

**Paper Title:** Carbon Storage due to Disposal of Biogenic Materials in U.S. Landfills

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**Topic Area:** Terrestrial – Sequestration in Forests

**Abstract:** This paper reviews the methodology used to quantify carbon storage in landfills for the *Inventory of U.S. Greenhouse Gas Emissions and Sinks*. Though the Intergovernmental Panel on Climate Change (IPCC) does not provide specific guidance on quantifying this carbon sink, the IPCC Guidelines recommend that countries be comprehensive in their coverage of greenhouse gas emissions and sinks. The US has developed separate methods for quantifying storage attributable to harvested wood products (HWP), and yard trimmings and food scraps. Net changes in carbon stored in HWP (including solid wood products and paper) in use and in dumps or landfills is estimated by tracking the disposition of HWP produced, imported, and exported beginning in 1910 and continuing up to the present using the WOODCARB model. WOODCARB uses assumptions on the fate of carbon in landfill environments to track annual carbon accumulation and emissions. For yard trimmings and food scraps, storage factors are used to estimate the proportion of annual disposal that will persist in the landfill environment.

Landfill carbon storage due to wood products, yard trimmings, and food scraps was about 165 Tg CO<sub>2</sub> Eq in 2002, comprising 24 percent of estimated carbon sequestration in U.S. forests and other carbon pools. To put this in perspective, if the annual carbon storage in the U.S. landfill pool were considered a separate category in the U.S. national greenhouse gas inventory, it would rank 5<sup>th</sup> out of the 47 categories of sources and sinks investigated, behind only fossil fuel combustion, forest carbon storage, agricultural soil emissions, and landfill methane, and ahead of such sources as natural gas systems, coal mining, and all of the “high-GWP” sources combined.

*Keywords:* Landfills; carbon sequestration; carbon sinks; global climate change; greenhouse gas inventories.

## 1. Introduction

Landfilling of wood products and yard trimmings represents a significant, but often overlooked, process of carbon sequestration, as much of the organic material deposited persists indefinitely. The international guidelines for conducting greenhouse gas (GHG) inventories, developed by the Intergovernmental Panel on Climate Change (IPCC), provide the basic framework and methodologies for developing national inventories. As stated therein, “The IPCC *Guidelines* are designed to estimate and report on national inventories of anthropogenic greenhouse gas emissions and removals. In general terms 'anthropogenic' refers to greenhouse gas emissions and removals that are a direct result of human activities or are the result of natural processes that have been affected by human activities” (IPCC/UNEP/OECD/IEA 1997). The source of carbon in wood products, yard trimmings, and food scraps is the atmosphere (it is stored via photosynthesis). Just as methane emissions from landfills are counted in inventories because they result from an anthropogenic process (waste disposal), removals of carbon from the atmosphere resulting from the same anthropogenic process should be included in national inventories. The *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2002* (referred to hereafter as the U.S. Inventory) estimates that net annual removals of carbon in U.S. landfills totaled 51.4 Teragrams of carbon equivalent (Tg C Eq.) in 1990 and 45.1 Tg C Eq. in 2002, respectively 2.7 percent and 2.4 percent of gross greenhouse gas emissions (gross emissions excludes sinks).

Despite the fact that this is potentially a significant carbon removal process globally, to our knowledge only Finland and the United States include this sink in their national GHG inventories. International inventory guidelines and the *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* produced by the IPCC do not provide specific methodologies for estimating landfill carbon storage, but they do recommend that countries be comprehensive in their

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coverage of emissions and sinks (IPCC 2000). The IPCC *Guidelines* note that “Users may include any human-induced emissions and removals in their inventory as long as they can be clearly documented and quantified. The IPCC *Guidelines* contain 'default' methodologies for the estimation of greenhouse gas emissions and removals. Users are encouraged to go beyond these minimum default methods where possible, and report the results.”

The objective of this paper is to summarize the methodologies used in the U.S. national GHG inventory for characterizing atmospheric removals through landfilling of harvested wood products (HWP, primarily paper and solid wood), yard trimmings, and food scraps.

The HWP approach is part of a large modeling framework developed by the U.S. Department of Agriculture, Forest Service, which tracks flows of carbon from forests starting with growth, and including removals, use, and disposal. In this framework, landfill disposal is one of several interconnected carbon pools – the others are forests, roundwood, and products-in-use (Row and Phelps 1996; Skog and Nicholson 2000). As explained later in this paper, carbon flows to and from each of these pools are tracked, and the annual changes in stocks represent carbon dioxide emissions to, or removals from, the atmosphere. The approach for handling carbon storage in landfills from HWP is designed to be consistent with the other parts of the framework.

Paper and wood products are the primary source of biogenic (i.e., biologically based) carbon in the U.S. waste stream. In 2000, paper comprised 29.2 percent by weight of discards, making it the most common material in the waste stream, and wood added another 7.5 percent (U.S. EPA 2002a). Yard trimmings (grass, leaves, branches) are another important source of biogenic carbon (7.4 percent of total discards in 2000). As yard trimmings are not harvested from U.S. forests, they are not tracked by USDA-FS as HWP, and EPA developed a separate methodology to predict landfill carbon storage for these materials. Food scraps also represent a significant carbon input (in terms of total weight, they comprise 15.6 percent of discards), but are more thoroughly decomposed than yard trimmings. The authors estimate that annual carbon storage attributable to food scraps is on the order of 0.5 – 0.8 Tg C Eq. annually.

Other organic materials are also disposed in landfills, but are not addressed by the current methodology. Biosolids, food processing byproducts, and other biogenic materials may also contribute to landfill carbon storage, but have not yet been addressed due to a lack of information on quantities landfilled and carbon fates.

