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Evaluation of landfill gas emissions from municipal solid waste landfills for the life-cycle analysis of waste-to-energy pathways

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ABSTRACT

Various waste-to-energy (WTE) conversion technologies can generate energy products from municipal solid waste (MSW). Accurately evaluating landfill gas (LFG, mainly methane) emissions from base case landfills is critical to conducting a WTE life-cycle analysis (LCA) of their greenhouse gas (GHG) emissions. To reduce uncertainties in estimating LFG, this study investigated key parameters for its generation, based on updated experimental results. These results showed that the updated parameters changed the calculated GHG emissions from landfills significantly depending on waste stream; they resulted in a 65% reduction for wood (from 2412 to 848 t CO_2e/dry t) to a 4% increase for food waste (from 2603 to 2708 t CO_2e/dry t). Landfill GHG emissions also vary significantly based on LFG management practices and climate. In LCAs of WTE conversion, generating electricity from LFG helps reduce GHG emissions indirectly by displacing regional electricity. When both active LFG collection and power generation are considered, GHG emissions are 44% less for food waste (from 2708 to 1524 t CO_2e/dry t), relative to conventional MSW landfilling. The method developed and data collected in this study can help improve the assessment of GHG impacts from landfills, which supports transparent decision-making regarding the sustainable treatment, management, and utilization of MSW.

1. Introduction

An estimated 234 million metric tons (Mt) of municipal solid waste (MSW) generated in the United States in 2014, 52.6% of which (123 Mt) was discarded in landfills (USEPA, 2016a). Because of its considerable energy potential and high organic content, MSW has received increasing interest as a feedstock for fuel and energy production (i.e., waste-to-energy [WTE]). As the U.S. Department of Energy (DOE) recently stated, using MSW for fuel and energy production has several advantages (USDOE, 2017). For example, waste feedstocks are available at low prices, or even at negative prices considering tipping fees. Waste feedstocks also can be collected using the current infrastructure for waste collection and separation, which further lowers the cost of waste-derived energy products. In addition to these economic advantages, diverting waste feedstocks from landfills for energy production avoids the emissions that otherwise would occur with landfilling. The U.S. Environmental Protection Agency (USEPA) reported that greenhouse gas (GHG) emissions from waste landfills amounted to 115.7 Mt of carbon dioxide equivalent (CO_2e) in 2015 (USEPA, 2017). Waste-derived fuels can displace conventional fossil fuels, and avoiding the energy use and emissions associated with the production of the fossil fuels can provide additional benefits.

To take advantage of these benefits, several biochemical (e.g., anaerobic digestion and fermentation) and thermochemical (e.g., hydrothermal liquefaction, pyrolysis, and gasification) processes are currently being researched to convert MSW to fuels. For example, anaerobic digestion has been used to produce biogas and renewable natural gas from food waste (Lee et al., 2016). Fermentation processes that generate bioethanol from MSW also have been investigated (Lee et al., 2016). Both pyrolysis and gasification processes convert MSW to fuel using thermochemical processes. Pyrolysis processes convert waste into bio-char, bio-oil, and gases (Chen et al., 2015), and this bio-oil can be further hydroprocessed to produce gasoline and diesel blendstocks (Wang et al., 2015). The gasification process generates syngas, which can be converted into various fuels (e.g., Fischer-Tropsch diesel and jet) (Lee et al., 2014). Hydrothermal liquefaction is a way of generating liquid fuels from organic materials such as MSW (Dimitriadis and Bezergianni, 2017).

Life-cycle analyses (LCAs) have been conducted to evaluate the energy and environmental impacts of these MSW-based fuel production pathways. A major LCA issue for these pathways is treating

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carbon emissions. One approach is to use the carbon neutrality assumption (zero carbon emissions from the combustion of energy products) for carbon in organic waste as the Intergovernmental Panel on Climate Change (IPCC) and USEPA do (IPCC, 2008; USEPA, 2010). Using the assumption of carbon neutrality, Kalogo et al. (2007) examined a MSW-to-ethanol facility, Pressley et al. (2014) assessed the conversion of MSW to liquid fuel through gasification and Fishcer-Tropsch, and Vergara et al. (2011) evaluated five waste treatment strategies in California.

The other method is a marginal approach, which evaluates the impact of waste diversion on the production of MSW-based fuels. This approach was used for several waste management LCAs. Chester and Martin (2009) examined cellulosic ethanol generated from MSW. The California Air Resources Board (2016) estimated GHG emission reductions by diverting landfilled waste to compost facilities. Lee et al. (2016) studied compressed natural gas and ethanol production from MSW. For the marginal analysis approach, two scenarios are needed: a scenario where fuel is produced from waste (the alternative scenario), and a scenario that assumes business as usual (the counterfactual scenario). The marginal approach accounts only for the differences between the two scenarios to assess energy and the environmental effects of the alternative scenario. Usually, these LCAs assume that MSW used for energy and fuel production in the alternative scenario would otherwise be landfilled (i.e., the counterfactual scenario).

