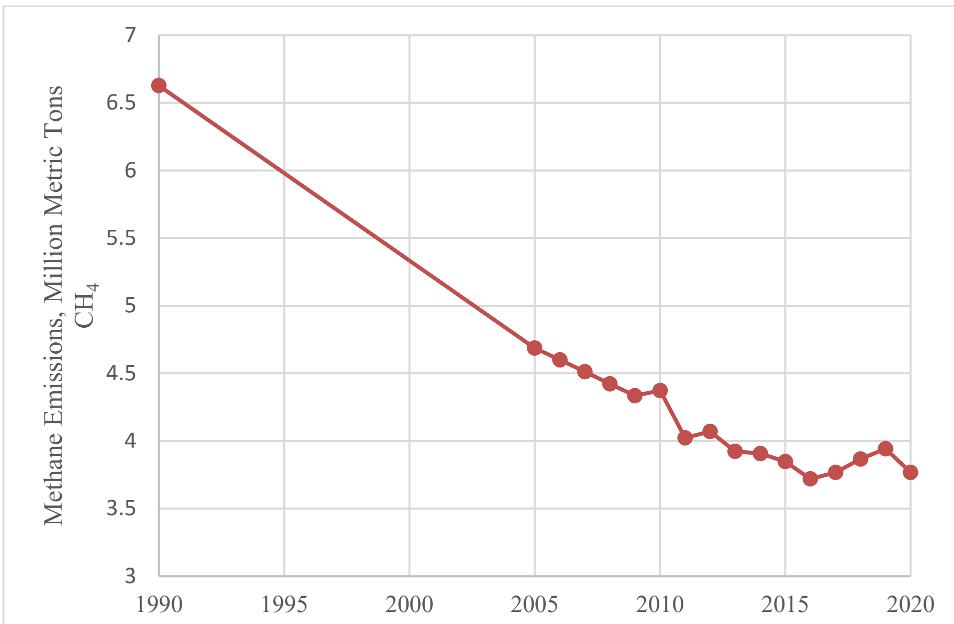


Figure 10.4. Methane Emissions from MSW Landfills in the US [*Source*: US EPA (2020b)]

Because total MSW methane emissions have been declining and the relative contribution of forest products has also been declining, total methane emissions attributable to forest products have decreased significantly since 1990. Methane emissions from MSW landfills attributable to forest products in 1990, 2005, and 2020 are shown in Table 10.1. The relative contribution of wood products is smaller than found in the previous profile (NCASI 2008; Heath et al. 2010). The primary reasons are likely to be (1) the differences in degradable carbon used in the calculations and (2) the reliance on US EPA's measurement-based estimates rather than model-only–based estimates. The previous profile assumed that 23% of carbon in wood products is degradable in MSW landfills, whereas this profile uses a value of 14%, representing a 40% decrease in degradable carbon attributable to wood products. The value used for the degradable fraction of carbon in paper and paperboard is also lower in this profile, but the reduction is much smaller than for wood products[24].

Table 10.1. Methane Releases from MSW and C&D Landfills Attributable to US Forest Products

Year	Million Metric Tons CO_2eq. Using GWP of 25 for Methane				Fraction of 1990	Fraction of 2005
	Paper and Paperboard	Wood Products	Wood in C&D Landfills[b]	Total		
1990	74.5	4.1	0	78.6	1.00	
2005	47.4	3.6	0	51.0	0.65	1.00
2020[a]	27.2	3.3	0	30.5	0.39	0.60

[a] 2020 estimates based on US EPA 2020 estimates of methane emissions and NCASI 2018 estimates of the fraction attributable to forest products. 2018 data are used to determine forest product contributions because these are the most recent disaggregated data available from US EPA.
[b] Assumed negligible. See text for explanation.

After adjusting for the different GWPs used for methane (25 in this profile vs. 21 in the previous profile), the previous profile estimates of methane emissions attributable to forest products can be compared to those earlier in this report. The earlier estimate for 1990 was 72.4 million metric tons CO_2eq. (NCASI 2008; Heath et al. 2010) compared to 78.6 million metric tons CO_2eq. in this profile. For 2005 emissions, the earlier estimate was 66.4 million metric tons CO_2eq. compared to 51.1 million metric tons CO_2eq. in this updated profile. There are several reasons for the differences beyond the change in GWP, the primary one being the use of different methods for the two profiles, as explained previously. It should be noted that the updated estimates include emissions from both domestic and non-domestic forest products in landfills, which means that, for reasons explained earlier, the result is likely an overstatement of emissions attributable to the US forest products sector.

10.2 Methane Emissions Associated with a Single Year's Production

In Section 3.4 of this report, the 100-year benefits of carbon stored in products are estimated. When looking at industry products in this way, it is also necessary to consider the 100-year release of methane associated with the decomposition of those same products when discarded in landfills.

The approach used to calculate the impact of methane emissions over 100 years requires the same input data used to calculate the value of stored carbon over 100 years. The data needed are described in the earlier section dealing with long-term storage of carbon in products and in Appendix A. The quantity of products removed from use each year was calculated using time-in-use data from Smith et al. (2005) and production data from Howard and Liang (2019). The fraction of discards placed in landfills was obtained from US EPA (2020e) and extrapolated based on professional judgment. The landfill model used degradation rates and fractions of nondegradable carbon from US EPA's WARM documentation (US EPA 2020b). Landfill gas capture rates for 1990 to 2020 were calculated as the ratio of MSW methane captured to MSW methane generated (US EPA 2022b). The annual rate of increase in percent captured from 2005 to 2020 was applied to capture

[24] The values from US EPA's WARM documentation, instead of other possible sources, are used here because this source provides consistently derived parameter values for difference components of the waste stream.

rates after 2020 until the rate reached 80%, judged to be the practical maximum as a national average. Natural oxidation was assumed to remove 10% of methane not captured. The products manufactured in 1990, 2005, and 2020 were followed for 100 years following manufacturing. The decomposition, methane generation, and methane release of each year's discards was modeled for each year.

Converting methane emissions to CO_2eq. was done using two approaches. First, the total emissions were multiplied by the 100-year GWP of methane, 25 (used throughout this study). However, this approach can overstate the radiative forcing caused by these emissions over 100 years because the emissions occur gradually instead of in the year of product manufacture. To calculate the radiative forcing, in CO_2eq., associated with the gradual release of methane, the annual methane releases were entered into the dynamic LCA calculator (described in Levasseur et al. 2010 and available online at https://ciraig.org/index.php/project/dynco2-dynamic-carbon-footprinter/). The calculations are described in detail in Appendix A.

The emissions over 100 years following manufacturing for products manufactured in 1990, 2005, and 2020 are shown in Table 10.2. The CO_2eq. results are shown in two ways. The middle section of the table shows the results of the dynamic radiative forcing calculations, while the right-hand section shows the result of simply multiplying the total 100-year emissions of methane by the GWP for methane of 25.

Table 10.2. Methane Emissions and Cumulative Radiative Forcing Associated with Products Manufactured in 1990, 2005, and 2020

	Million Metric Tons Methane			Cumulative Radiative Forcing, Million Metric Tons CO_2eq.			Million Metric Tons CO_2eq. Obtained by Multiplying the 100-Year Cumulative Methane Emissions by a GWP of 25		
M	Paper	Wood	Total	Paper	Wood	Total	Paper	Wood	Total
1990	2.67	0.85	3.52	65.6	20.5	86.1	66.7	21.3	88.0
2005	1.33	0.60	1.92	32.6	14.1	46.7	33.2	14.9	48.1
2020	0.55	0.44	0.99	13.4	10.3	23.8	13.7	11.0	24.7

Calculated 100-year methane emissions are much lower for production in more recent years. This is due to the reduced production in 2020, reduced fractions for discards going to landfill over time, and improved capture of landfill gas over time.

11.0 TOTAL FOREST PRODUCTS INDUSTRY EMISSIONS

The updated annual inventory emissions calculated earlier in this report are summarized in Table 11.1 and Figure 11.1. Later in this report, these estimates are compared to previously published estimates (NCASI 2008; Heath et al. 2010). In 2020, Scope 1 emissions from industry fuel combustion and Scope 2 emissions for manufacturing and converting operations accounted for 46.5% of the industry's value chain emissions. If all waste management and wood production emissions are assumed to be Scope 1, total Scope 1 and 2 emissions account for 54% of 2020 emissions. The other major contributors to value chain emissions are transport, accounting for 11% (not including transport of fuels to mills), and product end-of-life (specifically landfilling), accounting for 18%.

Table 11.1. Forest Products Industry Value Chain Gross Emissions in 1990, 2005, and 2020

Emission Category	Million Metric Tons CO_2eq.			% of 2020 Total
	1990[a]	2005[a]	2020	
Scope 1 Fuel Combustion	78.0	74.4	43.6	25%
Scope 2 (based on gross energy purchases)	64.0	68.8	36.4	21%
Waste Management	9.4	11.2	10.6	6%
Transport (not including fuels used at mills)	23.5	23.7	19.5	11%
Upstream – Purchased Electricity	3.7	4.7	4.0	2%
Upstream – Nonfiber input production	12.8	14.9	11.5	7%
Upstream – Fossil fuel production and transport	11.1	11.1	7.9	5%
Product Use (84 to 96% residential wood fuel emissions)	9.1	5.5	5.9	3%
Wood Supply Production	2.5	2.3	2.1	1%
Product End-of-Life	78.6	51.0	30.5	18%
Total Gross Value Chain Emissions	292.8	267.7	172.2	

[a] Based on new estimates rather than those in NCASI (2008) and Heath et al. (2010).

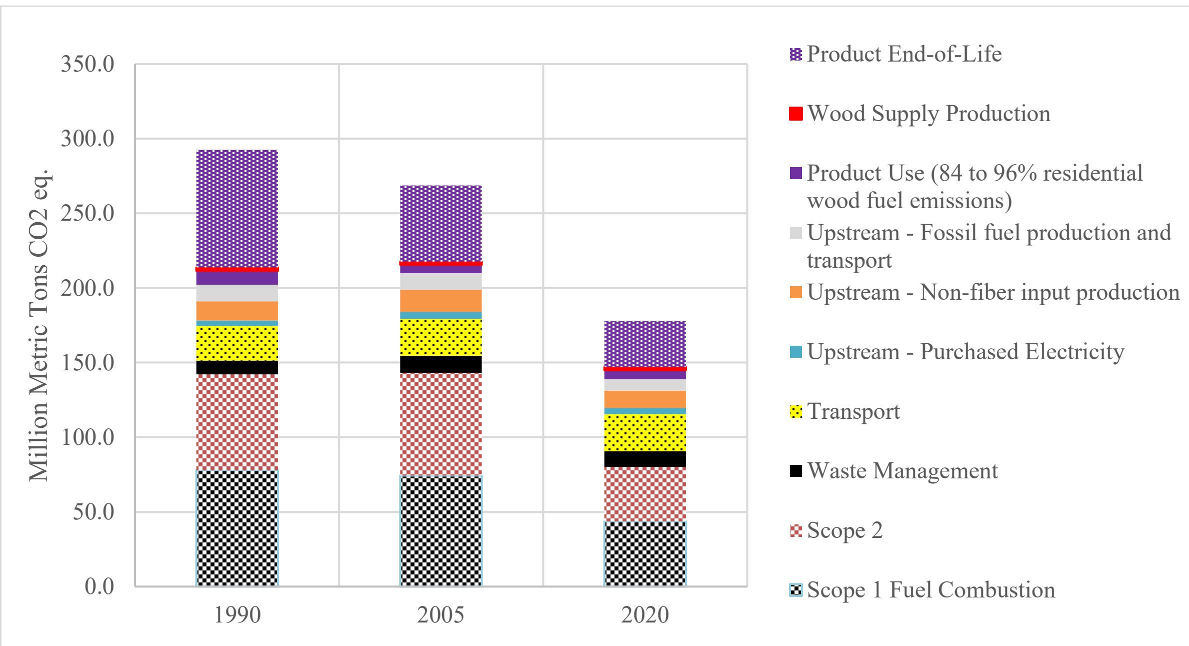


Figure 11.1. Forest Products Industry Value Chain Emissions in 1990, 2005, and 2020 [Note: 1990 and 2005 values based on new estimates rather than those in NCASI (2008) and Heath et al. (2010)]

Changes in emissions from 1990 and 2005 are shown in Table 11.2. The forest products industry's 2020 total emissions were 41% lower than 1990 emissions and 36% lower than 2005 emissions. This was driven almost entirely by reduced use of energy in manufacturing (fossil fuel and electricity consumption), the greening of the electricity grid, and reduced end-of-life emissions resulting from increased recycling and reduced landfilling of discarded forest products.

Table 11.2. Emissions in 2020 Relative to Emissions in 1990 and 2005

Emission Category	Fraction of 1990	Fraction of 2005
Scope 1 fuel combustion	0.56	0.59
Scope 2 (based on gross energy purchases)	0.57	0.53
Waste management	1.12	0.94
Transport	0.83	0.82
Upstream – Purchased electricity	1.08	0.86
Upstream – Nonfiber input production	0.89	0.77
Upstream – Fossil fuel production and transport	0.71	0.72
Product use (84 to 96% residential wood fuel emissions)	0.65	1.08
Wood supply production	0.84	0.92
Product end-of-life	0.39	0.60
Total	0.59	0.64

Note: Based on new estimates rather than those in NCASI (2008) and Heath et al. (2010).