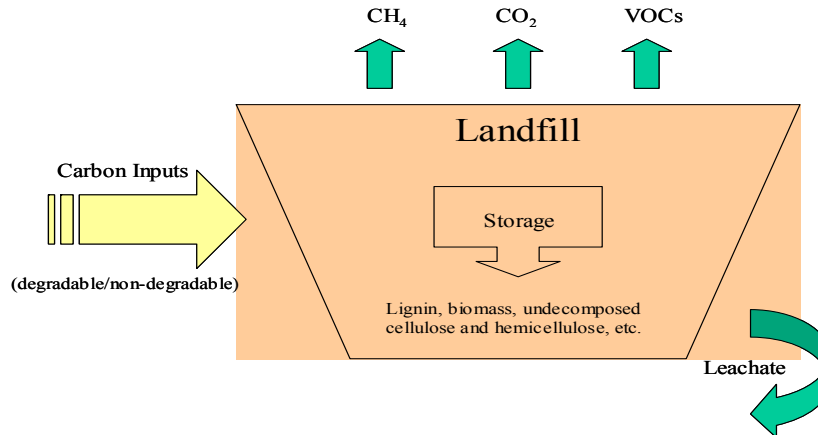
Synthetic organics – plastics, rubber, textiles, and the like – are not included because they are derived from fossil (rather than biogenic) sources of carbon. Biogenic sources store atmospheric CO<sub>2</sub> through photosynthesis, and long-term storage underground represents a process whereby carbon is removed from the biogeochemical cycle (at least on the timescale of interest for GHG inventory purposes). On the other hand, carbon derived from fossil fuels (oil and gas) and used in the synthesis of organic compounds does not represent a net removal from the atmosphere (at least not in this geologic era). Plastics and other polymers do not degrade in the landfill environment – they are essentially inert in an anaerobic setting – and for GHG inventory purposes, it is as if the fossil carbon had never been extracted from the ground in the first place.

This paper begins with a general discussion of the process by which carbon is stored in landfills. Sections 3 and 4 describe the current method for quantifying landfill carbon storage of wood products and yard trimmings, respectively, in the U.S. Inventory. Finally, Section 5 presents key uncertainties in the U.S. Inventory estimates.

## **2. Carbon Stocks and Flows in Landfills**

Exhibit 1 shows the carbon flows within a landfill system. As the exhibit shows, carbon entering the landfill can have one of several fates: exit as methane (CH<sub>4</sub>), exit as carbon dioxide (CO<sub>2</sub>), exit as volatile organic compounds (VOCs), exit dissolved in leachate, or storage in the landfill.

## Exhibit 1. Landfill Carbon Mass Balance



Using a mass balance approach, landfill carbon storage can be calculated as the total carbon entering the landfill minus the carbon exiting as methane, carbon dioxide, volatile organic compounds, or dissolved in leachate. The remainder of this section discusses these flows in more detail, beginning with a discussion of the carbon entering the system.

In 2000, of the 232 million tons of municipal solid waste generated within the United States, 128 million tons was managed through landfilling (U.S. EPA 2002a). The materials entering landfills vary dramatically with respect to composition, with some such as newsprint and food waste containing a relatively large proportion of carbon, and others (e.g., steel, glass) containing no carbon at all (Barlaz 1998). In addition, moisture contents can also vary significantly, ranging from 15-40 percent of waste mass (Eck 2000).

After entering landfills, a portion of the organic materials decomposes, during which it eventually is transformed into landfill gas and/or leachate. Aerobic bacteria initially decompose the waste until the available oxygen is consumed. This stage usually lasts less than a week and is followed by the anaerobic acid state, in which carboxylic acids accumulate, the pH decreases, and some cellulose and hemicellulose decomposition occurs. Finally, during the methanogenic state, bacteria further decompose the organic material into methane and carbon dioxide.

The rate of decomposition in landfills is affected by a number of factors including: (1) waste composition; (2) factors influencing microbial growth (moisture, available nutrients, pH, temperature); and (3) whether the landfill is operated to retard or enhance waste decomposition. Most studies have shown the amount of moisture in the waste, which can vary widely within a single landfill, to be a critical factor in the rate of decomposition (Barlaz, et al. 1990). As a result, there is increasing interest in the operation of a landfill as a bioreactor, in which leachate and possibly other liquids are recirculated to enhance decomposition and gas production (Pacey et al. 1999, Mehta et al. 2002, Reinhart et al. 2002). Bioreactor technologies, which optimize landfill moisture content in order to accelerate waste decomposition, have emerged as a leading technology for facilitating rapid decomposition of organic wastes and cost-effective methane collection.

Of the various components of the landfill carbon system, by far the most research to date has been conducted on the transformation of landfill carbon into methane (Bingemer and Crutzen 1987, Barlaz, et al. 1997). This interest has been spurred by a number of factors, including U.S. EPA's 1996 rule requiring large landfills to control landfill gas emissions (40 CFR Part 60, Subparts Cc and WWW), the importance of methane emissions in greenhouse gas inventories, and the market for methane as an energy source. Methane production occurs in the methanogenic stage of decomposition, as methanogenic bacteria break down the fermentation products from earlier decomposition processes. Since methane emissions result from waste decomposition, the quantity and duration of methane emissions is dependent on the same factors that influence waste degradability (e.g., waste composition, moisture).