Emissions associated with landfilling waste need to be estimated for the counterfactual scenario. One way to estimate emissions from landfilled waste is to directly measure the emissions from landfills. However, in practice, emissions from a mixture of waste streams are usually measured together at a certain point in time, while an LCA study requires lifetime emissions from specific waste streams (e.g., food waste, yard trimmings). Another way is to use engineering models to estimate the generation, collection, and oxidation of landfill gas (LFG). For example, first-order decay models are commonly used to estimate LFG generation as suggested by the IPCC (IPCC, 2008) and the USEPA's Landfill Gas Emissions Model (LandGEM) (USEPA, 2005).

Given the estimated LFG generation, LFG collection efficiency and a methane (CH_4) oxidation factor are used to estimate LFG collection and oxidation. Estimated emissions that use these modeling approaches are highly sensitive to a few key parameters: LFG generation depends largely on the types of waste components and climate conditions, and CH_4 collection depends on decay speed over time, which varies widely among waste components, LFG collection strategies, landfill cover types, climate conditions, and oxidation factors. Several previous studies used the IPCC and USEPA methods. For example, Bogner and Matthews (2003) evaluated global CH_4 emissions from landfills, and Kennedy et al. (2010) estimated GHG emissions from global cities.

To conduct LCA of the WTE pathways, it may be necessary to estimate emissions of counterfactual scenarios for specific waste streams with specific parameters. This study evaluated key parameters to estimate the emissions specific to major landfilled organic waste types (i.e., paper, wood, food, and yard trimmings) using available experimental data to improve the accuracy of our LFG emission simulations under the counterfactual scenario. Because the emissions in the counterfactual scenario can be avoided if WTE technologies displace current landfills, this study will enhance the reliability of LCAs for various WTE pathways.

2. System boundary

Once organic waste is landfilled, it starts decomposing under anaerobic conditions and generates LFG, a mixture of CH_4 and carbon dioxide (CO_2). Simulations of LFG generation are based on the assumption that the decomposition of degradable carbon remaining in the landfill follows first-order decay characteristics, and the simulation parameters are adjusted using measured data. Once generated, LFG is collected and its CH_4 is combusted to reduce global warming impacts. During CH_4 combustion, landfill operators may generate electricity to improve their revenue instead of flaring LFG. In this case, it is assumed that regional electricity is displaced, which leads to reductions in GHG emissions because it avoids the emissions associated with regional electricity generation. Not all LFG generated can be collected, and some of it passes through landfill covers and is emitted to the atmosphere. While LFG goes through landfill covers, a portion of non-collected CH_4 oxidizes into CO_2. In summary, while CO_2 generated is emitted without being converted into other molecules—regardless of LFG collection conditions—a portion of CH_4 generated from landfilled waste is combusted or oxidized into CO_2.

Because CH_4 has a higher global warming potential (GWP) than CO_2, the fate of CH_4 is important in estimating the GHG emissions from landfilled waste. Fig. 1 represents the fate of CH_4 generated from waste decomposition, and LFG emissions are expressed as the sum of four emission components: (1) CO_2 emissions from collected CH_4 combustion, (2) non-collected CH_4 emissions, (3) CO_2 emissions from oxidized CH_4 in the landfill cover, and (4) CO_2 emissions from waste decomposition. Note that total carbon emissions in these emission components are determined only by the LFG generation process (i.e., decomposition of degradable organic carbon). The share of carbon emissions among these four emission components depends on the CH_4 concentration in LFG, LFG collection efficiency, and CH_4 oxidation factor. In order to

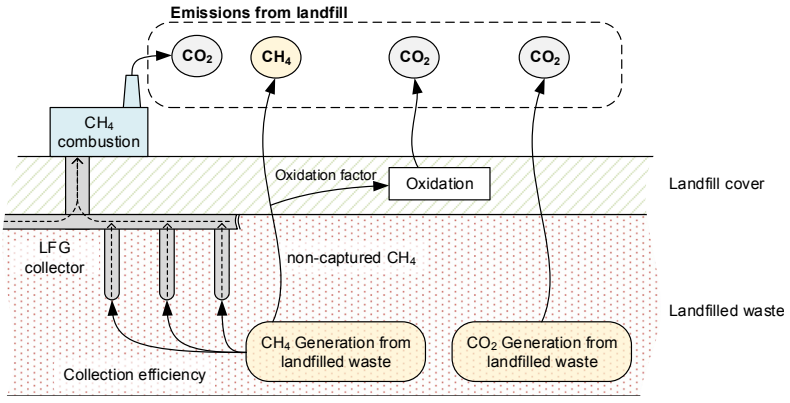


Fig. 1. Fate of LFG emissions generated from landfilled organic waste.

evaluate GHG emissions from landfills, LFG generation and composition, collection efficiency, and oxidation factor need to be investigated. These parameters are evaluated in Sections 3.1, 3.2, and 3.3.