12.0 NET TRANSFERS OF CARBON TO THE ATMOSPHERE FROM THE FOREST PRODUCTS VALUE CHAIN

Changes in forest ecosystem carbon stocks and forest product carbon stocks are part of the industry's profile. In Table 12.1, the impacts of these stock changes are added to gross emissions to allow calculations of net emissions values for 1990, 2005, and 2020. Like gross emissions, net emissions have declined over time, with net emissions in 2020 being 55% below 1990 and 53% below 2005 net emissions.

Table 12.1. Net Emissions Attributable to the Forest Products Industry Value Chain

Emission Category	Million Metric Tons CO_2eq.		
	1990[a]	2005[a]	2020
Forest ecosystem	0	0	0
Changes in product carbon stocks	−123.8	−106.0	−96.8
Scope 1 fuel combustion	78.0	74.4	43.6
Scope 2 (based on gross energy purchases)	64.0	68.8	36.4
Waste management	9.4	11.2	10.6
Transport	23.5	23.7	19.5
Upstream – Purchased electricity	3.7	4.7	4.0
Upstream – Nonfiber input production	12.8	14.9	11.5
Upstream – Fossil fuel production and transport	11.1	11.1	7.9
Product use (84 to 96% residential wood fuel emissions)	9.1	5.5	5.9
Wood supply production	2.5	2.3	2.1
Product end-of-life	78.6	51.0	30.5
Net value chain emissions	168.9	161.7	75.4

[a] Based on new estimates rather than those in NCASI (2008) and Heath et al. (2010).

13.0 COMPARISON TO 2008 PROFILE STUDY RESULTS

Estimates of emissions for 1990 and 2005, contained in the 2008 profile study (NCASI 2008; Heath et al. 2010), and those developed in this new report, are shown in Table 13.1. This new report includes estimates for several types of emissions not included in the 2008 study (e.g., upstream emissions associated with production of purchased electricity, emissions associated with fossil fuel production and transport, and emissions associated with product use). Where there are comparable estimates, the new estimates are generally higher than those in NCASI (2008) and Heath et al. (2010). For Scope 1 and 2 emissions, this is due primarily to having extended the new analysis to converting sectors not included in the 2008 analysis. This was especially impactful for emissions associated with purchased electricity. The other major differences are due to the use of improved methods and updated data in this study. Many of these differences are examined in more detail in earlier sections of this report addressing specific emissions estimates.

Table 13.1. Comparison of Estimates in 2008 Profile Study to Those in This Report

Emission Category	Million Metric Tons CO_2eq.			
	1990 Emissions		2005 Emissions	
	Original[a]	New	Original[a]	New
Changes in forest ecosystem carbon stocks	0	0	0	0
Changes in forest product carbon stocks	−132.6	−123.9	−108.5	−106.0
Scope 1 fuel combustion	74.1	78.0	62	74.4
Scope 2	42.4	64.0[b]	43.6	68.8[b]
Waste management	2.4[c]	9.4	3.1[c]	11.2
Transport	16.9	23.5	19.6	23.7
Upstream – Purchased electricity	–	3.7	–	4.7
Upstream – Nonfiber input production	24	12.8	24	14.9
Upstream – Fossil fuel production and transport	–	11.1	–	11.1
Product use	–	9.1	–	5.5
Wood supply production	4	2.5	4.2	2.3
Product end-of-life	72.4[c]	78.6	66.4[c]	51.0
Total gross emissions	236.4	292.8	223.2	267.7
Total net emissions	91.8	168.9	103.5	161.7

[a] Original estimates are those published in NCASI (2008) and Heath et al. (2010).
[b] Original estimates used net purchases to calculate Scope 2 emissions. The new values are based on gross purchases.
[c] Original estimates have been updated to a methane GWP of 25 from 21

14.0 SUMMARY AND CONCLUSIONS

This report contains the results of a GHG profile study of the US forest products industry. It updates an earlier study comparing 1990 and 2005 emissions and sinks (NCASI 2008; Heath et al. 2010) by adding 2020 emissions. It also expands the scope of the earlier study by including several types of emissions not previously addressed and uses the latest data and methods. In some cases, the methods and data are different from those used in the earlier profile; therefore, new estimates have been developed for 1990 and 2005. Table 14.1 provides an overview of the elements of the profile and the methods used to develop emissions estimates.

Table 14.1. Overview of Methods Used to Develop Estimates of Emissions and Sinks

Emission Category	Description
Changes in forest ecosystem carbon stocks	Analyses performed by the US Forest Service are used to conclude that while it is not possible to precisely estimate the effect of the industry's activities, there is no reason to suspect that the industry's activities are causing, over time, a decline in carbon stocks on lands used to produce the industry's wood.
Changes in forest product carbon stocks	Estimates developed by the US Forest Service and published by US EPA are used directly to determine actual year-to-year changes in carbon stocks in forest products in use and in landfills. Long-term storage of carbon in forest products manufactured in 1990, 2005, and 2020 has been estimated by calculating the impact of carbon in products on cumulative radiative forcing over 100 years.
Direct emissions from fuel combustion	Fuel consumption data obtained from surveys conducted by the industry and by the US government are converted into emissions using widely accepted emission factors.
Emissions from suppliers of electricity and steam used by the industry	Electricity consumption data obtained from surveys conducted by the industry and by the US government are converted into emissions based on data published by US EPA characterizing the national average grid characteristics.
Waste management	Waste production data obtained from industry surveys and life cycle studies are analyzed using a first order decay model and parameters used by IPCC and US EPA.
Transport	Quantities requiring transport are derived from a variety of published sources, and emissions are calculated using factors published by the US Census Bureau.
Upstream emissions for suppliers of purchased electricity	Upstream emissions associated with producing electricity purchased by the industry are estimated using factors published by DOE's National Energy Renewable Laboratory and data from US EPA on the sources of energy on the grid.
Upstream emissions associated with nonfiber inputs to production	Emissions associated with producing nonfiber, nonfuel inputs are estimated using factors derived from life cycle studies.
Upstream emissions associated with production and transport of fossil fuel used by the industry	Upstream emissions associated with producing and transporting fuels are estimated using factors derived from life cycle studies.
Product use	Emissions associated with using wood for fuel are estimated from industry and government data and widely used emission factors.
Wood supply production	Emissions from forestry operations associated with wood supply production are estimated using regional production data from the US Forest Service and factors developed by NCASI based on life cycle studies.
Product end-of-life	Emissions are based on US EPA's estimates of methane released from MSW landfills, allocated to different degradable types of waste (including forest products) using a model used by IPCC and US EPA. Long-term methane emissions attributable to forest products manufactured in 1990, 2005, and 2020 have been estimated by calculating the impact on cumulative radiative forcing over 100 years.

The results of this study are shown in Table 14.2. In 2020, direct GHG emissions from industry fuel combustion (Scope 1) and GHG emissions from suppliers of electricity and steam to manufacturing and converting operations (Scope 2) accounted for 46.5% of the industry's value chain emissions. If all industrial waste management and wood production emissions are included, the result, which can be thought of as total Scope 1 and 2 emissions, accounts for 54% of 2020 value chain emissions. The other major contributors to value chain emissions are transport, accounting for 11%, and product end-of-life (specifically landfilling), accounting for 18%.

The forest products industry's 2020 gross GHG emissions were 41% lower than 1990 emissions and 36% lower than 2005 emissions. This was driven almost entirely by reduced use of energy in manufacturing (fossil fuel and electricity consumption), the greening of the US electricity grid, and reduced end-of-life emissions resulting from increased recycling and reduced landfilling of discarded forest products.

Net GHG emissions in 2020, which consider net removals of CO_2 from the atmosphere attributable to the forest products industry value chain, were 55% and 53% lower than 1990 and 2005 net emissions, respectively. Over this time, growth in product carbon stocks (and net removals of CO_2 from the atmosphere attributable to that growth) declined significantly, primarily because of reduced production of paper and wood products. Annual changes in forest carbon stocks over time attributable to the forest products industry are conservatively assumed to be zero. This is based on findings that (1) carbon stocks on timberland are increasing and (2) data provide no reason to suspect that the industry's activities are causing reductions in forest carbon stocks over time.

The estimates of Scope 1 and 2 emissions developed in this study for 1990 and 2005 are generally higher than those developed in the earlier study (NCASI 2008; Heath et al. 2010). This is primarily due to (1) the inclusion of sources not included in the earlier study and (2) use of the latest data and methods to estimate the industry's profile.

Table 14.2. Gross and Net Emissions Attributable to the Forest Products Industry Value Chain

Emission Category	Million Metric Tons CO_2eq.		
	1990[a]	2005[a]	2020
Forest ecosystem	0	0	0
Changes in product carbon stocks	−123.9	−106.0	−96.8
Scope 1 fuel combustion	78.0	74.4	43.6
Scope 2 (based on gross energy purchases)	64.0	68.8	36.4
Waste management	9.4	11.2	10.6
Transport	23.5	23.7	19.5
Upstream – Purchased electricity	3.7	4.7	4.0
Upstream – Nonfiber input production	12.8	14.9	11.5
Upstream – Fossil fuel production and transport	11.1	11.1	7.9
Product use (84 to 96% residential wood fuel emissions)	9.1	5.5	5.9
Wood supply production	2.5	2.3	2.1
Product end-of-life	78.6	51.0	30.5
Gross value chain emissions	292.8	267.7	172.2
Net value chain emissions	168.9	161.7	75.4

[a] Based on new estimates rather than those in NCASI (2008) and Heath et al. (2010).

The values in Table 14.2 reflect the transfers of carbon to and from the atmosphere in the year of the inventory (i.e., in 1990, 2005, and 2020) attributable to current and past production. The impact of carbon in products, however, can also be characterized by examining the projected long-term effects attributable to production in a single year. In this study, long-term carbon storage associated with a single year's production has been calculated based on the net radiative forcing over 100 years attributable to additions to, and gradual losses from, stocks of carbon in products put into use in 1990, 2005, and 2020. For the purposes of this report, the radiative forcing-based metric is called "effective long-term carbon storage." The results are shown in Table 14.3. Tables 14.4, 14.5, and 14.6 show the amounts of carbon remaining in different pools 100 years after the product was manufactured.

Table 14.3. Total Effective Long-Term Carbon Storage
(Shown in Carbon Equivalents and CO_2 Equivalents [in parentheses])

Year	Million Metric Tons in Net Radiative Forcing		
	Paper	Wood	Total
1990	13.26 (48.6)	31.89 (116.9)	45.15 (165.1)
2005	10.11 (37.1)	32.48 (119.1)	42.59 (156.2)
2020	5.90 (21.6)	28.96 (106.2)	34.86 (127.8)

Table 14.4. Total Carbon Remaining Stored in Products 100 Years Following Manufacturing

Year	Million Metric Tons C		
	Paper	Wood	Total
1990	11.12	28.63	39.75
2005	8.03	29.05	37.08
2020	4.33	25.92	30.25

Table 14.5. Carbon Remaining Stored in Products in Use 100 Years Following Manufacturing

Year	Million Metric Tons C		
	Paper	Wood	Total
1990	0.00	6.67	6.67
2005	0.00	7.66	7.66
2020	0.00	6.35	6.35

Table 14.6. Carbon Remaining Stored in Products in Landfills 100 Years Following Manufacturing

Year	Million Metric Tons C		
	Paper	Wood	Total
1990	11.12	21.97	33.09
2005	8.03	21.39	29.42
2020	4.33	19.57	23.90

This long-term analysis of the impacts of a single year's production also requires assessment of the long-term emissions of methane from landfills. Landfill emissions, projected over 100 years following manufacturing, are shown in Table 14.7. For products manufactured in 2020, cumulative forcing from methane emissions offset 62% of the effective long-term storage benefits, shown in Table 75, for paper products, 10% for wood products, and 19% for all industry products.

Table 14.7. Methane Emissions and Related Cumulative Radiative Forcing Associated with Products Manufactured in 1990, 2005, and 2020

Year	Million Metric Tons Methane			Cumulative Radiative Forcing, Million Metric Tons CO_2eq.			Million Metric Tons CO_2eq. Obtained by Multiplying the 100-Year Cumulative Methane Emissions by a GWP of 25		
	Paper	Wood	Total	Paper	Wood	Total	Paper	Wood	Total
1990	2.67	0.85	3.52	65.6	20.5	86.1	66.7	21.3	88.0
2005	1.33	0.60	1.92	32.6	14.1	46.7	33.2	14.9	48.1
2020	0.55	0.44	0.99	13.4	10.3	23.8	13.7	11.0	24.7

REFERENCES

Abt, Karen L., Robert C. Abt, Christopher S. Galik, and Kenneth E. Skog. 2014. Effect of Policies on Pellet Production and Forests in the U.S. South: A Technical Document Supporting the Forest Service Update of the 2010 RPA Assessment. Gen. Tech. Rep. SRS-202. Asheville, NC: US Department of Agriculture Forest Service, Southern Research Station. https://doi.org/10.2737/srs-gtr-202.

AF&PA (American Forest and Paper Association). 2020. 2019 Statistics, Paper, Paperboard & Pulp. Washington, DC.

———. 2021a. 2020 Statistics, Paper, Paperboard & Pulp. Washington, DC.

———. 2021b. Capacity and Fiber Consumption – 61st Annual – 2020–2021. Washington, DC.

AHFA (American Home Furnishings Alliance). 2022. 2021 Home Furnishings Industry Overview. https://www.ahfa.us/media/industry-facts.