Carbon dioxide is produced in the initial aerobic stage and anaerobic acid stage of decomposition. However, relatively little research has been conducted to quantify CO<sub>2</sub> emissions during these stages. Emissions during the aerobic stage are generally assumed to be a small proportion of total organic carbon inputs, and the authors performed a screening level analysis indicating that less than 1 percent of carbon is likely to be emitted through this pathway. Once the methanogenic stage of decomposition begins, landfill gas *as generated* is comprised of approximately 50 percent methane and 50 percent carbon dioxide (Bingemer and Crutzen 1987) But landfill gas *as collected* generally has a higher CH<sub>4</sub> concentration than CO<sub>2</sub> concentration (sometimes as much as a 60%:40% ratio), because some of the CO<sub>2</sub> is dissolved in the leachate as part of the carbonate system (CO<sub>2</sub> ↔ H<sub>2</sub>CO<sub>3</sub> ↔ HCO<sub>3</sub><sup>-</sup> ↔ CO<sub>3</sub><sup>2-</sup>).

To date, very little research has been conducted on the role of volatile organic compound emissions in the landfill carbon mass balance. Given the thousands of compounds entering the landfill environment, tracking the biochemistry by which these compounds ultimately get converted to VOC is a complex undertaking. Existing research indicates that ethane, limonene, *n*-decane, *p*-dichlorobenzene, and toluene may be among the most abundant landfill VOCs (Eklund, et al. 1998). Hartog (2003) reported non-methane volatile organic compound concentrations in landfill gas at a bioreactor site in Iowa, averaging 1,700 ppm carbon by volume in 2001 and 925 ppm carbon by volume in 2002. Assuming that VOC concentrations in landfill gas are in the order of magnitude of 1000 ppm, carbon flux as CH<sub>4</sub> and CO<sub>2</sub> will each be hundreds of times larger than this, so VOCs probably have a small role in the overall carbon balance.

Leachate is produced as water percolates through landfills. Factors affecting leachate formation include the quantity of water entering the landfill, waste composition, and the degree of decomposition. Since it may contain materials capable of contaminating groundwater, leachate (and the carbon it contains) is typically collected and treated before being released to the environment, where it eventually degrades into carbon dioxide. However, leachate is increasingly being recycled into the landfill as a means of inexpensive disposal and to promote decomposition while the containment system is operating at peak efficiency (Chan, et al. 2002). Research shows that this recirculation can increase the mass of organics collected by the system, and consequently enhance aqueous degradation (Warith 1999). While a great body of literature exists on landfill leachate formation, little research exists on the carbon implications of this process. Based on a screening analysis, the authors found that loss as leachate may be the fate for less than 1 percent of total carbon inputs to landfills.

In mass balance terms, carbon storage can be characterized as the carbon that remains after accounting for the carbon exiting the system as landfill gas or dissolved in leachate. On a dry weight basis, municipal refuse contains 30-50% cellulose, 7-12% hemicellulose and 15-28% lignin (Hilger and Barlaz 2001). While the degradation of cellulose and hemicellulose in landfills is well documented, lignin does not degrade to a significant extent under anaerobic conditions (Colberg 1988). In fact, while cellulose and hemicellulose biodegradation does occur, the extent of decomposition varies with landfill conditions and these materials do not appear to completely degrade based on a number of excavation studies (Ham and Bookter 1982, Wang, et al. 1994, Ham, et al. 1993). In addition, the presence of lignin actually prevents some cellulose and hemicellulose biodegradation. Thus, landfills in effect store some of the cellulose and hemicellulose and all of the lignin that is buried initially. The amount of storage will vary with environmental conditions in the landfill, with pH and moisture content identified as the two most important variables controlling decomposition (Barlaz, et al. 1990).

Before describing the approaches used to characterize landfill carbon storage for HWP, yard trimmings, and food scraps, it is useful to define some of the terms used in the discussion. Total weight as disposed – i.e., wet weight – includes moisture and total dry matter. Because the moisture content of a given waste component may be quite variable depending on weather conditions at the time of collection, most analysts characterize composition in terms of the proportions of dry matter. For purposes of landfill carbon storage, dry matter can be subdivided into persistent carbon (i.e., carbon stored after all degradation is complete), degradable carbon (i.e., carbon that is potentially converted to CH<sub>4</sub>, CO<sub>2</sub>, VOCs, or dissolved in leachate), and other dry matter. In the ensuing discussion, we present information on carbon storage as a proportion of total dry matter.

### 3. Quantifying Carbon Storage in the US – Landfilled Harvested Wood Products

Harvested wood products (HWP) include any industrial products made from harvested trees, including lumber, wood panels, paper, and paperboard. Every year, HWP are manufactured and placed into end uses in the US. Some of those HWP are taken out of the pool of products in use, and some portion is sent to landfills (and, in past years, dumps). There is some decay (emissions) from this pool of HWP in landfills or dumps, but as noted above, after the decay process ends, some biogenic carbon remains.

The revised 1996 IPCC guidelines on GHG inventories (1997) provide the following basic framework for estimating carbon storage in HWP:

...the recommended default assumption is that all carbon in biomass harvested is oxidised in the removal year. This is based on the perception that stocks of forest products in most countries are not increasing significantly on an annual basis. ... The proposed method recommends that storage of carbon in forest products be included in a national inventory only in the case where a country can document that existing stocks of long term forest products are in fact increasing. If data permit, one could add...to...the changes in forest and other woody biomass stocks...to account for increases in the pool of forest products. This information would, of course, require careful documentation, including accounting for imports and exports of forest products during the inventory period. (IPCC 1997)

More recently, IPCC (2000) has provided good practice guidance for how countries should estimate carbon emissions and changes in sinks for various sectors. Efforts are underway to provide guidance for estimating carbon change related to land use, land use change, and forestry that could include guidance related to HWP (Skog et al. 2003).