Note that 75.5% of generated plastic wastes in the United States in 2014, an estimated at 22.8 t, were landfilled (USEPA, 2016a). Unlike degradable organic carbon in organic waste, inorganic carbon in plastics does not decompose in landfills; is sequestered, which leads to no GHG emissions. When non-recycled plastic is used for fuel production, there are no GHG emission credits from avoiding the landfilling plastics scenario.

3. Methodology and simulation parameters

3.1. Landfill gas generation and CH_4 concentration in LFG

The amount of carbon released from organic waste decomposition needs to be estimated. It was assumed that carbon is emitted as either CH_4 or CO_2 depending on the CH_4 concentration in the LFG. First, the amount of degradable carbon in landfilled waste is calculated by multiplying the wet weight of landfilled waste and the degradable organic carbon (*DOC*) content (wt% of wet waste). Note that not all carbon in degradable waste decomposes or converts to LFG. The fraction of the degradable carbon actually decomposed (DOC_F) is multiplied by the amount of degradable organic carbon in landfilled waste to estimate the total amount of carbon converted into LFG through waste decomposition. While the measured wet weight of degradable landfilled waste and *DOC* are used for the simulation, DOC_F is commonly set at 0.5 for generic MSW (IPCC, 2008). In other words, 50% of *DOC* in landfilled waste is assumed to decompose into LFG, and the rest stays in the landfill. Unlike the other measured parameters, this value is subject to huge uncertainties, and it is not reasonable to use fixed DOC_F for all types of wastes, considering that each waste component may have a different DOC_F value (IPCC, 2008). It is specifically important for WTE LCA to have DOC_F by type of waste, because WTE pathways generally use specific types of sorted wastes that are suitable for a given conversion process.

By definition, DOC_F is the amount of carbon in ultimate carbon emissions divided by the amount of carbon in organic material, where ultimate carbon emissions can be estimated by using the ultimate CH_4 emissions and CH_4 concentration in biogas. CH_4 emissions can be obtained by a biochemical methane potential (BMP) test, which examines the potential biodegradability of organic materials under optimal anaerobic digestion conditions. Fig. 2 summarizes the average DOC_F values for four major organic waste components (paper, wood, food, and yard trimmings) calculated from the literature, and the error bars indicate the lower and the upper end. Some studies do not specify the CH_4 concentration in biogas while reporting CH_4 generation. Among the studies reporting CH_4 concentration, however, the average, minimum, and maximum of CH_4 concentration are 60%, 54%, and 73%, respectively. These CH_4 concentrations were used to estimate the nominal, maximum, and minimum DOC_F, for the studies not reporting CH_4 concentration.

The estimated average DOC_F values for paper, wood, food waste, and yard trimmings are 0.37, 0.21, 0.64, and 0.23, respectively. DOC_F of typical mixed organic waste in the United States is estimated at 0.37, considering the *DOC* of each waste component and the composition of landfilled organic wastes estimated by the USEPA (2016a) (i.e., paper 14.3%, wood 8.1%, food 21.6%, and yard trimmings 7.9%). While this is slightly lower than the default DOC_F, the DOC_F of each feedstock is more important for the LCAs of WTE pathways because each WTE pathway typically requires a specific type of feedstock. It is expected that using a default DOC_F of 0.5 would underestimate GHG emissions from landfills for food waste, while the opposite would be expected for other feedstocks such as paper, wood, and yard trimmings. The ranges of DOC_F calculated from collected test results were 0.19–0.54, 0.02–0.57, 0.36–0.92, and 0.09–0.38 for paper, wood, food waste, and yard trimmings, respectively; these ranges were used to conduct a sensitivity analysis.

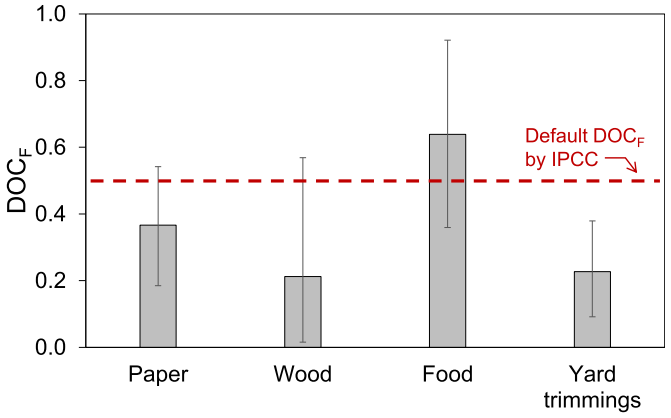


Fig. 2. Estimated DOC_F for waste components (paper, wood, food, and yard trimmings) using BMP test results. (Data sources: paper—Barlaz (1998), Eleazer et al., (1997), Jeon et al., (2007), Micales and Skog (1997), Owens and Chynoweth (1993); wood—Chynoweth et al., (1993), Jeon et al., (2007), Micales and Skog (1997); food—Barlaz (1998), Cho et al., (1995), Chynoweth et al., (1993), Eleazer et al., (1997), Jeon et al., (2007); yard trimmings—Barlaz (1998), Eleazer et al., (1997), Owens and Chynoweth (1993).)