Alderman, Delton. 2022. United States Forest Products Annual Market Review and Prospects, 2015–2021. General Technical Report FPL-GTR-289. Madison, WI: US Department of Agriculture, Forest Service, Forest Products Laboratory. https://www.fpl.fs.usda.gov/documnts/fplgtr/fpl_gtr289.pdf.

Barrett, Scott, M. Chad Bolding, and John Munsell. 2017. "Characteristics of Logging Businesses Across Virginia's Diverse Physiographic Regions." *Forests* 8, no. 12: 468. https://doi.org/10.3390/f8120468.

BEA (Bureau of Economic Analysis). 2024. Data Tools. US Department of Commerce. https://www.bea.gov/tools.

Bergman, Richard D., and Scott A. Bowe. 2008. Life-Cycle Inventory of Hardwood Lumber Manufacturing in the Northeast and North Central United States. https://corrim.org/wp-content/uploads/2018/03/lci-hardwood-lumber-ne-nc-us.pdf.

Campbell, Alton G. 1990. "Recycling and Disposing of Wood Ash." *Tappi Journal* 73, no. 9: 141–46.

Conrad, Joseph L. 2018. "The Evolution of Logging Businesses in Georgia 1987–2017 and South Carolina 2012–2017." *Forest Science* 64, no. 6: 671–81. https://doi.org/10.1093/forsci/fxy020.

———. 2021. "Evaluating Profitability of Individual Timber Deliveries in the US South." *Forests* 12, no. 4: 437. https://doi.org/10.3390/f12040437.

———. 2023. "Log Truck Insurance Premiums, Claims and Safety Practices Among Logging Business in the US South." *International Journal of Forest Engineering* 34, no. 2: 204–15. https://doi.org/10.1080/14942119.2022.2124704.

Davis, Stacy C., and Robert G. Boundy. 2022. Transportation Energy Data Book Edition 40. Report ORNL/TM-2022/2376. Oak Ridge National Laboratory. https://tedb.ornl.gov/wp-content/uploads/2022/03/TEDB_Ed_40.pdf.

EIA (US Energy Information Administration). 2023a. Manufacturing Energy Consumption Survey. https://www.eia.gov/consumption/manufacturing/.

———. 2023b. Real Prices Viewer. https://www.eia.gov/outlooks/steo/realprices/.

EPRI (Electric Power Research Institute). 1979. Coal Ash Disposal Manual. FP-1257, Research Project 1404-1. Prepared by GAI Consultants. https://doi.org/10.2172/5310803.

FAO, ITTO, and United Nations. 2020. *Forest Product Conversion Factors*. Rome, Italy: FAO, ITTO, and United Nations. https://doi.org/10.4060/ca7952en.

FAOSTAT. 2023. Food and Agriculture Data. https://www.fao.org/faostat/en.

Fearnside, Philip M., Daniel A. Lashof, and Pedro Moura-Costa. 2000. "Accounting for Time in Mitigating Global Warming Through Land-Use Change and Forestry." *Mitigation & Adaptation Strategies for Global Change* 5, no. 3: 239–70. https://doi.org/10.1023/A:1009625122628.

Forester Media. 2004. Hydrogen Sulfide from Landfilled Construction-and-Demolition Debris: When and How. https://www.mswmanagement.com/collection/article/13002757/hydrogen-sulfide-from-landfilled-constructionanddemolition-debris-when-and-how.

GHG Protocol. 2004. *GHG Protocol Corporate Standard*. World Resources Institute and World Business Council for Sustainable Development.

GHG Protocol. 2011. *Corporate Value Chain (Scope 3) Accounting and Reporting Standard*. World Resources Institute and World Business Council for Sustainable Development.

Gomez, Dario R., John D. Watterson, Branca B. Americano, Chia Ha, Gregg Marland, Emmanuel Matsika, Lemmy Nenge Namayanga, Balgis Osman-Elasha, John D. Kalenga Saka, and Karen Treanton. 2006. "Chapter 2: Stationary Combustion." In *2006 IPCC Guidelines for National Greenhouse Gas Inventories*, volume 2. Institute for Global Environmental Strategies. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/2_Volume2/V2_2_Ch2_Stationary_Combustion.pdf.

Heath, Linda S., Van Maltby, Reid Miner, Kenneth E. Skog, James E. Smith, Jay Unwin, and Brad Upton. 2010. "Greenhouse Gas and Carbon Profile of the US Forest Products Industry Value Chain." *Environmental Science & Technology* 44, no. 10: 3999–4005. https://doi.org/10.1021/es902723x.

Howard, James L., and Shaobo Liang. 2019. U.S. Timber Production, Trade, Consumption, and Price Statistics, 1965–2017. Research Paper FPL-RP-701. Madison, WI: United States Department of Agriculture, Forest Service, Forest Products Laboratory.

IMO (International Marine Organization). 2020. Fourth IMO Greenhouse Gas Study. London, UK. https://wwwcdn.imo.org/localresources/en/OurWork/Environment/Documents/Fourth%20IMO%20GHG%20Study%202020%20-%20Full%20report%20and%20annexes.pdf.

LaGore, Rick. 2020. "What is the Legal Weight Shippers are Permitted to Load Cargo?" InTek, September 1. https://blog.intekfreight-logistics.com/what-is-legal-weight-shippers-can-load-53-trailers.

Levasseur, Annie, Pascal Lesage, Manuele Margni, Louise Deschênes, and Réjean Samson. 2010. "Considering Time in LCA: Dynamic LCA and Its Application to Global Warming Impact Assessments." *Environmental Science & Technology* 44, no. 8: 3169–74. https://doi.org/10.1021/es9030003.

Malyan, Sandeep K., Arti Bhatia, Amit Kumar, Dipak Kumar Gupta, Renu Singh, Smita S. Kumar, Ritu Tomer, O. Kumar, and Niveta Jain. 2016. "Methane Production, Oxidation and Mitigation: A Mechanistic Understanding and Comprehensive Evaluation of Influencing Factors." *Science of the Total Environment* 572: 874–96. https://doi.org/10.1016/j.scitotenv.2016.07.182.

McKeever, David. 2003. "Taking Inventory of Woody Residuals." *BioCycle* (July): 31–5.

Milota, Mike. 2020a. Life Cycle Assessment for the Production of Pacific Northwest Softwood Lumber. Consortium for Research on Renewable Industrial Materials (CORRIM). https://corrim.org/wp-content/uploads/2020/06/CORRIM-AWC-PNW-Lumber.pdf.

———. 2020b. Life Cycle Assessment for the Production of Southeastern Softwood Lumber. Consortium for Research on Renewable Industrial Materials (CORRIM). https://corrim.org/wp-content/uploads/2020/06/CORIRM-AWC-SE-Lumber.pdf.

Miner, R., and B. Upton. 2002. "Methos for Estimating Greenhouse Gas Emissions from Lime Kilns at Kraft Pulp Mills." *Energy* 27, no. 8: 729–38. https://doi.org/10.1016/S0360-5442(02)00017-8.

Miner, Reid, and Jay Unwin. 1991. "Progress in Reducing Water Use and Wastewater Loads in the U.S., Paper Industry." *Tappi Journal* 74, no. 8: 127–31.

NCASI (National Council for Air and Stream Improvement, Inc.). 1992. Solid Waste Management and Disposal Practices in the US Paper Industry. Technical Bulletin No. 641. Cary, NC: National Council for Air and Stream Improvement, Inc.

———. 1999a. Solid Waste Management and Disposal Practices in the US Paper Industry−1995. Technical Bulletin No. 793. Cary, NC: National Council of the Paper Industry for Air and Stream Improvement, Inc.

———. 1999b. A Summary of Available Data on the Chemical Composition of Forest Products Industry Solid Wastes. Special Report No. 99-04. Cary, NC: National Council of the Paper Industry for Air and Stream Improvement, Inc.

———. 2008. The Greenhouse Gas and Carbon Profile of the US Forest Products Sector. Special Report No. 08-05. Cary, NC: National Council for Air and Stream Improvement, Inc.

———. 2010. Life Cycle Assessment of North American Printing and Writing Paper Products [Unpublished report]. Cary, NC: National Council for Air and Stream Improvement, Inc.

———. 2017. Life Cycle Assessment of US Average Corrugated Product – Final Report. Report prepared for the Corrugated Product Alliance (CPA). Cary, NC: National Council for Air and Stream Improvement, Inc. http://www.corrugated.org/wp-content/uploads/PDFs/LCA/NCASI_2014_LCA_Final.pdf.

———. 2019. Solid Residual Generation and Beneficial Use, and Wastewater Treatment Performance and Practices of the North American Pulp and Paper Industry. Technical Bulletin No. 1063. Cary, NC: National Council for Air and Stream Improvement, Inc.

———. 2021. Greenhouse Gas Reductions for the US Pulp and Paper Industry. White Paper (WP-21-07). Cary, NC: National Council for Air and Stream Improvement, Inc. https://www.ncasi.org/resource/greenhouse-gas-reductions-for-the-us-pulp-and-paper-industry/

———. 2023a. Scope 3 GHG Screening Tool, version 1.1a. Cary, NC: National Council for Air and Stream Improvement, Inc. https://www.ncasi.org/resource/scope-3-greenhouse-gas-screening-tool/.

———. 2023b. Forest Industry Carbon Assessment Tool (FICAT). Cary, NC: National Council for Air and Stream Improvement, Inc. https://www.ncasi.org/resource/forest-industry-carbon-assessment-tool/.

NREL (National Renewable Energy Laboratory). 2021. Life Cycle Greenhouse Gas Emissions from Electricity Generation: Update. https://www.nrel.gov/docs/fy21osti/80580.pdf.

Oneil, Elaine, and Maureen E. Puettmann. 2017. "A Life-Cycle Assessment of Forest Resources of the Pacific Northwest, USA." *Forest Products Journal* 67, no. 5–6: 316–30. https://doi.org/10.13073/FPJ-D-17-00011.

Oswalt, Sonja N., W. Brad Smith, Patrick D. Miles, and Scott A. Pugh. 2019. Forest Resources of the United States, 2017: A Technical Document Supporting the Forest Service 2020 RPA Assessment. Gen. Tech. Rep. WO-97. Washington, DC: US Department of Agriculture, Forest Service, Washington Office. https://doi.org/10.2737/WO-GTR-97.

Pipatti, Riitta, Per Svardal, Joao Wagner Silva Alves, Qingxian Gao, Carlos López Cabrera, Katarina Mareckova, Hans Oonk, Elizabeth Scheehle, Chhemendra Sharma, Alison Smith, and Masato

Yamada. 2006. "Chapter 3: Solid Waste Disposal." In *2006 IPCC Guidelines for National Greenhouse Gas Inventories,* volume 5. Institute for Global Environmental Strategies. https://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/5_Volume5/V5_3_Ch3_SWDS.pdf.

Puettmann, Maureen, Dominik Kaestner, and Adam Taylor. 2020a. Life Cycle Assessment for the Production of PNW Softwood Plywood. Consortium for Research on Renewable Industrial Materials (CORRIM). https://corrim.org/wp-content/uploads/2020/06/CORRIM-AWC-PNW-Plywood.pdf.

———. 2020b. Life Cycle Assessment for the Production of Southeast Softwood Plywood. Consortium for Research on Renewable Industrial Materials (CORRIM). https://corrim.org/wp-content/uploads/2020/06/CORRIM-AWC-SE-Plywood.pdf.

———. 2020c. Life Cycle Assessment for the Production of Oriented Strand Board. Consortium for Research on Renewable Industrial Materials (CORRIM). https://corrim.org/wp-content/uploads/2020/12/CORRIM-AWC-OSB-Final.pdf.

Puettmann, Maureen, Francesca Pierobon, Indroneil Ganguly, Hongmei Gu, Cindy Chen, Shaobo Liang, Susan Jones, Ian Maples, and Mark Wishnie. 2021. "Comparative LCAs of Conventional and Mass Timber Buildings in Regions with Potential for Mass Timber Penetration." *Sustainability* 13, no. 24: 3987. https://doi.org/10.3390/su132413987.

RTI International. 2015. Memorandum: Review of Site-Specific Industrial Waste Degradable Organic Content Data. Docket EPA-HQ-OAR-2015-0526-0019. https://www.regulations.gov/document/EPA-HQ-OAR-2015-0526-0019.

Smith, James E., Linda S. Heath, Kenneth E. Skog, and Richard A. Birdsey. 2005. Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States. General Technical Report NE-343. Newton Square, PA: US Department of Agriculture, Forest Service.

Someshwar, Arun V. 1996. "Wood and Combination Wood-Fired Boiler Ash Characterization." *Journal of Environmental Quality* 25, no. 5, Sept.–Oct.: 962–72. https://doi.org/10.2134/jeq1996.00472425002500050006x.

Staley, Bryan, and Debra Kantner. 2018. "Quantification of Municipal Solid Waste Management in the United States – With Comparative Analysis to Other Estimates." *Detritus* 3: 167–70. https://doi.org/10.31025/2611-4135/2018.13699.

Staley, Bryan F., and Morton A. Barlaz. 2009. "Composition of Municipal Solid Waste in the United States and Implications for Carbon Sequestration and Methane Yield." *Journal of Environmental Engineering* 135, no. 10: 901–9. https://doi.org/10.1061/(ASCE)EE.1943-7870.0000032.

Stefanie, J.W.H. Oude Elferink, Andre Visser, Look W. Hulshoff Pol, and Alfons J.M. Stams. 1994. "Sulfate Reduction in Methanogenic Bioreactors." *FEMS Microbiology Reviews* 15, no. 2–3: 119–36.