Three alternate accounting methods have been identified (IPCC 1999) for estimating changes in the stock of carbon in wood product pools within a country. Each of these methods defines the system boundary of the product pool differently.

- The Stock Change Approach defines a boundary around the country and includes carbon additions from imports, but excludes carbon in exports.
- The Atmospheric Flow Approach defines the system boundaries to include removals of carbon from the atmosphere to wood carbon stocks in a country, and equals the Stock Change amount minus net imports of wood product carbon for the year.
- The Production Approach effectively defines the system boundary around forest lands such that it includes net additions to all stocks of wood product carbon that came from trees in the home country, including additions in the home country and additions in other countries where products were exported.

The IPCC has not yet made a recommendation on which, if any, of the three accounting approaches is good practice for reporting additions to wood carbon stocks. So far, the US has been reporting the change in carbon using the Production Approach. That is, an estimate is provided of the amount of wood carbon stored in products in use and products in landfills that originated from trees harvested in the U.S. This estimate includes carbon stored in products that are exported (but it does not include carbon in products that are imported). The selection of the Production Approach by analysts does not indicate any formal recommendation by the US that this method should be preferred generally in reporting changes in carbon stored in wood and paper products. The difference in estimates for the US using various approaches is relatively small (less than 10 percent) because wood product imports and exports are relatively small compared to domestic production.

#### **Method**

Net changes in carbon stored in HWP in use and in dumps or landfills is estimated by tracking the disposition of HWP produced, imported, and exported beginning in 1910 and continuing up to the present using the WOODCARB model (Skog and Nicholson 2000).

Solid wood products, including lumber and panels, are tracked to nine end uses, where each is assigned a unique life span. Paper products are not tracked to end uses, but the time in use is estimated directly for various primary products.

When products are initially placed in end uses, such as construction or publications, we assume 8 percent loss (to disposal) for solid wood products and 5 percent for paper and paperboard products. Subsequently, the rate of retirement (to disposal) is simulated as a linear decay process where the initial mass is lost at a constant rate up to the maximum lifetime of the product, when (by definition) none of the mass remains. The maximum lifetime for solid wood products varies from 200 years for single-family homes to 12 years for pallets. The maximum lifetime is about 90 years (weighted by the mass of carbon in products). The maximum lifetime for free sheet paper (publication paper) is 12 years; for all other paper it is 2 years.

When wood or paper is discarded, it is tracked to recycling, disposal in landfills or dumps, or emission by burning. We estimate approximately 24 percent of discarded paper and paperboard (after recycling) was burned in 1993, and this percentage increases to 26 percent by the year 2000 (U.S. EPA 1994).

Prior to 1972, we assume dumps were used to dispose of waste. Between 1972 and 1986 there was a shift to disposal in landfills. For materials placed in dumps we estimated 65 percent was burned and the remainder decayed completely over a number of years, so there is no carbon storage or methane generation. For landfills, decay of wood and paper is more limited as oxygen is depleted. As shown in Table 1, the portion of wood carbon that decays to carbon dioxide and methane is estimated to be 3 percent and the portion of paper carbon that decays is 16 to 38 percent depending on the type of paper (Micales and Skog 1997). The half-life of decay in landfills is assumed to be 20 years for both paper and solid wood.

**Table 1: Estimate of the proportion of HWP carbon converted to methane and carbon dioxide in landfills. From Micales and Skog, 1997.**

Carbon source	Methane released (gm CH <sub>4</sub> /dry gm)	Fraction carbon released as CH <sub>4</sub>	Fraction carbon released as CO <sub>2</sub>	Total	Fraction of carbon stored	Fraction of dry matter stored*
				fraction released as landfill gas		
Wood	0.00 - 0.013	0 - 1.9%	0 - 1.3%	0 - 3.2%	97 - 100%	48 - 50%
Paper (average)	0.09	16%	10%	26%	74%	37%
Newspaper	0.54	9%	6%	16%	84%	42%
Boxes	0.108	19%	13%	32%	69%	35%
Office paper	0.131	23%	15%	38%	62%	31%
Coated paper	0.06	11%	7%	18%	83%	42%

We assume exported products have the same use life, disposition after use, and landfill decay characteristics as in the U.S.

## Results

Table 2 provides results for 1990–2002 for the four carbon pools that are tracked in the WOODCARB model – HWP in use, HWP in dumps/ landfills, HWP burned with energy production, and HWP decayed or burned. As the table shows, the annual change in stocks of carbon in landfilled HWP has been relatively constant in the range of 41 to 44 Tg C over the period from 1990-2002 (i.e., the rate of decomposition has been counterbalanced by the pace of ongoing disposal).

**Table 2: Annual Net Additions of Wood Product Carbon to Four Pools in the US, 1990 to 2002 (Tg C Eq.)**

Year	Products in Use	Dumps/ Landfills	Burned for Energy	Decay or Other Burning
1990	13	44	49	34
1991	11	43	48	34
1992	13	43	51	35
1993	15	41	47	36
1994	17	41	46	34
1995	15	41	47	36
1996	15	41	49	38
1997	16	42	51	37
1998	14	42	50	37
1999	17	42	51	37
2000	16	41	50	37
2001	16	42	52	37
2002	16	42	52	38

From a long-term perspective, the rate of annual net additions of carbon to products in use and landfills increased dramatically between 1970 and 2001—from 22 to 58 Tg per year. This increase was due in part to the increase in product consumption; roundwood use increased 150 percent between 1970 and 1991, from  $35 \times 10^7 \text{ m}^3$  ( $12.5 \text{ to } 18.7 \times 10^9 \text{ ft}^3$ ) (Heath and Birdsey 1993). It is also due to a sharp increase in the rate of accumulation of carbon in landfills with the shift from dumps to landfills in the 1970s and 1980s. Net accumulation in dumps or landfills increased from 9 Tg per year in 1970 to 42 Tg per year in 2002. This increase was due to elimination of open air burning in dumps and a decrease in the amount of decay of wood and paper in landfills compared with that in dumps.