Given the amount of carbon converted into LFG, the IPCC suggests estimating the total amount of CH_4 generation by multiplying the total amount of carbon converted into LFG with the methane correction factor (*MCF*), CH_4 concentration (*F*) in LFG, and carbon content of CH_4 (i.e., 16/12). The rest of the carbon decomposed is assumed to be converted into CO_2 (IPCC, 2008). Landfills cannot create the optimal anaerobic conditions under which the studies summarized in Fig. 2 were conducted. One way to account for this partial aerobic condition is to use an *MCF* lower than 1. The IPCC suggests default *MCF* values of 0.4 and 1 for unmanaged shallow landfills and managed landfills, respectively. Because landfills in the United States are managed well, an *MCF* of 1 is generally accepted (USEPA, 2017). For well-managed landfills like those in the United States, the CH_4 concentration monitored and measured in landfills can be used. The USEPA's Landfill Methane Outreach Program (LMOP) provides information on landfills in the United States (USEPA, 2016b); 1070 of a total of 2450 landfills reported the CH_4 concentration of LFG. The average and median CH_4 concentrations are 47.1% and 50%, respectively, while the 25th percentile (p25) and 75th percentile (p75) are 44.3% and 51.7%, respectively. Due to the suboptimal anaerobic conditions, the measured CH_4 concentration is lower than the CH_4 concentration from the studies used for Fig. 2 (54%–73%). Note that *F* values from various landfills are within a narrow range and do not vary meaningfully by region. The median value (50%), the same as the IPCC value, was used for the baseline case, and the lower and the upper ends were used for the sensitivity analysis.

3.2. Landfill gas collection

In order to control odor and to meet emission regulations, landfill operators collect and burn LFG. LFG collection efficiency varies mainly with the LFG collection strategy, such as LFG collector operation, types of landfill covers, and when the covers were

installed. A landfill commonly consists of multiple cells and is managed on a cell-by-cell basis in order to operate the entire landfill effectively (Lee et al., 2016). To evaluate the LFG collection efficiency of a landfill, LFG generation and collection at a cell level should be taken into account.

There are two types of LFG collectors: horizontal and vertical. Due to installation and physical constraints, horizontal collectors are commonly installed and used to collect LFG in the early stages of cell development, even while they are still accepting waste. Vertical collectors are typically installed and operated after the final covers have been installed. Because final covers have very low permeability and landfilled waste around the vertical collectors builds a high vacuum, the collection efficiency of vertical collectors is typically higher compared to horizontal collectors.

To estimate the overall LFG collection efficiency, LFG collection stages can be divided into three phases based on installation and operation of LFG collectors and landfill covers in individual cells: (1) Phase 1—active cells, (2) Phase 2—closed cells with temporary covers, and (3) Phase 3—closed cells with final covers. Collection efficiencies for Phase 1, Phase 2, and Phase 3 are usually set at 50%, 75%, and 95% (Barlaz et al., 2009), respectively. Because landfill collectors are expected to be operational for 30 years after closure, collection efficiency is set to zero before Phase 1 and after the post-closure care period. Lifetime collection efficiency can be estimated based on when each of these phases is operational. Interviews were conducted for landfill operators to reflect actual landfill operation, and two landfills were selected that represent moderate and active LFG collection. Their landfill operation conditions were used to simulate LFG collection and to calculate the lifetime LFG collection efficiency. Detailed information is available in our previous work (Lee et al., 2016).

Collection efficiency also can be influenced by the waste decay speed. Because LFG released would not be collected before collectors are operational, fast decomposition may lead to more CH_4 emissions. If the speed of waste decomposition is too slow, it may lose a lot of LFG after the post-closure care period. LFG generation over time was taken in to account to estimate the LFG collection efficiency.

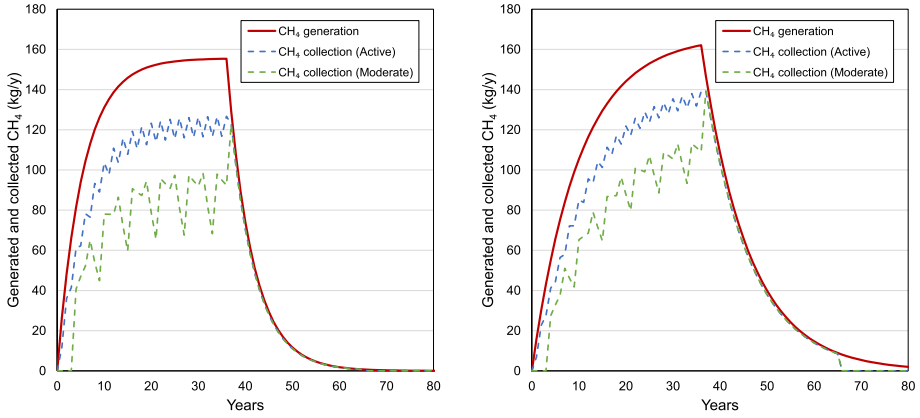
The amount of CH_4 generation from a unit mass of landfilled organic waste decreases over time exponentially following the first order decay curve. Decay speed rates depend on the type of waste and climate conditions. The IPCC classified decay speed rates with respect to type of waste under four different climate conditions (i.e., boreal/dry, boreal/wet, tropical/dry, and tropical/wet) (IPCC, 2008). For example, food waste under tropical and wet climate conditions would decompose the fastest, whereas it takes a long time for wood to decompose under boreal and dry climate conditions. This analysis used values provided by the IPCC (2008), because they are from measured data. De la Cruz and Barlaz (2010) also investigated waste component-specific decay rates, which can be used instead of the IPCC's decay speed rates.