US Census Bureau. 2023. The 2017 Commodity Flow Survey Final Tables. https://www.census.gov/data/tables/2017/econ/cfs/aff-2017.html.

US DA (US Department of Agriculture). 2023. Global Agricultural Trade System Online. https://apps.fas.usda.gov/Gats/default.aspx.

US DOT (US Department of Transportation). 2024. Class I Rail Freight Fuel Consumption and Travel. https://www.bts.gov/content/class-i-rail-freight-fuel-consumption-and-travel.

US EPA (United States Environmental Protection Agency). 2010. Technical Support Document for the Industrial Waste Landfill Sector: Final Rule for Mandatory Reporting of Greenhouse Gases. https://www.epa.gov/sites/default/files/2015-07/documents/tsd_landfills_epa_02_04_09_2.pdf.

———. 2014a. Best Management Practices to Prevent and Control H2S and Reduced Sulfur Gases at Landfills That Dispose of Gypsum Drywall. EPA/600/R-14/039. Washington, DC.

———. 2014b. Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Facts and Figures – A Methodology Document. https://www.epa.gov/sites/default/files/2018-03/documents/methodolgy_document_for_selected_municipal_solid_waste_products.pdf.

———. 2014c. Emission Factors for Greenhouse Gas Inventories. https://www.epa.gov/sites/default/files/2015-07/documents/emission-factors_2014.pdf.

———. 2015. Annexes to Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2013. EPA 430-R-14-004. Washington, DC. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2013.

———. 2016. Greenhouse Gas Inventory Guidance: Direct Emissions from Stationary Combustion Sources Climate Leadership Program. https://www.epa.gov/sites/default/files/2016-03/documents/stationaryemissions_3_2016.pdf.

———. 2020a. Annexes to Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2018. EPA 430-R-20-002. Washington, DC. https://www.epa.gov/ghgemissions/draft-inventory-us-greenhouse-gas-emissions-and-sinks-1990-2018.

———. 2020b. WARM Documentation – Documentation for Greenhouse Gas Emission and Energy Factors Used in the Waste Reduction Model. https://www.epa.gov/sites/default/files/2020-12/documents/warm_management_practices_v15_10-29-2020.pdf.

———. 2020c. Advancing Sustainable Materials Management: 2018 Fact Sheet. https://www.epa.gov/sites/default/files/2021-01/documents/2018_ff_fact_sheet_dec_2020_fnl_508.pdf.

———. 2020d. Construction and Demolition Debris Management in the United States. https://www.epa.gov/sites/default/files/2020-03/documents/final_cd-eol-management_2015_508.pdf.

———. 2020e. Advancing Sustainable Materials Management: 2018 Tables and Figures. https://www.epa.gov/sites/default/files/2021-01/documents/2018_tables_and_figures_dec_2020_fnl_508.pdf.

———. 2021. Annexes to Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2019. EPA 430-R-21-005. Washington, DC. https://www.epa.gov/ghgemissions/draft-inventory-us-greenhouse-gas-emissions-and-sinks-1990-2019.

———. 2022a. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2020. EPA 430-R-22-003. Washington, DC. https://www.epa.gov/ghgemissions/draft-inventory-us-greenhouse-gas-emissions-and-sinks-1990-2020.

———. 2022b. Annexes to Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2020. EPA 430-R-22-003. Washington, DC. https://www.epa.gov/ghgemissions/draft-inventory-us-greenhouse-gas-emissions-and-sinks-1990-2020.

———. 2023a. Greenhouse Gas Reporting Program (GHGRP). https://www.epa.gov/ghgreporting.

———. 2023b. GHG Emissions by Industry – GHGRP vs. US GHG Inventory. https://www.epa.gov/sites/default/files/2015-07/documents/ghgrp_inventory_emissions_comparison.pdf.

———. 2023c. Emissions & Generation Resource Integrated Database (eGRID). https://www.epa.gov/egrid.

———. 2023d. Quantifying Methane Emissions from Landfilled Food Waste. EPA-600-R-23-064. Washington, DC.

———. 2024a. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2022. EPA 430-R-24-003. Washington, DC. https://www.epa.gov/ghgemissions/draft-inventory-us-greenhouse-gas-emissions-and-sinks-1990-2022.

———. 2024b. Annexes to Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2022. EPA 430-R-24-003. Washington, DC. https://www.epa.gov/ghgemissions/draft-inventory-us-greenhouse-gas-emissions-and-sinks-1990-2022.

Wang, Xiaoming, and Jennifer M. Padgett. 2011. "Wood Biodegradation in Laboratory-Scale Landfills." *Environmental Science & Technology* 45, no. 16: 6864–71. https://doi.org/10.1021/es201241g.

Whitaker, Michael, Garvin A. Heath, Patrick O'Donoughue, and Martin Vorum. 2012. "Life Cycle Greenhouse Gas Emissions of Coal-Fired Electricity Generation – Systematic Review and Harmonization." *Journal of Industrial Ecology* 16, no. s1. https://doi.org/10.1111/j.1530-9290.2012.00465.x.

APPENDIX A

CALCULATING CARBON STORAGE AND METHANE EMISSIONS, ATTRIBUTABLE TO A SINGLE YEAR'S PRODUCTION, OVER 100 YEARS

Carbon Storage in Products

In this study, carbon storage is calculated for products in use and in landfills. The calculations start with the amount of carbon in the specified year's production. This is put into use in the year of production and then reduced annually by the amount discarded, as explained in this appendix, to yield the amount remaining in use. The fraction of discards going to landfill is determined, and the losses of carbon due to decomposition in the landfill are modeled over time. With these calculations, the stocks remaining in use and in landfills are determined for each year in the 100-year period following manufacturing. The two pools are added to calculate the total stored carbon stocks for each year. Year-over-year losses in total stocks are treated as emissions.

The impacts of stored carbon can be presented in several ways. Two approaches are used in this study. The first approach simply reports the amounts of carbon remaining stored after 100 years following manufacturing. The second approach considers the effect of emissions timing on the warming caused over the 100 years following manufacturing. This second approach calculates the total cumulative radiative forcing associated with the gradual losses of stocks of stored product carbon over the 100 years after manufacturing and converts this into carbon equivalents. The total effective long-term storage is then calculated as the carbon in original product minus the amount of carbon equivalents lost from product stocks during the 100 years following manufacturing.

Carbon Storage in Use

The time-in-use data in Table 8 of Smith et al. (2005) for wood products were assumed to be appropriate for products produced in all years. For paper products, however, the values in Table 8 in Smith et al. (2005) were not used directly because they include the effect of recycling. Specifically, they assume "a half-life of 2.6 years, a paper recovery rate of 0.48, and an efficiency of reuse of 0.70" (Smith et al. 2005, 194). As a result, the Table 8 values for fraction of paper remaining in use are higher than associated with a first order half-life of 2.6 years for paper products and represent an estimate of virgin fiber half-life rather than product half-life. In addition, the values are tied to a single recovery rate. In this study, however, we (1) need to allow for varying recovery rates and (2) are interested in the amount of carbon stored in products in use, regardless of whether the fiber is virgin or recycled. Therefore, instead of the values in Smith et al. (2005) Table 8, the half-life for paper products (2.6 years) was used to calculate total discards. Each year, the amount remaining in carbon stocks in use was calculated by subtracting annual discards from the previous year's stocks.

Carbon Storage in Landfills

Products from a single year's production are removed from use over time (as described earlier), and those placed in landfills decay over time. Therefore, landfill carbon stocks need to be modeled considering varying input and output over time. Smith et al. (2005) values for landfill carbon were not used because the fate of discarded products and landfilling practices are held constant in the Smith et al. (2005) calculations.

The fraction of discards placed in landfills varies by year and was calculated using data from US EPA (2020e)[1,2]. US EPA data were used for values from 1990 to 2018. Because the fate of carbon over 100 years after production is to be calculated, the trends in the US EPA data needed to be extrapolated. An exponential model was used for this purpose. The extrapolations and model parameters are shown in Figure A1. The model included a lower limit on the fraction landfilled. Based on professional judgment, the lower limit on the fraction of discards landfilled was set at 0.10 for paper and 0.7 for wood products.

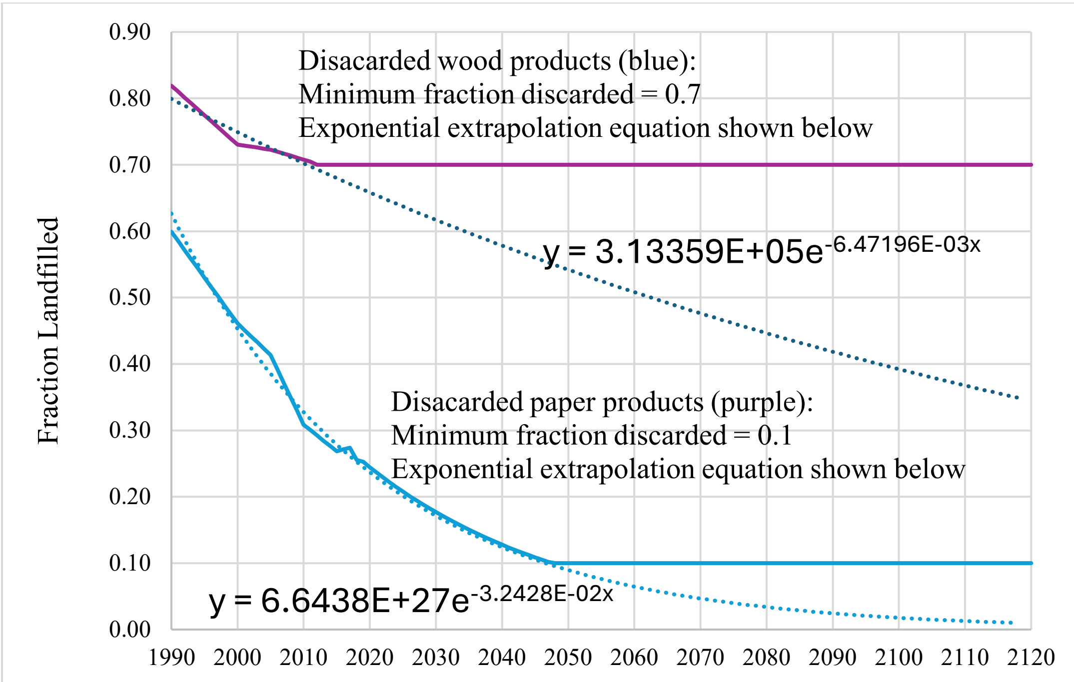


Figure A1. Model Used to Determine the Fraction of Paper and Wood Product Discards Sent to Landfills

Landfilled discards of products manufactured in 1990, 2005, and 2020 (or 2018 for wood products) were modeled for 100 years using a first order decay model. Each year's discards were modeled separately. Values for decay rates (or half-lives) and fraction of organic carbon that can degrade in anaerobic landfills were taken from US EPA (2020b). Using the simple average of products in US EPA (2020b), the rate constant, k, for paper products was calculated to be 0.0583 yr^{-1} (half-life of 11.9 years) and that for wood, 0.0950 yr^{-1} (half-life of 7.3 years). The carbon content of discarded paper and wood products was assumed to be 50%. Based on US EPA (2020b), the fraction of carbon in products degradable in landfills was calculated to be 0.44 for paper and 0.14 for wood products. The model was used to calculate the carbon stocks in landfills each year, for 100 years following the year of production.

Total Carbon Stored in Products

The annual stock changes in carbon in products in use and in landfills were summed to obtain the total annual loss of carbon contained in the original product for 100 years after manufacturing. The impact of

[1] All landfilling of forest products was assumed to occur at MSW landfills. C&D debris landfills were not addressed separately for wood products due to a lack of data on decay rates and decomposable carbon in C&D landfills. This approach has an unknown impact on carbon storage calculations.

[2] US EPA data for paper and paperboard are not adjusted to account for paper and paperboard that is used in packaging for imported goods. This may result in an overstatement of the fraction of domestically produced paper and paperboard recovered for recycling and an understatement of the fraction disposed. For more details, see the discussion in the text of this report and US EPA (2014b).

storage is presented in two ways. First, the amounts of carbon remaining stored after 100 years are presented. Second, the storage is calculated as the net cumulative radiative forcing, in carbon equivalents, caused by the addition to, and losses of, stocks of carbon over 100 years. In this report, the result of the second approach is given the label "effective long-term storage."

The impacts of emissions on radiative forcing were developed using version 2.0 of the dynamic LCA calculator (described in Levasseur et al. (2010) and available online at https://ciraig.org/index.php/project/dynco2-dynamic-carbon-footprinter/). Table A1 shows the calculations. The dynamic LCA calculator is used to calculate the cumulative radiative forcing, in watts per square meter, associated with the carbon (as CO_2) released in year zero of one kg of CO_2 (column 2 in the table). The third column shows the remaining length of time within a 100-year window after manufacturing. The fourth column shows the cumulative radiative forcing, over 100 years, associated with the metric tons of C in CO_2 released in each year.

Over 100 years, the factors in the final column are multiplied by the amount of carbon lost from total carbon stocks each year. The results are summed, providing the total cumulative warming, in watts per square meter, associated with the stock losses during the 100-year period following production. This is converted to carbon equivalents by dividing by 2.50121E-11, which is the cumulative watts per square meter associated with emission of an amount of CO_2 containing one metric ton of C. The amount of stored carbon is calculated by subtracting these carbon equivalent emissions from the carbon in the original product. To convert the result to CO_2eq. the sum is multiplied by 44/12.