The estimate of 57 Tg carbon added to landfills, dumps, and products in use in 1990 is supported by the research of Winjum, et al. (1998), who used data from the UN Food and Agriculture Organization (FAO) to develop an independent estimate that 57 Tg C were stored in wood and paper products for 5 years or more in the U.S.

#### **4. Quantifying Carbon Storage in the US – Yard Trimmings and Food Scraps**

As is the case with carbon in landfilled forest products, carbon contained in landfilled yard trimmings and food scraps can be stored indefinitely. In the United States, yard trimmings (i.e., grass clippings, leaves, branches) and food scraps comprise a significant portion of the municipal waste stream, and a large fraction of the collected yard trimmings and food scraps are discarded in landfills. For yard trimmings, both the amount collected annually and the fraction that is landfilled have declined over the last decade. In 1990, nearly 32 million metric tons (wet weight) of yard trimmings were generated (i.e., put at the curb for collection or taken to disposal or composting facilities) (U.S. EPA 2003). Since then, programs banning or discouraging disposal have led to an increase in backyard composting and the use of mulching mowers, and a consequent 20 percent decrease in the amount of yard trimmings collected. At the same time, a dramatic increase in the number of municipal yard waste composting facilities has reduced the proportion of collected yard trimmings that are discarded in landfills—from 72 percent in 1990 to 34 percent in 2001. There is considerably less centralized composting of food scraps; generation has grown by 26 percent since 1990, though the proportion of food scraps discarded in landfills has

decreased slightly from 81 percent in 1990 to 77 percent in 2002. Overall, there has been a decrease in the net yard trimmings and food scrap landfill disposal rate (U.S. EPA 2004).

The IPCC does not provide specific guidance on estimating carbon storage associated with landfilled yard trimmings. The U.S. Inventory quantifies this sink by applying a carbon storage factor to the tons of yard trimmings landfilled.

### **Method**

The U.S. Inventory methodology for estimating carbon storage in landfilled yard trimmings and food scraps draws from a life-cycle analysis of GHG emissions and sinks associated with solid waste management (U.S. EPA 2002b). Estimates of net carbon flux resulting from landfilled yard trimmings and food scraps were developed by estimating the change in landfilled carbon stocks between inventory years. Carbon stock estimates were calculated by determining the mass of landfilled carbon resulting from yard trimmings or food scraps discarded in a given year; adding the accumulated landfilled carbon from previous years; and subtracting the portion of carbon landfilled in previous years that decomposed.

To determine the total landfilled carbon stocks for a given year, the following were estimated: 1) the composition of the yard trimmings, 2) the mass of yard trimmings and food scraps discarded in landfills, 3) the carbon storage factor (CSF) of the landfilled yard trimmings and food scraps, and 4) the rate of decomposition of the degradable carbon. The composition of yard trimmings was assumed to be 30 percent grass clippings, 40 percent leaves, and 30 percent branches on a wet weight basis (Oshins and Block 2000) to determine a weighted average CSF. The yard trimmings were subdivided because each component has its own unique carbon storage factor and rate of decomposition. The mass of yard trimmings and food scraps disposed of in landfills was estimated by multiplying the quantity of yard trimmings and food scraps discarded by the proportion of discards managed in landfills. Data on discards (i.e., the amount generated minus the amount diverted to centralized composting facilities) for both yard trimmings and food scraps were taken primarily from *Municipal Solid Waste in the United States: 2001 Facts and Figures* (U.S. EPA 2003) and earlier versions of this report. These reports do not subdivide discards of individual materials into volumes landfilled and combusted, although they provide an estimate of the proportion of overall wastestream discards managed in landfills and combustors (i.e., ranging from 81 percent and 19 percent respectively in 1990, to 79 percent and 21 percent in 2001).

The amount of carbon disposed of in landfills each year, starting in 1960, was estimated by converting the discarded landfilled yard trimmings and food scraps from a wet weight to a dry weight basis, and then multiplying by the initial (i.e., pre-decomposition) carbon content (as a fraction of dry weight). The dry weight of landfilled material was calculated using dry weight to wet weight ratios (Tchobanoglous, et al. 1993 cited by Barlaz 1998) and the initial carbon contents were determined by Barlaz (1998).

The amount of carbon remaining in the landfill for each subsequent year was tracked based on a simple model of carbon fate. According to Barlaz (1998), a portion of the initial carbon resists decomposition and is essentially persistent in the landfill environment; the modeling approach applied here builds on his findings. Barlaz (1998) and coworkers at North Carolina State University conducted a series of laboratory experiments designed to measure biodegradation of yard trimmings, food scraps, and other materials, in conditions designed to promote decomposition (i.e., by providing ample moisture and nutrients). Specific waste components (e.g., grass, branches, leaves, paper) were dried, analyzed for cellulose, hemicellulose, and lignin content, weighed, placed in two-liter plastic containers (i.e., reactors), and allowed to decompose anaerobically under moist conditions (Barlaz, et al. 1997). The reactors were seeded with a small amount of well-decomposed refuse containing an active population of microorganisms. Phosphate and nitrogen concentrations were maintained at sufficient levels to assure that they were not limiting factors for biodegradation. The reactors were allowed to run until either no more methane was produced or an extrapolation of gas production data indicated that the reactors had produced 95 percent of the methane that would ultimately be emitted if allowed to run forever. At the end of the experiment, the contents of the reactors were dried, weighed, and analyzed for cellulose, hemicellulose, and lignin content. Once decomposition was complete, the yard trimmings and food scraps were re-analyzed for carbon content. The mass of carbon remaining, divided by the original dry weight of the material, was reported as the carbon storage factor (CSF).