To estimate annual CH_4 generation from landfills, a waste disposal scenario needs to be assumed. For example, Fig. 3 shows annual CH_4 generation and collection over time when 1 dry t of food waste and yard trimmings is landfilled every year under boreal and wet climate conditions for 36 y. The IPCC's default *DOC* and DOC_F values were used to estimate the CH_4 generation. The CH_4 generation shown by the red lines (Fig. 3) increases for 36 y (since new waste is added every year) and decreases after that (since the waste landfilled earlier generates a smaller amount of CH_4).

Fig. 3 shows LFG collection for two representative LFG collection strategies—active and moderate—as blue and green dashed lines, respectively. The collection efficiency is the ratio between the area under the red curve and the other lines. The results show that the active LFG collection has a higher collection efficiency than the moderate one, because it collects LFG immediately after waste is disposed of in a cell. The moderate collection case starts collecting LFG from an individual cell several years after the first waste is landfilled in the cell. This is why the moderate collection cases show significant intermittent drops in LFG collection because of the non-collected CH_4 emission when a new cell starts accepting waste.

When food waste and yard trimmings are compared, food waste has lower LFG collection efficiency under the same landfill operation, mainly because there is more non-collected CH_4 before collectors are operational. However, slow decomposition does not necessarily mean high CH_4 collection over the landfill's lifetime. At year 66, after 30 y of the post-closure care period, the hypothetical landfill stops collecting LFG. It shows that there are still significant CH_4 emissions coming from decomposition of yard trimmings due to their slow decay speed. In the case of food waste, CH_4 emissions after the post-closure care period are almost negligible.

Table 1 shows estimated lifetime collection efficiencies for four types of waste components under four different climate conditions. It shows that cases with very fast or slow decomposition speed lead to low collection efficiency. For fast decomposition cases such as food waste, wet tropical conditions accelerate decomposition speed, which leads to significant LFG losses before operating collectors. In contrast, wood under dry boreal conditions has the slowest decomposition speed, and delayed emissions that are not collected during the post-closure period reduce the lifetime



a) Food waste b) Yard trimmings

Fig. 3. Methane generation and collection over time from landfilled waste (1 dry t/y for 36 y) from (a) food waste and (b) yard trimmings.

Table 1
Collection efficiencies with respect to types of landfilled waste and climate conditions with active and moderate LFG collection strategies.

LFG collection	Active				Moderate			
Climate condition	Boreal Dry (%)	Tropical Dry (%)	Boreal Wet (%)	Tropical Wet (%)	Boreal Dry (%)	Tropical Dry (%)	Boreal Wet (%)	Tropical Wet (%)
Materials								
Paper	78	79	82	83	70	70	71	71
Wood	68	71	74	76	63	65	67	68
Food	82	84	79	72	71	70	59	46
Yard trimmings	80	83	83	80	71	71	69	61
MSW mixture	79	80	80	78	70	70	66	60

collection efficiency. In general, active LFG collection shows much higher collection efficiency compared to landfills with moderate collection strategies for a given climate and waste condition.

Collected LFG can be either flared or used to generate electricity; both cases generate the same amount of CO_2. However, the electricity generated from LFG combustion can be regarded as displacing regional electricity, which indirectly avoids CO_2 emissions. CO_2 emission credits depend on the power generation mix in the region where the electricity is displaced, because each power generation technology has different CO_2 emissions per unit of power generation. This analysis assumed that electricity generated in landfills leads to a displacement credit of 550 g CO_2e/kWh based on the U.S. power generation mix (Argonne National Laboratory, 2016). It assumed that a micro-turbine with an electric power generation efficiency of 34% was used.

3.3. Landfill gas oxidation

As shown in Fig. 1, non-collected CH_4 goes through landfill covers and is partially oxidized through biological CH_4 oxidation. It is difficult to determine the oxidation factor because only the emissions that result from partial oxidation of CH_4 can be measured while LFG generation is estimated. Although 10% is commonly used for the oxidation factor of managed landfills in the United States (USEPA, 2017), it is questionable whether this value is representative of the oxidation factor in general. The value has been used since it was agreed upon in 1997 (IPCC, 2000), based on three studies conducted in New Hampshire. Czepiel et al. (1996a) studied the factors that influence the CH_4 oxidation in landfill cover, and detailed CH_4 emission measurements were discussed in Czepiel et al. (1996b). Liptay et al. (1998) also determine methane oxidation using stable isotopes and obtained consistent oxidation factor of 10%. Since then, efforts have been made to quantify CH_4 oxidation using various measurement methods. Chanton et al. (2009) reviewed the literature and summarized the CH_4 oxidation factors from 42 determinations. They found that the CH_4 oxidation factor in landfills is higher than 10%. The mean oxidation factor of examined studies was 36% with a standard error of 6%. The average oxidation factor of 36%, collected by Chanton et al. (2009), was used in this analysis.