Table A1. Calculation of Radiative Forcing Associated with Gradual Emissions of Product Carbon

Year (0 to 100)	1 kg CO_2 Release in Year Zero. Cumulative Watts per m^2 Over Time – from Dynamic LCA Calculator Ver. 2.0	Years in Atm. = 100 Minus Year	Cumulative Watts per m^2 Over Remaining Years for 1 Metric Ton C in CO_2 Released in This Year
0	0	100	2.50121E-11
1	1.69238E-15	99	2.48162E-11
2	3.28106E-15	98	2.462E-11
3	4.78495E-15	97	2.44232E-11
4	6.21912E-15	96	2.4226E-11
5	7.59558E-15	95	2.40283E-11
6	8.9239E-15	94	2.38301E-11
7	1.02117E-14	93	2.36314E-11
8	1.14652E-14	92	2.34323E-11
9	1.26892E-14	91	2.32326E-11
10	1.38877E-14	90	2.30323E-11
11	1.50638E-14	89	2.28316E-11
12	1.62202E-14	88	2.26303E-11
13	1.7359E-14	87	2.24285E-11
14	1.84817E-14	86	2.22261E-11
15	1.959E-14	85	2.20231E-11
16	2.06848E-14	84	2.18196E-11
17	2.17672E-14	83	2.16154E-11
18	2.28379E-14	82	2.14107E-11
19	2.38977E-14	81	2.12053E-11
20	2.49472E-14	80	2.09994E-11
21	2.59867E-14	79	2.07928E-11
22	2.70169E-14	78	2.05855E-11
23	2.8038E-14	77	2.03776E-11
24	2.90504E-14	76	2.0169E-11
25	3.00545E-14	75	1.99597E-11
26	3.10504E-14	74	1.97498E-11
27	3.20385E-14	73	1.95391E-11
28	3.3019E-14	72	1.93277E-11
29	3.39921E-14	71	1.91155E-11

Year (0 to 100)	1 kg CO_2 Release in Year Zero. Cumulative Watts per m^2 Over Time – from Dynamic LCA Calculator Ver. 2.0	Years in Atm. = 100 Minus Year	Cumulative Watts per m^2 Over Remaining Years for 1 Metric Ton C in CO_2 Released in This Year
30	3.4958E-14	70	1.89026E-11
31	3.59169E-14	69	1.8689E-11
32	3.68691E-14	68	1.84745E-11
33	3.78146E-14	67	1.82592E-11
34	3.87536E-14	66	1.80432E-11
35	3.96864E-14	65	1.78262E-11
36	4.0613E-14	64	1.76085E-11
37	4.15336E-14	63	1.73898E-11
38	4.24484E-14	62	1.71703E-11
39	4.33575E-14	61	1.69498E-11
40	4.4261E-14	60	1.67285E-11
41	4.51591E-14	59	1.65062E-11
42	4.60518E-14	58	1.62829E-11
43	4.69394E-14	57	1.60586E-11
44	4.78219E-14	56	1.58333E-11
45	4.86994E-14	55	1.5607E-11
46	4.95721E-14	54	1.53796E-11
47	5.04401E-14	53	1.51512E-11
48	5.13034E-14	52	1.49216E-11
49	5.21622E-14	51	1.46909E-11
50	5.30166E-14	50	1.44591E-11
51	5.38667E-14	49	1.42261E-11
52	5.47125E-14	48	1.39918E-11
53	5.55543E-14	47	1.37564E-11
54	5.63919E-14	46	1.35197E-11
55	5.72256E-14	45	1.32817E-11
56	5.80555E-14	44	1.30423E-11
57	5.88815E-14	43	1.28017E-11
58	5.97039E-14	42	1.25596E-11
59	6.05226E-14	41	1.23161E-11
60	6.13377E-14	40	1.20712E-11
61	6.21494E-14	39	1.18248E-11
62	6.29577E-14	38	1.15768E-11
63	6.37627E-14	37	1.13274E-11
64	6.45644E-14	36	1.10763E-11
65	6.53629E-14	35	1.08236E-11
66	6.61582E-14	34	1.05692E-11
67	6.69505E-14	33	1.03131E-11
68	6.77398E-14	32	1.00552E-11
69	6.85262E-14	31	9.79553E-12
70	6.93097E-14	30	9.534E-12
71	7.00903E-14	29	9.27057E-12
72	7.08682E-14	28	9.00518E-12
73	7.16433E-14	27	8.73777E-12
74	7.24158E-14	26	8.46829E-12
75	7.31857E-14	25	8.19667E-12
76	7.3953E-14	24	7.92284E-12
77	7.47178E-14	23	7.64673E-12
78	7.54802E-14	22	7.36824E-12
79	7.62401E-14	21	7.08729E-12
80	7.69977E-14	20	6.80377E-12
81	7.77529E-14	19	6.51756E-12
82	7.85058E-14	18	6.22852E-12
83	7.92565E-14	17	5.9365E-12
84	8.0005E-14	16	5.6413E-12
85	8.07514E-14	15	5.34272E-12
86	8.14956E-14	14	5.04047E-12
87	8.22377E-14	13	4.73426E-12

Year (0 to 100)	1 kg CO_2 Release in Year Zero. Cumulative Watts per m^2 Over Time – from Dynamic LCA Calculator Ver. 2.0	Years in Atm. = 100 Minus Year	Cumulative Watts per m^2 Over Remaining Years for 1 Metric Ton C in CO_2 Released in This Year
88	8.29778E-14	12	4.4237E-12
89	8.37158E-14	11	4.10832E-12
90	8.44519E-14	10	3.78755E-12
91	8.51861E-14	9	3.46069E-12
92	8.59183E-14	8	3.12687E-12
93	8.66486E-14	7	2.78502E-12
94	8.73771E-14	6	2.43379E-12
95	8.81038E-14	5	2.07152E-12
96	8.88287E-14	4	1.69612E-12
97	8.95518E-14	3	1.30499E-12
98	9.02732E-14	2	8.94834E-13
99	9.09929E-14	1	4.61559E-13
100	9.17109E-14	0	0

The results of these calculations are shown in Tables A2–A5. The total effective long-term storage is somewhat higher than the actual amounts of carbon remaining stored for 100 years because the forcing associated with the gradual release of carbon from stocks of stored carbon is less than would be the case if the same amount of carbon was released all at once in the year the product was manufactured.

Table A2. Total Effective Long-Term Carbon Storage, Estimated as Equivalent Net Radiative Forcing over 100 Years

Year of Manufacture	Million Metric Tons C in Net Radiative Forcing Equivalents		
	Paper	Wood	Total
1990	13.26	31.89	45.15
2005	10.11	32.48	42.59
2020	5.90	28.96	34.86

Table A3. Total Carbon Remaining Stored in Products 100 Years Following Manufacturing

Year of Manufacture	Million Metric Tons C		
	Paper	Wood	Total
1990	11.12	28.63	39.75
2005	8.03	29.05	37.08
2020	4.33	25.92	30.25

Table A4. Carbon Remaining Stored in Products in Use 100 Years Following Manufacturing

Year of Manufacture	Million Metric Tons C		
	Paper	Wood	Total
1990	0.00	6.67	6.67
2005	0.00	7.66	7.66
2020	0.00	6.35	6.35

Table A5. Carbon Remaining Stored in Products in Landfills 100 Years Following Manufacturing

Year of Manufacture	Million Metric Tons C		
	Paper	Wood	Total
1990	11.12	21.97	33.09
2005	8.03	21.39	29.42
2020	4.33	19.57	23.90

The sizes of the pools of stored carbon over time, for products manufactured in 2020, are shown in Figure A2. The analysis indicates that the relative importance of the two pools varies over time.

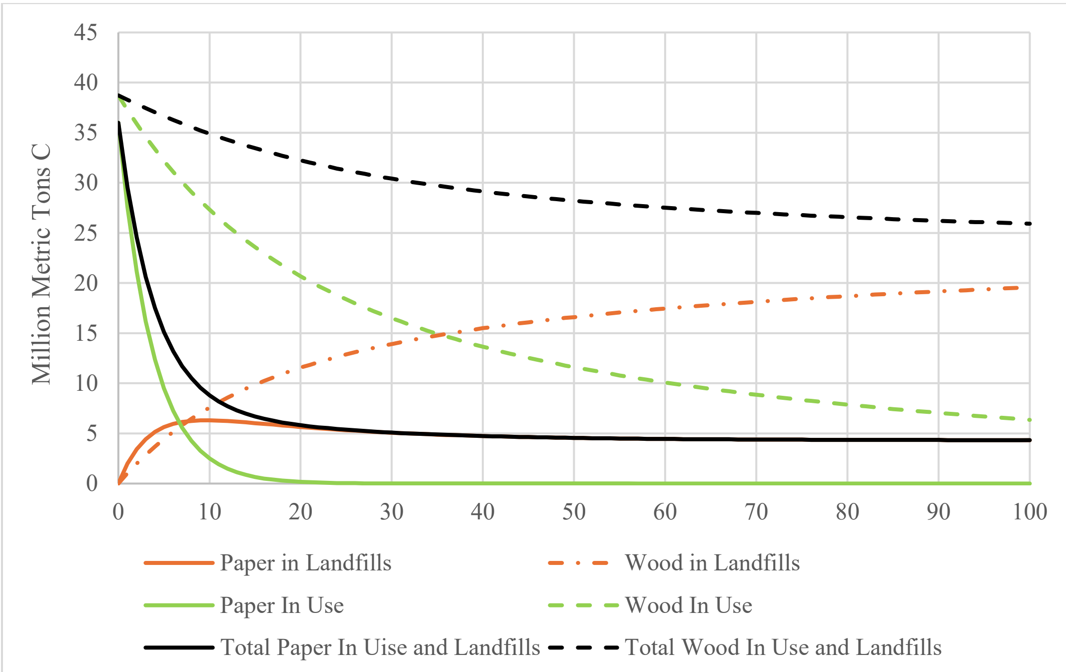


Figure A2. Distribution of Stored Carbon Attributable to Products Made in 2020

Landfill Methane Emissions

Calculating methane emissions over 100 years requires the additional step of converting landfill carbon losses into methane losses[3]. Each year's landfill carbon losses were multiplied by 0.5 to estimate the carbon in as-generated methane for that year. Carbon was converted to methane by multiplying by 16/12. Because landfills are equipped with systems to capture methane for combustion, the capture efficiency of these systems needs to be considered. US EPA (2022b) publishes data showing the amounts of MSW landfill methane generated and captured over past years. From 1990 to 2020 capture efficiency was calculated (equal to the ratio of amounts of MSW methane captured to amounts of MSW methane generated). The data required extrapolation to allow modeling over 100 years. To do this, the average annual rate of improvement from 2005 to 2020 was continued until the capture efficiency reached what was judged to be a practical maximum value.

Data collected by US EPA indicate that individual landfill collection efficiencies range from 60 to 95% (US EPA 2022b). For this study, we assumed a maximum national average capture efficiency of 80%. The trend in capture efficiency used in this study is shown in Figure A3. The capture efficiency used in the calculations was based on the year the emission took place, regardless of the year in which the product was manufactured or discarded. The methane not captured was subject to 10% oxidation as it filtered

[3] As noted previously, the decay of landfilled wood products has been modeled using assumptions for MSW landfills. This is necessary due to the lack of decomposition data for C&D landfills. This approach overestimates methane generation from wood products because it is known that the conditions in C&D landfills are unfavorable to methane production.

through the landfill cap, with the remaining 90% being the emissions to the atmosphere, a common default (e.g., see US EPA 2023a). The total methane releases over 100 years were calculated by simply summing the annual emissions.

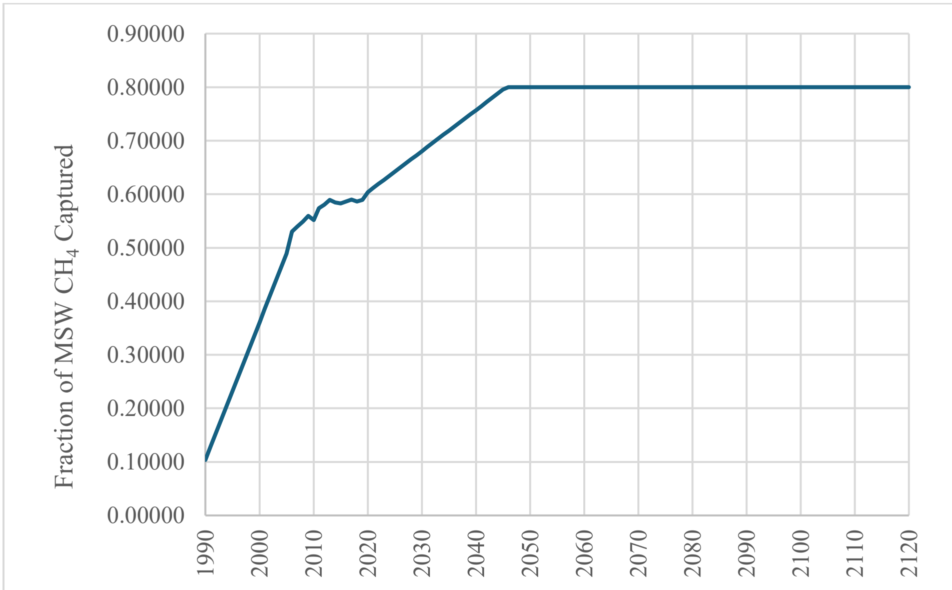


Figure A3. MSW Landfill Gas Capture Efficiencies Based on US EPA (2022b) up to 2020, Extrapolated to 2120

Converting these emissions to CO_2eq. was done using two approaches. First, the total emissions were multiplied by the 100-year GWP of methane, equal to 25 in this study. However, this approach can overstate the radiative forcing caused by these emissions over 100 years because the emissions occur gradually instead of in the year of production. To calculate the radiative forcing, in CO_2eq., associated with the gradual release of the methane, the annual methane released (in kilograms) was entered into version 2.0 of the dynamic LCA calculator (described in Levasseur et al. 2010 and available at https://ciraig.org/index.php/project/dynco2-dynamic-carbon-footprinter/).