Some inconsistencies existed in the carbon mass balance in the experiments, which included initial carbon (measured), carbon emitted as CH<sub>4</sub> (measured), carbon emitted as CO<sub>2</sub> (assumed to equal CH<sub>4</sub>-C), carbon in leachate (in the form of chemical oxygen demand, measured), and carbon remaining (Eleazer, et al. 1997). The most significant discrepancy was for leaves, where the carbon storage factor (0.54 kg C per dry kg) was higher than the initial carbon content (0.49 kg C per dry kg). We adjusted the reported CSF by assuming that initial C was correct, and that final C could be calculated as initial C minus C emitted as CH<sub>4</sub> and CO<sub>2</sub>. We only corrected the mass balance for leaves; for grass, branches, and food scraps we used carbon storage factors as reported by Barlaz (1998). The resulting storage factors, again in terms of C remaining as a fraction of initial dry weight, were 0.32, 0.46, 0.38, and 0.08 kg C sequestered per dry kg for grass, leaves, branches, and food scraps, respectively. This method will give the minimum possible estimate of storage.

For purposes of the inventory analysis, these were converted from a dry weight basis to wet weight basis using Barlaz's reported moisture contents of 70 percent, 30 percent, 10 percent, and 70 percent, for grass, leaves, branches, and food scraps, respectively. To simulate U.S. landfill carbon flows, the CSFs are divided by the initial carbon content to determine the proportion of initial carbon that does not decompose. The remaining portion is assumed to degrade (and results in emissions of methane and carbon dioxide). For example, for branches Barlaz (1998) reported the CSF as 38 percent (of dry weight), and the initial carbon content as 49 percent (of dry weight). Thus, the proportion of initial carbon that does not decompose is  $0.38 / 0.49 = 77$  percent. The remaining 23 percent degrades.

The degradable portion of the carbon is assumed to decay according to first order kinetics. Grass and food scraps are assumed to have a half-life of 5 years; leaves and branches are assumed to have a half-life of 20 years.

For each of the four materials (grass, leaves, branches, food scraps), the stock of carbon in landfills for any given year is calculated according to the following formula:

$$LFC_{i,t} = \sum_n W_{i,n} * (1 - MC_i) * ICC_i * \{ [CSF_i / ICC_i] + [(1 - (CSF_i / ICC_i)) * e^{-k*(t-n)}] \}$$

where,

t is the year for which carbon stocks are being estimated,

LFC<sub>i,t</sub> is the stock of carbon in landfills in year t, for waste i (grass, leaves, branches, food scraps)

W<sub>i,n</sub> is the mass of waste i disposed in landfills in year n, in units of wet weight

n is the year in which the waste was disposed, where  $1960 \leq n \leq t$

MC<sub>i</sub> is moisture content of waste i,

ICC<sub>i</sub> is the initial carbon content of waste i,

CSF<sub>i</sub> is the carbon storage factor of waste i,

e is the natural logarithm,

k is the first order rate constant for waste i, and is equal to 0.693 divided by the half-life for decomposition.

For a given year t, the total stock of carbon in landfills (TLFC<sub>t</sub>) is the sum of stocks across all four materials. The annual flux of carbon in landfills (F<sub>t</sub>) for year t is calculated as the change in stock compared to the preceding year:

$$F_t = TLFC_t - TLFC_{t-1}$$

Thus, the carbon placed in a landfill in year n is tracked for each year t through the end of the inventory period (2002). For example, disposal of food scraps in 1960 resulted in depositing about 1,140,000 tonnes of carbon. Of this amount, 16 percent (180,000 tonnes) is persistent; the remaining 84 percent (960,000 tonnes) is degradable. By 1965, half of the degradable portion (480,000 tonnes)

decomposes, leaving a total of 660,000 tonnes (the persistent portion, plus the remaining half of the degradable portion).

Continuing the example, by 2002, the total food scraps carbon originally disposed in 1960 had declined to 182,000 tonnes (i.e., virtually all of the degradable carbon had decomposed). By summing the carbon remaining from 1960 with the carbon remaining from food scraps disposed in subsequent years (1961 – 2002), the total landfill carbon from food scraps in 2002 was 28.7 million tonnes. This value is then added to the carbon stock from grass, leaves, and branches to calculate the total landfill carbon stock in 2002, yielding a value of 239.6 million tons. In exactly the same way total net flux is calculated for forest carbon and harvested wood products, the total net flux of landfill carbon for yard trimmings and food scraps for a given year is the difference in the landfill carbon stock for a given year minus the stock in the preceding year. For example, the net change in 2002 (2.8 Tg C) is equal to the stock in 2002 (239.6 Tg C) minus the stock in 2001 (236.8 Tg C).