Actual oxidation may not be proportional to non-collected CH_4; the oxidation factor may not be expressed as a fixed number, but vary based on the amount of CH_4 flux that goes through the cover, type and thickness of landfill covers, soil texture, moisture content, temperature, CH_4 and O_2 concentrations, and nutrients. Various parameters such as cracks in landfill covers and lateral diffusion may lead to significant variation even with similar conditions.

4. Results and discussion

4.1. GHG emissions from landfilled wastes

The GHG emissions from a given amount of landfilled organic wastes depend on the fate of the carbon in the waste: (1) C sequestered in landfills, (2) C in CO_2 from waste decomposition, (3) C in CO_2 from collected and combusted CH_4, (4) C in CO_2 from CH_4 oxidation, and (5) C in CH_4 emitted to the atmosphere. Fig. 4 compares the fate of carbon in four types of landfilled wastes (i.e.,

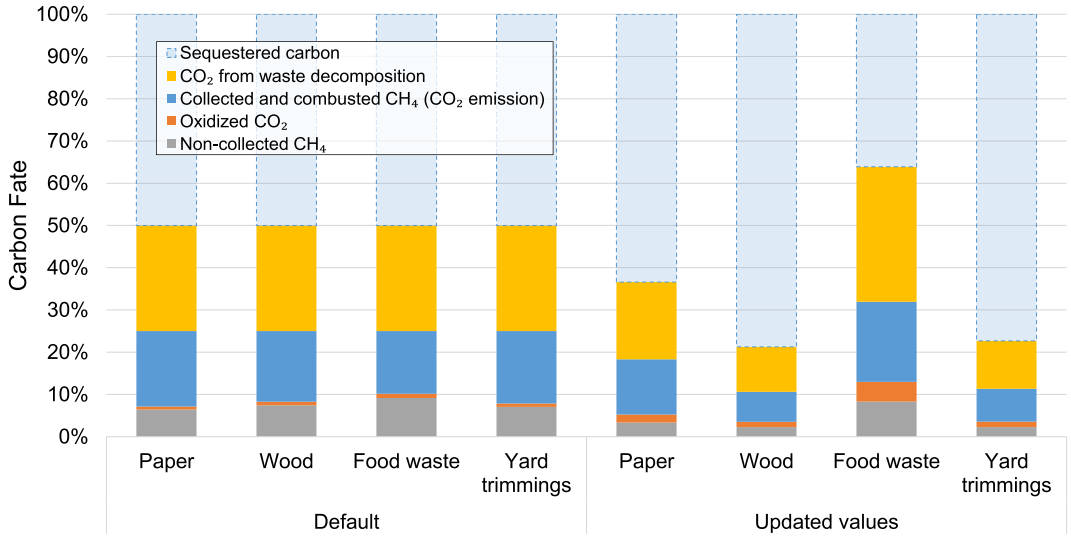


Fig. 4. The fates of carbon from four types of landfilled wastes using default and updated parametric values for fraction of degradable materials to be decomposed and the oxidation factor of CH_4 from landfills.

paper, wood, food waste, and yard trimmings) on a *DOC* basis using two different sets of parameters (i.e., IPCC default and updated values). The default cases in Fig. 4 were simulated using 0.5 and 10% for DOC_F and the oxidation factor, while the cases on the right-hand side used the updated DOC_F in Fig. 2 and the oxidation factor of 36%. Both cases were assumed to be under wet boreal conditions with moderate LFG collection and LFG flaring.

As mentioned earlier, total carbon emissions (the remaining carbon after sequestration) depend directly on *DOC* and DOC_F for the given amount of landfilled wastes. For the default cases, total carbon emissions are proportional to the *DOC* of the waste, since the default DOC_F is fixed at 0.5. The result shows that the IPCC's method using default values does not yield any difference in carbon emissions regardless of the type of waste. There are only categorical differences depending on the amount of oxidized CH_4 through LFG flaring or CH_4 oxidation. A feedstock-specific DOC_F allows us to see the variation in carbon emissions with respect to the waste component. While the bars using updated values have a slightly higher oxidation factor than the default case, the oxidation factor does not influence total carbon emissions since it only determines whether the carbon is released in the form of CO_2 or CH_4. However, this affects GHG emissions significantly because of the difference in GWP between CO_2 and CH_4.