Because a GWP for methane of 25 is used throughout this update, it was necessary to modify the output of the dynamic LCA calculator to yield a cumulative 100-year forcing from 1 kg of methane that was equivalent to 25 times the cumulative forcing from 1 kg of CO_2. This required multiplying the calculator results for total cumulative forcing and equivalent CO_2eq. by 0.808.

The methane emissions generated from landfilled paper and wood discards associated with products manufactured in 2020 are shown in Figure A4. Paper-related emissions occur earlier than wood-related emissions because of shorter time in use and faster degradation in the landfill.

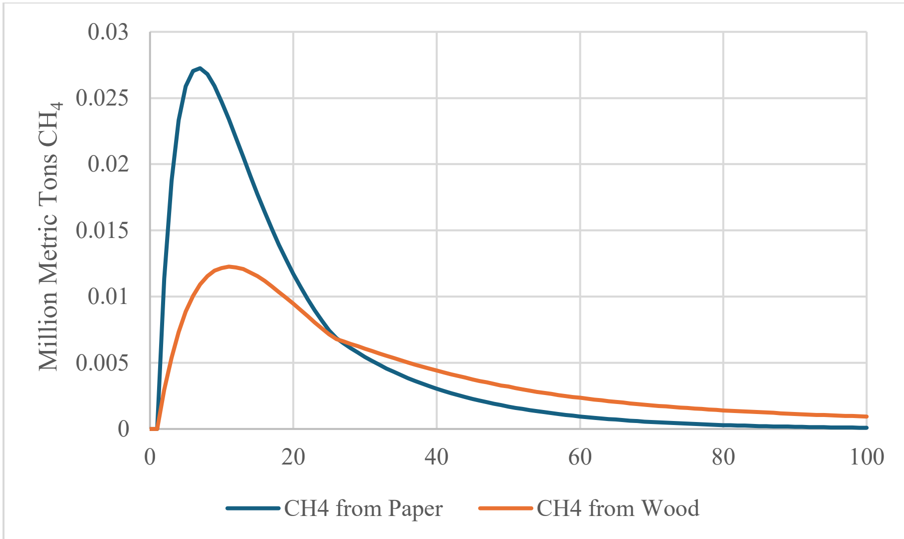


Figure A4. Methane Generation from Products Manufactured in 2020

The methane emissions for products manufactured in 1990, 2005, and 2020 are shown in Table A6. The CO_2eq. results are shown in two ways. The middle section of the table shows the results of the dynamic radiative forcing calculations, while the right-hand section shows the result of simply multiplying the total 100-year emissions of methane by the GWP for methane of 25. While, as expected, the simple approach yields higher results, the differences are relatively small.

Table A6. Methane Emissions and Cumulative Radiative Forcing Associated with Products Manufactured in 1990, 2005, and 2020

Year of Manufacture	Million Metric Tons Methane			Cumulative Radiative Forcing, Million Metric tons CO_2eq.			Million Metric Tons CO_2eq. Obtained by Simply Multiplying the 100-Year Cumulative Methane Emissions by a GWP of 25		
	Paper	Wood	Total	Paper	Wood	Total	Paper	Wood	Total
1990	2.67	0.85	3.52	65.6	20.5	86.1	66.7	21.3	88.0
2005	1.33	0.60	1.92	32.6	14.1	46.7	33.2	14.9	48.1
2020	0.55	0.44	0.99	13.4	10.3	23.8	13.7	11.0	24.7

APPENDIX B

GREENHOUSE GAS EMISSION FACTORS

Fuel-specific emission factors[4] and GWPs[5] published by the Intergovernmental Panel on Climate Change were used to convert subsector fuel use data into Scope 1 GHG emissions. However, in calculating CO_2eq.[6] emissions for CH_4 and N_2O from biomass combustion, factors from the World Resources Institute/World Business Council for Sustainable Development[7] were used because of their better representation of CH_4 and N_2O emissions from US pulp and paper facilities. Note that only GHG contributions of CH_4 and N_2O are considered for biomass fuels. GWPs of 25 and 298 are used for CH_4 and N_2O, respectively, as these values are required under US EPA's GHGRP (US EPA 2023a). Biogenic CO_2 emissions are not included in calculations of GHG emission totals as they are accounted for in the biomass carbon stock change calculations (see US EPA 2022a for more information on biogenic carbon accounting). Nonetheless, these emissions are included for informational purposes, as required in the GHG Protocol Corporate Standard (GHG Protocol 2004).

GHG emission factors for purchased electricity are taken from eGRID and are based on national averages for a given reporting year[8]. Purchased electricity emission factors from eGRID date back to 1996. eGRID factors from 1996 have been applied to calculate GHG emissions from purchased electricity prior to 1996. Purchased steam emission factors over time are based on the approach used in US DOE Instructions for Form EIA-1605 (2007) but using time series EIA MECS data. In form EIA-1605, the purchased steam emission factor is based on the weighted average from EIA's 1998 MECS data on quantities of natural gas, coal, and residual and distillate fuel oils consumed as boiler fuel, carbon coefficients provided in EIA's Assumptions to the Annual Energy Outlook 2003, and EIA/OIAF efficiency assumptions of 80%, 81%, and 82% for natural gas, coal, and petroleum boilers, respectively. The value also includes 10% loss during transmission. Emission intensities were calculated by dividing GHG emissions by 2018 sector-wide production amounts[9].

4 Eggleston, Simon, Leandro Buendia, Kyoko Miwa, Todd Ngara, and Kiyoto Tanabe, eds. 2006. *2006 IPCC Guidelines for National Greenhouse Gas Inventories*. Institute for Global Environmental Strategies.

5 IPCC (Intergovernmental Panel on Climate Change). 2013. "Anthropogenic and Natural Radiative Forcing." In: *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Stocker, T.F., Qin., D., Plattner, G.-K., Tignor, M., Allen, S.K., Borshung, J., Nauels, A., Xia, Y., Bex, V., and Midgley, P.M., eds. Cambridge, UK and New York, USA: Cambridge University Press.

6 CO_2 equivalent normalizes the GHG effect of various GHGs based on the radiative forcing of a unit of carbon dioxide over a specified timeframe (generally 100 years).

7 WRI/WBCSD (World Resources Institute and World Business Council for Sustainable Development). 2009. Calculation Tools for Estimating Greenhouse Gas Emission from Pulp and Paper Mills. Report version 1.3. Published under the WRI/WBCSD Greenhouse Gas Protocol Initiative. Cary, NC: National Council for Air and Stream Improvement, Inc. www.ghgprotocol.org.

8 US EPA 2023c

9 AF&PA 2020

APPENDIX C

CLASSIFICATION OF THE INDUSTRY

The forest products industry is categorized by the SIC system or NAICS by the US government. NAICS is a common classification system shared by the US, Canada, and Mexico and enables direct comparison of economic data across North America (available at https://www.bls.gov/ces/naics/). NAICS classifications for various sectors are provided in Table C1, and SIC classifications of the pulp, paper, and paperboard sectors are provided in Table C2. The major change between SIC and NAICS for the US pulp and paper sector is that the SIC 2621 – paper mills category was divided into NAICS 322121 – paper (except newsprint) mills and 322122 – newsprint mills categories. The US EIA collects information on manufacturing energy use every 4 years as part of its MECS effort. The latest MECS data set that is available is for calendar year 2018. Since 1998, EIA MECS has used NAICS for the US pulp and paper industry. Prior to 1998, EIA MECS classifications for the US pulp and paper industry were based on SIC classification. The detailed description of the sectors and subsectors is available online at: https://www.census.gov/naics/reference_files_tools/2022_NAICS_Manual.pdf.

Table C1. NAICS Classification System for Forest Products and Related Industries

NAICS Code	Wood Product Industry Sector		NAICS Code	Paper Industry Sector
321	Wood Product Manufacturing		322	Paper Manufacturing
3211	Sawmills and Wood Preservation		3221	Pulp, Paper, and Paperboard Mills
32111	Sawmills and Wood Preservation		32211	Pulp Mills
321113	Sawmills		322110	Pulp Mills
321114	Wood Preservation		32212	Paper Mills
3212	Veneer, Plywood, and Engineered Wood Product Manufacturing		322120	Paper Mills
32121	Veneer, Plywood, and Engineered Wood Product Manufacturing		32213	Paperboard Mills
321211	Hardwood Veneer and Plywood Manufacturing		322130	Paperboard Mills
321212	Softwood Veneer and Plywood Manufacturing		3222	Converted Paper Product Manufacturing
321215	Engineered Wood Member Manufacturing		32221	Paperboard Container Manufacturing
321219	Reconstituted Wood Product Manufacturing		322211	Corrugated and Solid Fiber Box Manufacturing
3219	Other Wood Product Manufacturing		322212	Folding Paperboard Box Manufacturing
32191	Millwork		322219	Other Paperboard Container Manufacturing
321911	Wood Window and Door Manufacturing		32222	Paper Bag and Coated and Treated Paper Manufacturing
321912	Cut Stock, Resawing Lumber, and Planing		322220	Paper Bag and Coated and Treated Paper Manufacturing
321918	Other Millwork (including Flooring)		32223	Stationery Product Manufacturing
32192	Wood Container and Pallet Manufacturing		322230	Stationery Product Manufacturing
321920	Wood Container and Pallet Manufacturing		32229	Other Converted Paper Product Manufacturing
32199	All Other Wood Product Manufacturing		322291	Sanitary Paper Product Manufacturing
321991	Manufactured Home (Mobile Home) Manufacturing		322299	All Other Converted Paper Product Manufacturing
321992	Prefabricated Wood Building Manufacturing		323	Printing and Related Support Activities
321999	All Other Miscellaneous Wood Product Manufacturing		3231	Printing and Related Support Activities
			32311	Printing
			323111	Commercial Printing (except Screen and Books)
			323113	Commercial Screen Printing
			323117	Books Printing
			32312	Support Activities for Printing
			323120	Support Activities for Printing

Table C2. Standard Industrial Classification System

SIC Classification
26 – Paper and Allied Products
2611 – Pulp Mills
2621 – Paper Mills
2631 – Paperboard Mills
24 – Lumber and Wood Products, Except Furniture
2420 and 24211 – Sawmills and Planing Mills, General
2426 – Hardwood Dimension and Flooring Mills
2429 – Special Product Sawmills, Not Elsewhere Classified
2430 – Millwork, Veneer, Plywood, and Structural Wood
2435 and 2436 – Hardwood Veneer and Plywood
2439 – Structural Wood Members, Not Elsewhere Classified

APPENDIX D

USING DATA FROM THE COMMODITY FLOW SURVEY

Every 5 years (ending in 2 or 7) a congressionally mandated survey, called the Commodity Flow Survey (CFS), is distributed under the auspices of the US Department of Transportation and the US Department of Commerce. The survey collects data on commodity shipments from a sample of establishments in specific industries (identified by NAICS code). The results of the survey can be used to develop sample-based estimates of, for instance, total ton-miles of wheat shipped in the US. The commodities included in the survey are identified by SCTG codes. The two-digit SCTG codes are shown in Table D1, with the lower level SCTG codes of most significance to the forest products industry also shown.

The CFS is distributed to a sample of facilities representing industries considered “in scope.” The in-scope industries for the 2017 survey, identified by NAICS code, are shown in Table D2.

The CFS 2017 shipment data for SCTG codes of most relevance to the forest products industry (SCTG codes 26, 27, 28 and 29) are shown in Table D3.

An examination of the SCTG codes and the in-scope NAICS codes reveals several factors that need to be considered when using CFS data for studies of forest products.

Roundwood Shipments

Since 2002, the in-scope NAICS code lists for the CFS have not included NAICS code 113, forestry and logging. Therefore, the data on shipments of logs and other wood in the rough (SCTG 25) do not include wood shipments from the forest to mills by companies classified as forestry and logging companies. This is why quantities of SCTG 25 shipments reported in the CFS (approximately 23.3 million metric tons) are much less than total roundwood production in the US of over 250 million metric tons (calculated from 15.9 billion cubic feet of roundwood harvested reported for 2017 by Howard and Liang (2019)).

Wood Product Shipments

The CFS provides estimates of shipments of a given commodity from all in-scope industries. In the case of wood products, this means that multiple shipments will often be reported for the same piece of wood. For instance, if a lumber mill (NAICS in-scope code 321) ships lumber (SCTG code 26212) to a door manufacturer, the shipment of lumber is reported. When the door manufacturer (also NAICS code 321) ships the door (SCTG code 26401) to a garden shed company, that shipment is reported. When the garden shed company (also NAICS in-scope code 321) ships the shed (SCTG code 40920) to a retailer, that shipment is recorded. In this process, shipments involving a portion of the original wood have been reported four times.