When applying the CSF data reported by Barlaz (1998), an adjustment was made to the reported value for leaves, because the CSF was higher than the initial carbon content. This anomalous result, probably due to errors in the laboratory measurements, was addressed by applying a mass balance calculation, and assuming that (a) the initial carbon content was correctly measured, and (b) the CSF was incorrect. The same experiment measured not only the persistence of carbon (i.e., the CSF), but also the yield of methane for each of the individual waste materials (Eleazer et al. 1997). The anaerobic decomposition process results in release of equal molar volumes of methane and carbon dioxide. Thus, to derive a more realistic estimate of the carbon storage factor for leaves, the carbon released in the form of methane during decomposition was multiplied by two (to include the loss of carbon through carbon dioxide), and then subtracted from the initial carbon content of the leaves. This estimate of carbon remaining was used to derive the carbon storage factor (0.46).

**Table 3: Carbon Storage Factors for Landfilled Yard Trimmings and Food Scraps**

<b>Component</b>	<b>Storage Factor (kg C/kg dry yard trimmings)</b>	<b>Moisture Content (kg water/kg wet yard trimmings)</b>	<b>Composition (Percent wet weight)</b>
Grass	0.32	0.70	30
Leaves	0.46 <sup>a</sup>	0.30	40
Branches	0.38	0.10	30
Food Scraps	0.08	0.70	NA

<sup>a</sup> Modified from Barlaz (1998), adjusted using CH<sub>4</sub> yields in Eleazer, et al. (1997).

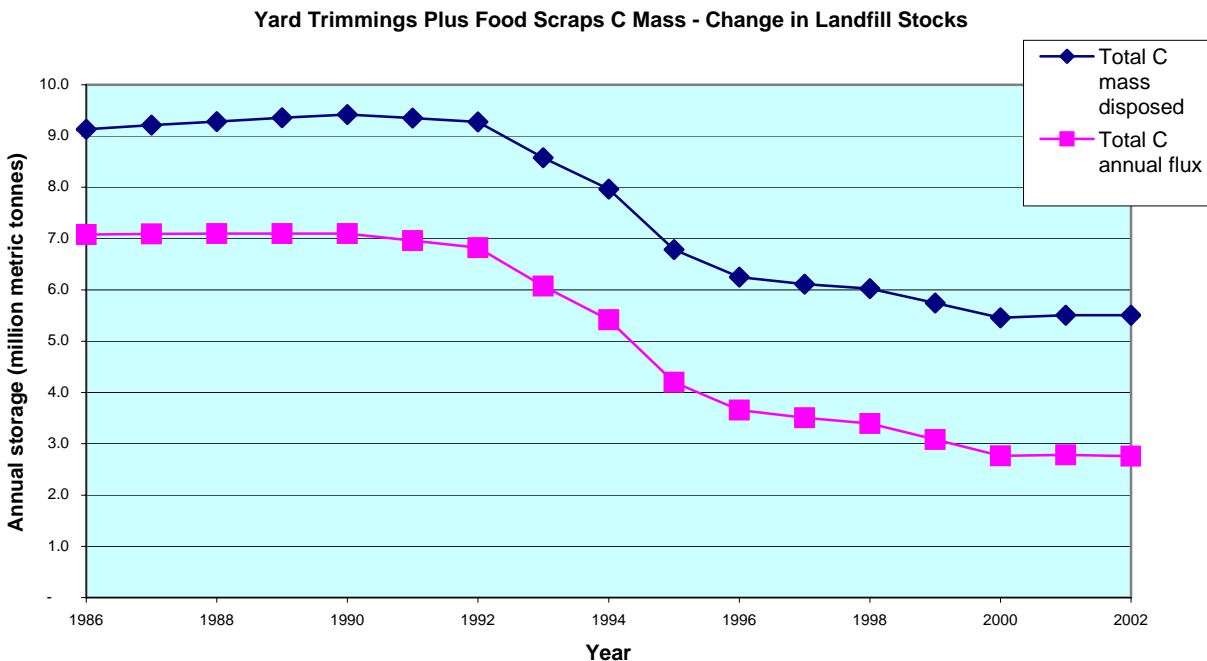
## Results

As reported in the U.S. Inventory, landfill carbon storage of yard trimmings and food scraps has decreased from about 7.1 Tg C in 1990 to 2.8 Tg C in 2002. The decline in the rate of landfill carbon storage is primarily due to an increase in programs that encourage composting or disposal of yard trimmings in landfills (U.S. EPA 2002a). Annual estimates for yard trimmings and food scraps are provided in Table 4, along with estimates for HWP. As the table shows, landfilling of these materials contributes much less to overall carbon storage than landfilling of wood and paper.

**Table 4: Mass of Carbon Stored in Landfills from Harvested Wood Products and Yard Trimmings in the U.S. (Tg C)**

Landfilled Material	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002
Wood	44.3	42.9	42.6	40.7	40.8	40.7	41.3	42.3	42.1	41.8	41.5	42.2	42.3
Yard Trimmings	6.3	6.2	6.1	5.4	4.7	3.7	3.1	2.8	2.6	2.3	2.0	2.0	2.0
Food Scraps	0.8	0.7	0.7	0.7	0.7	0.5	0.6	0.7	0.8	0.8	0.8	0.8	0.7
<b>Total Landfill Carbon Storage</b>	<b>51.4</b>	<b>49.8</b>	<b>49.4</b>	<b>46.8</b>	<b>46.2</b>	<b>44.9</b>	<b>45.0</b>	<b>45.8</b>	<b>45.4</b>	<b>44.9</b>	<b>44.3</b>	<b>45.0</b>	<b>45.1</b>

**Figure 2. Comparison of Total Carbon Disposed and Total Carbon Flux for Yard Trimmings and Food Scraps.**



### 5. Uncertainties in the Landfill Carbon Storage Estimates

The state-of-the-art for predicting landfill carbon storage is still in its infancy, and is based on a handful of studies on decomposition behavior in landfills. Uncertainty in the landfill carbon storage estimates stems primarily from the quality and comprehensiveness of data. Predictions of long-term storage in landfilled HWP and yard trimmings are based on limited laboratory studies conducted under experimental conditions intended to provide favorable conditions for anaerobic decomposition, and which are thus likely to understate storage. In field conditions, there may actually be considerable heterogeneity in the rates of carbon storage.