Fig. 5 shows the estimated GHG emissions generated from 1 dry t of four types of landfilled wastes using default and updated parametric values (in kg of CO_2e). It shows that non-collected CH_4 in Fig. 4 (gray bars) becomes a major GHG emission source because of its high GWP. Although carbon emissions through non-collected CH_4 make up only 26%–37% of total carbon emissions, updated values show that CH_4 makes up 52%–62% of GHG emissions in terms of CO_2e. The collection efficiency that mostly determines the CH_4 and CO_2 emission ratio is an important parameter for GHG landfill emissions.

When default values are used, food waste and yard trimmings have similar carbon emissions (0.5 t per dry t of waste) due to their similar *DOC* in dry conditions. However, GHG emissions from food waste are 18% higher than those from yard trimmings because of the difference in the collection efficiencies caused by their differing decay speeds, as shown in Table 1. The collection efficiencies are 71%, 67%, 59%, and 69% for paper, wood, food waste, and yard trimmings, respectively, under wet boreal conditions with a moderate LFG collection strategy.

For updated cases, GHG emissions from landfilled wastes vary significantly by type of wastes due to huge variations in DOC_F. For food waste, the newly estimated GHG emissions are 4% higher than those of the default food waste landfill case, while the other three types of wastes lead to 39%–65% reductions in estimated GHG emissions due to changes in DOC_F and the oxidation factor.

Unlike most other WTE pathways that eventually release carbon into the atmosphere, landfilled wastes keep a high fraction of carbon in the landfill, considering that DOC_F is not 1. For example, when 1 dry t of food waste is used for compressed natural gas (CNG) production, carbon in food waste (0.44 t of C/dry t) would be released either during fuel production processes or during fuel combustion in vehicles, mostly in the form of CO_2 (2078 kg CO_2e/dry t food waste) (Lee et al., 2016). The CNG production pathway would displace fossil CNG and synthetic fertilizer, whose displacement credits were estimated at 598 and 88.6 kg CO_2e/dry t food waste (Lee et al., 2016). In contrast, under the counterfactual scenario, only 64% of carbon (0.32 t) in landfilled food waste would be released over 100 y, and the remaining 36% (0.18 t) would remain in the landfill (Lee et al., 2016). A notable portion of LFG is CH_4 (56 kg CH_4/dry t food waste), whose GWP is 30 times that of CO_2. This non-collected CH_4 from landfills results in substantially higher GHG emissions associated with the counterfactual scenario (2708 kg CO_2e/dry t food waste) than those of the food-waste based CNG (1392 kg CO_2e/dry t food waste).

Using yard trimmings shows the opposite result. Because of their low DOC_F, net GHG emissions from landfilled yard trimmings become 829 kg CO_2e/dry t yard trimmings. This is 35% lower than that of the yard trimmings-based ethanol production pathway (1268 kg CO_2e/dry t yard trimmings), even considering the fuel displacement credit (Lee et al., 2016). Note that net GHG emissions estimated using the IPCC's default values from landfilled yard trimmings is 2197 kg CO_2e/dry t yard trimmings, which is 73% higher than that of ethanol from yard trimmings. This shows the significant influence of DOC_F on WTE LCA results.

As mentioned previously, inorganic carbon in plastics does not decompose through the anaerobic digestion processes, which results in no emissions if it is landfilled. WTE using non-recycled

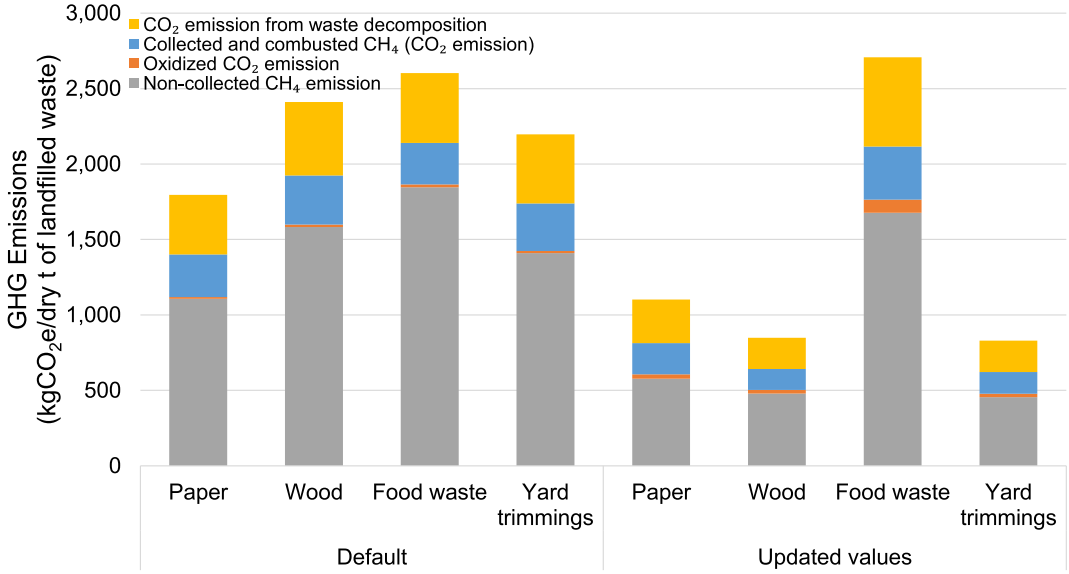


Fig. 5. GHG emissions from landfilled organic wastes (kg CO_2e/dry t of landfilled wastes) for four types of waste materials using default and updated parametric values (with GWPs of CO_2 and CH_4 for a 100-y time horizon).