It should also be noted that the shipments of some wood products in SCTG 26 include non-wood materials, for instance the glass panes in windows (SCTG 26401), that are not part of the forest industry value chain. On the other hand, SCTG 26 does not include some wood products (e.g., wood furniture). Imports and exports can also influence the interpretation of CFS data. While the shipments associated with moving products into the US are not included, subsequent shipments within the US (e.g., lumber produced in the US from imported logs) are included. While transport of exports to other countries is included, transport of US products within foreign countries is not included.

The CFS reports total shipments of wood products (SCTG 26) originating from all in-scope NAICS codes of 293 million metric tons in 2017. Looking only at wood products shipments from NAICS code 321 (wood products manufacturing, including secondary products like flooring, windows, doors, premanufactured wood structures), CFS reports that shipments equaled about 187 million metric tons. This constituted 64% of all shipments of wood products. An additional 28% of the total wood product shipments were from durable goods merchant wholesalers (NAICS 423). These values can be compared to 2017 US production of primary wood products (i.e., lumber, panels, engineered wood products) by NAICS 321 in the range of 65 to 80 million metric tons (66 million metric tons calculated from Table 8b in Howard and Liang (2019), adding lumber, plywood, veneer, panels, and other industrial products and 78 million metric tons estimated from FAOSTAT data). Total shipments from wood products mills (NAICS 321) reported in the CFS are, therefore, 2.3 to 2.9 times total US production. Most of this can be attributed to the sequential shipments.

The inclusion of sequential shipments of the same piece of wood helps capture the movement of wood along the value chain and may, therefore, provide a reasonable basis for estimating transport-related value chain emissions. On the other hand, as noted earlier, total shipments include the weight of non-wood components in some products while excluding other wood-based products. The relative importance of these two factors, and others that might affect wood products data in the CFS, is difficult to assess.

Paper Product Shipments

AF&PA reports that in 2017 the US industry produced 71.9 million metric tons of paper and paperboard (including newsprint) and 8.3 million metric tons of market pulp (AF&PA 2021a). US mills, therefore, shipped 80.2 million metric tons of primary product in 2017. By comparison, the CFS indicates that, in 2017, shipments of pulp, newsprint, paper, and paperboard (SCTG 27) equaled 126 million metric tons. Of this, 70% or 88 million metric tons were from NAICS 322, which includes pulp, paper, and paperboard mills as well as a variety of converting operations. An additional 26% of the total shipments of SCTG 27 was from merchant wholesalers of durable and nondurable goods. The shipments of pulp, newsprint, paper, and paperboard (SCTG 27) from NAICS 322 (88 million metric tons) are much closer to the annual production (80.2 million metric tons) than is the case for wood products, discussed earlier.

As noted previously for wood products, one must be aware of the potential for the data to be impacted by (1) non-paper components of some items in SCTG 27 and (2) imports and exports.

These shipments of paper products (SCTG 27) are separate from the 74 million metric tons of shipments of “paper or paperboard articles” (SCTG 28), 67% of which were from paper and paperboard mills (i.e., NAICS 322) and 25% from merchant wholesalers. In addition, shipments of SCTG 28, printed products (which includes products printed on non-paper substrates) amounted to an additional 25 million metric tons in 2017.

Table D1. Two-Digit SCTG Commodity Code List with Expanded Code Lists for Items of Interest to the Forest Products Industry
[*Source*: https://bhs.econ.census.gov/bhsphpext/brdsearch/scs_code.html]

SCTG 2-Digit Code	Commodity Description
01	Live Animals and Fish
02	Cereal Grains (including seed)
03	Agricultural Products Except for Animal Feed, Cereal Grains, and Forage Products
04	Animal Feed, Eggs, Honey, and Other Products of Animal Origin
05	Meat, Poultry, Fish, Seafood, and Their Preparations
06	Milled Grain Products and Preparations, and Bakery Products
07	Other Prepared Foodstuffs, Fats and Oils
08	Alcoholic Beverages and Denatured Alcohol
09	Tobacco Products
10	Monumental or Building Stone
11	Natural Sands
12	Gravel and Crushed Stone Except Dolomite and Slate
13	Other Nonmetallic Minerals Not Elsewhere Classified
14	Metallic Ores and Concentrates
15	Coal
16	Crude Petroleum
17	Gasoline, Aviation Turbine Fuel, and Ethanol, Including Kerosene, and Fuel Alcohols
18	Fuel Oils Including Diesel, Bunker C, and Biodiesel
19	Other Coal and Petroleum Products, Not Elsewhere Classified
20	Basic Chemicals
21	Pharmaceutical Products
22	Fertilizers
23	Other Chemical Products and Preparations
24	Plastics and Rubber
25	Logs and Other Wood in the Rough (<u>Note that the CFS does not include shipments from the forest to mills by companies classified as forestry and logging companies</u>) 25010 Logs for pulping (pulpwood) 25020 Logs for lumber 25091 Fuel wood 25092 Wood in the rough, treated with paint, stains, creosote, or other preservatives

26	Wood Products 26100 Wood chips or particles 26211 Lumber, treated 26212 Lumber, untreated 26221 Wood continuously shaped along any of its edges or faces 26222 Shingles and shakes 26310 Veneer sheets and sheets for plywood 26320 Particle board, fiberboard, and similar board of wood or other ligneous materials 26330 Plywood, veneered panels, and similar laminated wood including door skins 26401 Windows, doors, and frames and thresholds 26409 Other builders' joinery and carpentry of wood, not elsewhere classified (except shingles and shakes, see 26222) 26901 Wood packing containers, cable drums, pallets and skids, and coopers' products such as cask and barrels 26909 Other Wood products, not elsewhere classified, including wood charcoal, densified wood, and coffins
27	Pulp, Newsprint, Paper, and Paperboard 27110 Mechanical wood pulp 27120 Non-dissolving grades of soda or sulfate chemical wood pulp 27191 Dissolving grades of chemical wood pulp 27199 Other Pulp of fibrous cellulosic materials, not elsewhere classified (including recycled pulp) 27200 Newsprint in large rolls or sheets 27311 Uncoated paper for writing, printing, or other graphic purposes, in large rolls or sheets 27312 Toilet or facial tissue stock, towel or napkin stock, and similar paper stock used for household or sanitary purposes, in large rolls or sheets 27319 Other uncoated paper in large rolls or sheets, not elsewhere classified 27320 Uncoated paperboard in large rolls or sheets 27410 Paper, coated, impregnated, treated, or worked, in large rolls or sheets 27420 Paperboard, coated, impregnated, treated, or worked, in large rolls or sheets
28	Paper or Paperboard Articles Pulp 28010 Toilet paper, facial tissues, towels, tampons, sanitary napkins, disposable diapers, and similar articles of paper for household, sanitary, or hospital use, and paper articles of apparel 28021 Sacks and bags of paper, paperboard, cellulose wadding, or cellulose fiber webs 28029 Other packing containers of paper, paperboard, cellulose wadding, or webs of cellulose fibers, not elsewhere classified 28091 Wallpaper and similar wall coverings 28092 Envelopes, letter cards, plain postcards and correspondence cards, and

29	Printed Products 29100 Printed books, brochures, leaflets, and similar printed products (except advertising materials including catalogs, see 29300; atlases and music books, see 29999) 29210 Newspapers 29220 Journals and periodicals 29300 Advertising material, commercial or trade catalogs, and similar printed products, including flyers 29910 Printed or illustrated postcards, messages, or announcements, and printed cards bearing personal greetings 29991 Manifold business-forms and interleaved carbon-sets 29999 Other printed products, not elsewhere classified, including blank books, binders, and albums
30	Textiles, Leather, and Articles of Textiles or Leather
31	Nonmetallic Mineral Products
32	Base Metal in Primary or Semifinished Forms and in Finished Basic Shapes
33	Articles of Base Metal
34	Machinery
35	Electronic and Other Electrical Equipment and Components, and Office Equipment
36	Motorized and Other Vehicles (including parts)
37	Transportation Equipment, N.E.C.
38	Precision Instruments and Apparatus
39	Furniture, Mattresses and Mattress Supports, Lamps, Lighting Fittings, and Illuminated Signs 39019 Other household or office furniture, not elsewhere classified, including kitchen cabinets (except desk top furniture, which is classified by its material; and TV and stereo cabinets, see 35820) 39021 Medical, surgical, dental, or veterinary furniture 39029 Other Furniture, not elsewhere classified
40	Miscellaneous Manufactured Products 40920 Prefabricated buildings including tool or garden sheds 40999 Other miscellaneous manufactured products, not elsewhere classified
41	Waste and Scrap 41210 Sawdust and wood waste and scrap 41220 Waste and scrap of paper or paperboard 41299 Other nonmetallic waste and scrap, not elsewhere classified
43	Mixed Freight

Table D2. In-Scope Industries for the 2017 CFS
[*Source*: Table 2 in 2017 Commodity Flow Survey Report[10]]

NAICS	Industry
212	Mining (except oil and gas)
311	Food manufacturing
312	Beverage and tobacco product manufacturing
313	Textile mills
314	Textile product mills
315	Apparel manufacturing
316	Leather and allied product manufacturing
321	Wood product manufacturing
322	Paper manufacturing
323	Printing and related support activities
324	Petroleum and coal products manufacturing
325	Chemical manufacturing
326	Plastics and rubber products manufacturing
327	Nonmetallic mineral product manufacturing
331	Primary metal manufacturing
332	Fabricated metal product manufacturing
333	Machinery manufacturing
334	Computer and electronic product manufacturing
335	Electrical equipment, appliance, and component manufacturing
336	Transportation equipment manufacturing
337	Furniture and related product manufacturing
339	Miscellaneous manufacturing
4231	Motor vehicle and parts merchant wholesalers
4232	Furniture and home furnishing merchant wholesalers
4233	Lumber and other construction materials merchant wholesalers
4234	Commercial equipment merchant wholesalers
4235	Metal and mineral (except petroleum) merchant wholesalers
4236	Electrical and electronic goods merchant wholesalers
4237	Hardware and plumbing merchant wholesalers
4238	Machinery, equipment, and supplies merchant wholesalers
4239	Miscellaneous durable goods merchant wholesalers
4241	Paper and paper product merchant wholesalers
4242	Drugs and druggists' sundries merchant wholesalers
4243	Apparel, piece goods, and notions merchant wholesalers
4244	Grocery and related product merchant wholesalers

[10] US Department of Transportation, Bureau of Transportation Statistics, and US Department of Commerce. 2020. US Census Bureau, 2017 Commodity Flow Survey. EC17TCF-US, 2017 Economic Census: Transportation, Washington, DC. http://www.census.gov/content/dam/Census/library/publications/2017/econ /ec17tcf-us.pdf.

4245	Farm product raw material merchant wholesalers
4246	Chemical and allied products merchant wholesalers
4247	Petroleum and petroleum products merchant wholesalers
4248	Beer, wine, and distilled alcoholic beverage merchant wholesalers
4249	Miscellaneous nondurable goods merchant wholesalers
4541	Electronic shopping and mail-order houses
45431	Fuel dealers
4841	General freight trucking
4842	Specialized freight trucking
4931	Warehousing and storage
5111	Newspaper, periodical, book, and directory publishers
51223	Music publishers
551114	Corporate, subsidiary, and regional managing offices

Table D3. Shipping Data from the 2017 Commodity Flow Survey for SCTG Categories 26, 27, 28 and 29
[*Source*: Table CFSArea2017.CF1700A19 downloaded from CFS website: https://data.census.gov/table/CFSAREA2017.CF1700A19]

Commodity >>>>	Pulp, Newsprint, Paper, and Paperboard SCTG 27			Paper or Paperboard Articles SCTG 28			Printed Products SCTG 29			Wood Products SCTG 26		
NAICS code	Metric tons (1000s)	Metric ton-km (millions)	Average km per shipment	Metric tons (1000s)	Metric ton-km (millions)	Average km per shipment	Metric tons (1000s)	Metric ton-km (millions)	Average km per shipment	Metric tons (1000s)	Metric ton-km (millions)	Average km per shipment
321 Wood product manufacturing	113	155	650	52	77[b]	196	52[c]	77[b]	1,873	186,574	98,496	605
322 Paper manufacturing	88,253	88,175	814	49,179	25,637	695	943	802	1,220	744	404	267
323 Printing and related support	1,019	774	951	352	407	996	15,878	10,976	1,223	0[d]	0[d]	1,067
423 Merchant wholesalers, durable goods	16,515	5,082[a]	378	7,635	2,584	781	1,583	397	1,099	83,288	23,133	238
424 Merchant wholesalers, nondurable goods	16,856	5,082	206	12,888	6,846	1,048	2,177	1,657	1,041	0[d]	0[d]	349
Other	4,155	22,457		3,397	2,852		4,767			22,821	7,547	
Total	126,912	121,725	429	73,504	38,405	928	25,400	16,233	772	293,426	129,580	563

[a] Assumed to be the same as NAICS code 424.
[b] Assumed one-half of value for SCTG 27.
[c] Assumed to be the same as paperboard articles SCTG 28.
[d] Assumed to be zero.

APPENDIX E

RELATIONSHIP BETWEEN DOC, L_0, AND DOCF

The term “degradable organic carbon” (DOC) can be thought of as the fraction of carbon that can degrade under ideal conditions. For practical purposes, it can be considered the biogenic carbon content of a waste. DOC is commonly reported in units of weight fraction on a wet basis (e.g., tons biogenic carbon per wet ton).