Skog, et al. (2003) developed a method to evaluate uncertainty for the entire analytic framework for carbon stored in HWP, including imports, exports, products in use, and disposal. The uncertainty analysis uses probability density functions for 14 variables, and Monte Carlo simulation to provide the distribution of estimates. Using data on HWP production and trade (starting several decades in the past), they tracked annual additions to pools of HWP in use, removals from use, additions to solid waste disposal sites (SWDS), and decay from SWDS. Results suggest that total uncertainty is most sensitive to uncertainty in production data for solidwood products, the carbon content estimates (i.e., factors used to

convert product volumes to carbon mass), and the proportions of solidwood and paper in SWDS. The overall results were not as sensitive to the proportion of total carbon that is degradable, and the rate of decay (the two factors most relevant to landfill carbon storage).

Another source of uncertainty relates to the variability in moisture content in landfills. Both climate and landfill operation affect moisture content, which is one of the principal factors controlling the decay rate of organic materials. Climatic conditions at U.S. landfill sites span a broad range. In addition, the presence of a final landfill cover has a profound effect on the availability of moisture. During active waste disposal operations, daily cover and intermediate cover (generally, a few inches of loose soil applied at the end of a day's operation) are not effective in reducing infiltration. However, final cover must meet rigorous standards in terms of permeability, grading, and maintenance. Typically, after final cover has been put in place, systems with leachate collection systems see a dramatic decrease in the volumes of leachate collected.

As noted earlier, carbon storage from other sources of biogenic carbon (carbon derived from a natural source) is not accounted for in the U.S. Inventory. Examples of these sources include construction and demolition debris and biosolids. Including these sources would increase the landfill carbon storage estimates.

Despite these limitations, the methods are consistent with the objectives of IPCC's Good Practice Guidance (2000) – that is, they provide an unbiased estimate with the best available information.

## 6. Conclusions

In the US, landfill carbon storage due to HWP and yard trimmings accounted for about 45 Tg C Eq. in 2002 (42 Tg C from HWP and 3 Tg C from yard trimmings and food scraps), or 24 percent of total estimated carbon storage (U.S. EPA 2004) from all land-use activities.<sup>1</sup> Excluding this source from national emissions inventories could significantly underestimate total carbon sinks in the country, and could overestimate net emissions (in the US omitting landfill carbon storage would increase net emissions by 2.4%). Particularly in countries where landfills function as a significant component of a nation's waste management strategy, accounting for the carbon storage pools created in landfills could provide a more accurate estimate of net emissions.

As noted earlier, our suggestion to include landfill carbon as a removal process in national GHG inventories should not be interpreted as an endorsement of landfilling as an unambiguously beneficial approach to solid waste management. Putting aside the resource recovery, odor, disease, and ground water issues associated with landfills, from the GHG perspective, methane emissions are likely to outweigh carbon storage. In the US, even with landfill gas control systems at many landfills, methane emissions from landfills totaled 52.6 Tg C Eq. in 2002, accounting for 32 percent of total methane emissions and nearly 3 percent of total net emissions by all gases (and exceeding landfill carbon storage) (U.S. EPA 2004).

Moreover, the methane emissions avoided by employing other waste management options – such as recycling, composting, and source reduction – can often outweigh the benefits of carbon storage in landfills. Recycled materials used in place of virgin raw material inputs can reduce GHG emissions from energy intensive processes such as mining or extraction of raw materials. In addition, “remufacturing” processes (i.e., manufacturing with recycled inputs) are generally less energy-intensive than manufacturing with virgin inputs. For organic wastes, composting is another option. Composting facilities produce little to no methane emissions and minimal fossil carbon dioxide emissions (mainly from machinery), and they promote storage of carbon in soils. Source reduction – which refers to the lightweighting of packaging, reuse of materials, and other such techniques to reduce waste – can result in significant GHG reductions due to avoided emissions from raw materials acquisition and manufacturing

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<sup>1</sup> These activities include 1) changes in forest carbon stocks (including wood products), 2) changes in carbon stocks in urban trees, 3) changes in agricultural soil carbon stocks, and 4) changes in carbon stocks in landfilled yard trimmings.

as well as those associated with disposal. Weighing the relative impacts of each of these waste management practices, on a life-cycle basis, can help communities and countries determine if carbon storage or methane reductions will prove most beneficial (U.S. EPA 2002b; Solano, et al. 2002).

As with many other GHG inventory methodologies, estimating landfill carbon storage is fraught with uncertainty. Additional research on long-term carbon fate in landfills and the effect of management systems (e.g., containment, landfill gas) on carbon storage is needed. Nevertheless, even when fully recognizing the uncertainties, it is clear that landfill carbon storage is a large-scale process that deserves careful consideration for inclusion in national GHG inventories.

As a final note, although it is clear that carbon storage in landfills is a large-scale process that, to some extent, offsets GHG emissions (including methane emissions from landfills), the authors do not suggest that landfilling is preferable to waste prevention, recycling, or combustion (which may have lower GHG emissions when viewed from a life-cycle perspective). Indeed, material that is landfilled represents a lost opportunity for continued use within the materials system. Rather, the intent of this paper is to describe a process that is significant from the perspective of GHG accounting, and to report the methods that are currently used to characterize it.

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