Lo is the ultimate methane generation potential of a waste, reported in units of cubic meters of methane per unit of wet waste (usually cubic meters of methane per wet metric ton).

The ratio Lo/DOC is, therefore, in units of cubic meters of methane per unit of degradable (i.e., biogenic) carbon. It is not a function of water content. With the appropriate conversion factors, it can be converted into DOCf as follows:

> Assume methane density of 0.67 kg per cubic meter
>
> Lo (m^3 ultimate methane generation/wet metric ton waste) / DOC (metric ton organic C/metric ton wet waste) = Lo/DOC (m^3 ultimate methane generation/metric ton organic C)
>
> Lo/DOC (m^3 ultimate methane generation/metric ton organic C)*0.67 (kg methane/m^3 methane) = 0.67*Lo/DOC (kg ultimate methane generation/metric ton organic C)
>
> Or 0.00067*Lo/DOC (kg ultimate methane generation/kg organic C)
>
> Converting this to units of carbon = (0.00067*12/16)*Lo/DOC (kg C in ultimate methane generation/kg organic C)
>
> One-half of the degradable carbon is converted into methane while the other half is converted into carbon dioxide. Therefore:
>
> 2*(0.00067*12/16)*Lo/DOC = kg C ultimately degradable carbon/kg organic C
>
> Which equals DOCf
>
> Therefore: DOCf = 0.001*Lo/DOC

The original GHGRP rule contained a single DOC value for pulp and paper mill wastes. In its Technical Support Document for the original GHGRP Subpart TT rules (pg. 18), US EPA addressed the DOC-moisture issue by observing that:

> “Pulp and paper waste, the IPCC default DOC value [0.4] is based on 10 percent moisture content. However, Kraft and Orender (1993) present data for some pulp and paper waste streams suggesting the moisture content is commonly 50 percent… Correcting the IPCC default DOC value for pulp and paper waste to be on a wet basis of 50 percent moisture, the IPCC value would be 0.22. Considering all of these data, a default DOC value of 0.20 [subsequently changed to 0.15] is recommended for the pulp and paper industry.” (US EPA 2010)

However, in response to comments, US EPA reviewed the data submitted in early years of the GHGRP and, in December 2016, created new categories of pulp and paper mill wastes with different DOC values. The data are reviewed in a 2015 memorandum from RTI International to US EPA (RTI

2015). The DOC data examined in that memorandum are those submitted by companies and are all on a wet basis. The default DOC values subsequently added to the GHGRP are based directly on these data and are therefore most appropriately applied at the solids content expected of the wastes tested by the reporting companies. Therefore, in this report, when using GHGRP default DOC values, we have applied them to waste materials at solids contents that, based on best judgment and data, are representative of what is disposed.

APPENDIX F

CALCULATIONS TO DETERMINE MSW METHANE EMISSIONS ATTRIBUTABLE TO FOREST PRODUCTS

US EPA has published data on inputs to landfills going back to 1960. These data, calculated as a percentage of all materials placed in landfills, are shown in Table F1.

Table F1. Materials Going to Landfills, Percent of Total Landfilled Material. [*Source*: US EPA (2020c)][a]

Percent of Landfilled Materials										
	1960	1970	1980	1990	2000	2005	2010	2015	2017	2018
Paper andpPaperboard	30.20%	33.20%	31.70%	30.00%	28.80%	24.70%	16.10%	13.30%	13.10%	11.80%
Glass	8.00%	11.10%	10.50%	6.00%	5.80%	5.80%	5.10%	5.00%	5.40%	5.20%
Metals										
Ferrous	12.40%	10.80%	8.90%	6.00%	5.60%	6.00%	6.80%	7.20%	7.40%	7.20%
Aluminum	0.40%	0.70%	1.00%	1.00%	1.40%	1.60%	1.80%	1.80%	1.90%	1.80%
Other nonferrous	0.20%	0.30%	0.40%	0.20%	0.30%	0.30%	0.40%	0.50%	0.50%	0.50%
Total Metals	*13.00%*	*11.80%*	*10.30%*	*7.20%*	*7.30%*	*7.90%*	*9.00%*	*9.50%*	*9.80%*	*9.50%*
Plastics	0.50%	2.60%	5.00%	9.50%	14.20%	16.40%	17.90%	18.90%	19.10%	18.50%
Rubber and leather	1.80%	2.40%	3.00%	3.20%	2.80%	2.90%	3.20%	3.30%	3.50%	3.40%
Textiles	2.10%	1.70%	1.70%	2.90%	4.50%	5.30%	6.50%	7.70%	7.90%	7.70%
Wood	3.70%	3.30%	5.10%	6.90%	7.10%	7.50%	8.20%	8.00%	8.70%	8.30%
Other	0.10%	0.40%	1.50%	1.40%	1.80%	1.80%	2.10%	2.20%	2.20%	2.00%
Total Materials in Products	***59.40%***	***66.50%***	***68.80%***	***67.10%***	***72.30%***	***72.30%***	***68.10%***	***67.90%***	***69.70%***	***66.40%***
Other Wastes										
Food	14.80%	11.30%	9.50%	13.60%	17.30%	18.50%	21.00%	22.00%	21.80%	24.10%
Yard trimmings	24.20%	20.50%	20.10%	17.60%	8.50%	7.00%	8.60%	7.80%	6.20%	7.20%
Miscellaneous inorganic wastes	1.60%	1.60%	1.60%	1.70%	1.90%	2.20%	2.30%	2.30%	2.30%	2.30%
Total Other Wastes	***40.60%***	***33.40%***	***31.20%***	***32.90%***	***27.70%***	***27.70%***	***31.90%***	***32.10%***	***30.30%***	***33.60%***
	100.0%	99.9%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%	100.0%

[a] The US EPA data do not account for materials used to package or ship exported and imported goods. These materials can include paper, paperboard, wood, plastics, and metals. After use, these are discarded in the US but are not accounted for in the methods used for US EPA (2020c) estimates. The effects on the values in Tables E1 and E2 are not known.

From these data, the landfilled amounts of degradable materials were isolated. The degradable fractions include paper and paperboard, wood, food, and yard trimmings. The results are shown in Table F2.

Table F2. Biodegradable Materials Sent to MSW Landfills [*Source*: Based on US EPA (2020c)][a]

As fraction of all MSW going to landfills										
	1960	1970	1980	1990	2000	2005	2010	2015	2017	2018
Paper and Paperboard	30.20%	33.20%	31.70%	30.00%	28.80%	24.70%	16.10%	13.30%	13.10%	11.80%
Wood	3.70%	3.30%	5.10%	6.90%	7.10%	7.50%	8.20%	8.00%	8.70%	8.30%
Food	14.80%	11.30%	9.50%	13.60%	17.30%	18.50%	21.00%	22.00%	21.80%	24.10%
Yard Trimmings	24.20%	20.50%	20.10%	17.60%	8.50%	7.00%	8.60%	7.80%	6.20%	7.20%
Total biodegradable	72.90%	68.30%	66.40%	68.10%	61.70%	57.70%	53.90%	51.10%	49.80%	51.40%
As fraction of total biodegradable (leather and textiles not included to make fraction of forest products "conservative")										
	1960	1970	1980	1990	2000	2005	2010	2015	2017	2018
Paper and Paperboard	41.43%	48.61%	47.74%	44.05%	46.68%	42.81%	29.87%	26.03%	26.31%	22.96%
Wood	5.08%	4.83%	7.68%	10.13%	11.51%	13.00%	15.21%	15.66%	17.47%	16.15%
Food	20.30%	16.54%	14.31%	19.97%	28.04%	32.06%	38.96%	43.05%	43.78%	46.89%
Yard Trimmings	33.20%	30.01%	30.27%	25.84%	13.78%	12.13%	15.96%	15.26%	12.45%	14.01%
	100.00%	100.00%	100.00%	100.00%	100.00%	100.00%	100.00%	100.00%	100.00%	100.00%

[a] The US EPA data do not account for materials used to package or ship exported or imported goods. These materials can include paper, paperboard, wood, plastics, and metals. After use, these are discarded in the US but are not accounted for in the methods used for US EPA (2020c) estimates. The effects on the values in Tables E1 and E2 are not known.

After interpolating to provide annual data, the individual year's contributions were modeled to determine the amount of degradable carbon lost from each type of waste in every year after it was deposited. A first order decay model was used to estimate the decomposition of degradable carbon. Nondegradable carbon was not included in the calculations. The parameter values used were those used by US EPA in version 15 of the WARM, which are shown in Table F3. Also shown are the values used in the previous profile report (NCASI 2008; Heath et al. 2010). There are significant differences between them. The values from US EPA's WARM documentation are used in this profile to estimate methane emissions attributable to forest products because other US EPA sources for parameter values do not allow separate examination of different components of the waste stream.

Table F3. Parameters Used to Estimate the Contribution of Forest Products to MSW Landfill Methane

Waste Material	This Report		Previous Profile (NCASI 2008; Heath et al. 2010)	
	Nondegradable Fraction of Carbon (US EPA 2020b)	Half-Life of Degradable Carbon in Landfills, Years (US EPA 2020b)	Nondegradable Fraction of Carbon	Half-Life of Degradable Carbon in Landfills, Years
Paper and paperboard[a]	0.56	11.9	0.44	14.5
Wood products[b]	0.86	7.3	0.77	29
Food waste	0.16	3.6	Not Applicable	Not Applicable
Yard waste	0.69	2.7	Not Applicable	Not Applicable

[a] Simple average of values for different types of paper and paperboard.
[b] Simple average of lumber and MDF.

The calculation results, shown in Table F4, are estimates of the fraction of the loss of degradable carbon (the same as the fraction of methane generation) attributable to each type of waste over time. These are multiplied by the total methane generation reported by US EPA (US EPA 2022b) in the corresponding year to determine the methane emissions attributable to each type of waste.

Staley and Barlaz (2009), using US EPA's 2006 MSW disposal data, calculated that forest products may have produced approximately 60% of total ultimate landfill methane potential for waste disposed in that year. For that same year, NCASI estimates that forest products were responsible for approximately 40% of MSW landfills emissions (Table F4). It is important to note that Staley and Barlaz did not consider degradation rates, only ultimate methane potential; therefore, the two estimates are not directly comparable.

These estimates can also be compared to those recently developed by US EPA for methane from food waste. US EPA estimates that food waste was responsible for 58% of landfill methane releases in 2020 (US EPA 2023d), which compares to an estimate for 2018 in Table F4 of 60%.

Table F4. Fraction of Methane Attributable to Each Type of Waste in MSW

Year	Paper and paperboard	Wood	Food	Yard Waste
1960	0.412	0.016	0.361	0.211
1961	0.412	0.016	0.361	0.211
1962	0.408	0.016	0.362	0.214
1963	0.406	0.016	0.362	0.217
1964	0.406	0.016	0.361	0.218
1965	0.406	0.016	0.359	0.219
1966	0.408	0.015	0.357	0.219
1967	0.411	0.016	0.355	0.219
1968	0.414	0.016	0.352	0.219
1969	0.418	0.016	0.348	0.218
1970	0.422	0.016	0.345	0.218
1971	0.427	0.016	0.341	0.217
1972	0.432	0.016	0.336	0.216
1973	0.436	0.016	0.332	0.216
1974	0.440	0.016	0.328	0.216
1975	0.443	0.016	0.324	0.216
1976	0.447	0.017	0.321	0.216
1977	0.450	0.017	0.317	0.216
1978	0.453	0.018	0.313	0.217
1979	0.456	0.018	0.309	0.217
1980	0.459	0.019	0.305	0.218
1981	0.461	0.019	0.301	0.218
1982	0.463	0.020	0.300	0.218
1983	0.464	0.021	0.300	0.216
1984	0.463	0.021	0.301	0.214
1985	0.463	0.022	0.304	0.211
1986	0.461	0.022	0.308	0.208
1987	0.459	0.023	0.313	0.205
1988	0.457	0.023	0.318	0.201
1989	0.454	0.024	0.324	0.198
1990	0.451	0.025	0.330	0.194
1991	0.448	0.025	0.337	0.190
1992	0.445	0.026	0.345	0.185
1993	0.442	0.026	0.353	0.179
1994	0.439	0.027	0.362	0.172
1995	0.437	0.027	0.372	0.164
1996	0.434	0.027	0.382	0.157
1997	0.432	0.028	0.392	0.149
1998	0.429	0.028	0.402	0.141
1999	0.427	0.028	0.413	0.132
2000	0.424	0.029	0.423	0.124
2001	0.422	0.029	0.434	0.115
2002	0.419	0.029	0.443	0.108
2003	0.415	0.030	0.453	0.102
2004	0.411	0.030	0.462	0.097
2005	0.406	0.030	0.472	0.092
2006	0.401	0.031	0.481	0.088
2007	0.394	0.031	0.490	0.085
2008	0.385	0.031	0.500	0.084
2009	0.375	0.032	0.509	0.084
2010	0.364	0.032	0.520	0.084
2011	0.353	0.032	0.530	0.085
2012	0.342	0.032	0.540	0.086
2013	0.332	0.033	0.550	0.086
2014	0.322	0.033	0.559	0.086
2015	0.313	0.033	0.569	0.085
2016	0.304	0.034	0.578	0.084
2017	0.297	0.034	0.587	0.082
2018	0.290	0.035	0.596	0.079