plastics releases carbon into the atmosphere during fuel production and combustion processes as in other WTE technologies. Benavides et al. (2017) analyzed plastic-to-fuel (PTF) conversion through a pyrolysis process and estimated well-to-wheels GHG emission at 3600 kg CO_2e/dry t plastics. However, due to their high energy conversion efficiency (73%), fossil fuel displacement GHG emission credits of energy products (i.e., ultra-low sulfur diesel, naphtha, char, and fuel gas) were estimated at 3950 kg CO_2e/dry t plastics, which exceeds GHG emissions from fuel production and combustion. Net GHG emission of PTF becomes −350 kg CO_2e/dry t plastics.

Considering the emissions from landfilled wastes in Fig. 5, WTE would be a good alternative, especially for waste feedstocks with high LFG emissions such as food waste. This is not only because the WTE pathways have the benefit of significantly avoiding GHG emissions from landfills, but also because controlling LFG from food waste may demand higher investment and operating costs compared to other waste streams. Fuel conversion efficiency also plays an important role in determining net GHG emission, as shown in the example of the PTF conversion technologies.

4.2. Sensitivity analysis

Sensitivity analysis results presented in Fig. 6 show the impact of each parameter on estimated GHG emissions from landfills. The sensitivity analysis results of food waste are shown, and GHG emissions of other waste feedstocks show similar trends (see supplementary material). The baseline case is LFG from 1 dry t landfilled food waste under wet boreal conditions using updated simulation parameters (i.e., DOC_F values in Fig. 2 and an oxidation factor of 36%), and moderate LFG collection with LFG flaring was assumed. The GHG emissions for the baseline case were estimated at 2708 kg CO_2e/dry t food waste.

The most influential parameter is DOC_F. The estimated GHG emissions using the updated DOC_F are 28% higher than those from using the IPCC's default DOC_F of 0.5; this is because of the changes in estimating total carbon emissions. When the lower and upper DOC_F values in Fig. 2 were used for sensitivity analysis, GHG emission ranges became 1522–3904 kg CO_2e/dry t food waste.

Other examined parameters are related to the collection efficiency and the oxidation factor. These influence the concentration of CH_4 and CO_2 in a given carbon emission condition, which leads to lower impacts on GHG emission compared to DOC_F. When the IPCC 10% oxidation factor is used rather than the updated one, estimated GHG emissions increase by 23% because less CH_4 is converted into CO_2. Changes in climate conditions influence decay speed and lead to changes in collection efficiencies, as shown in Table 1. Wet tropical climate conditions result in faster decay and reduce the collection efficiency by releasing LFG before LFG collectors are installed at each landfill cell, which increases the estimated GHG emissions by 19%. Dry boreal conditions lead to the opposite trend, a 17% reduction in estimated GHG emissions. As examined, CH_4 concentration in LFG is within a very narrow range, and the relative changes in GHG emissions from food waste corresponding to upper 25% and lower 25% CH_4 concentrations are 2% and −7%, respectively.

While all the studied parameters have a high impact on the GHG emissions from landfills, they are unlikely to be controlled. However, landfill operators can actively collect and burn more CH_4 to reduce GHG emissions. Generating electricity through LFG combustion can indirectly reduce GHG emissions by displacing electric power that would otherwise be generated. When LFG is actively collected from the early stage of each cell development, GHG emissions can be reduced by 27% compared to the moderate LFG collection case. GHG emission credits for power generation that result from displacing the U.S. average electricity generation mix reduces GHG emissions by 12%. When both the active LFG collection strategy and power generation are applied, overall reduction in GHG emissions relative to the default case is 44%. When other feedstocks are used, sensitivity analyses show a similar trend (see supplementary material), while emissions from food waste are more influenced by parameters other than DOC_F, compared to other feedstocks, because of food waste's low collection efficiency.

5. Conclusions

Of the GHG emission sources from MSW landfilling, emissions of non-collected CH_4 are the highest contributor of GHG emissions on a CO_2e basis because of its high GWP, even with a small amount of carbon emissions. This study showed that DOC_F and the oxidation factor significantly impact results. For example, updating DOC_F causes the GHG emissions from landfilled food waste to vary from 1522 to 3904 kg CO_2e/dry t food waste, and using the updated oxidation factor (36%) from the default (10%) reduces the GHG emissions by 18%. Climate conditions of the landfills influence collection efficiency, resulting in changes in GHG emissions. However, these parameters are based on conditions that cannot be controlled. In order to reduce GHG emissions from landfills, landfill operators can choose to actively collect and burn CH_4. Generating

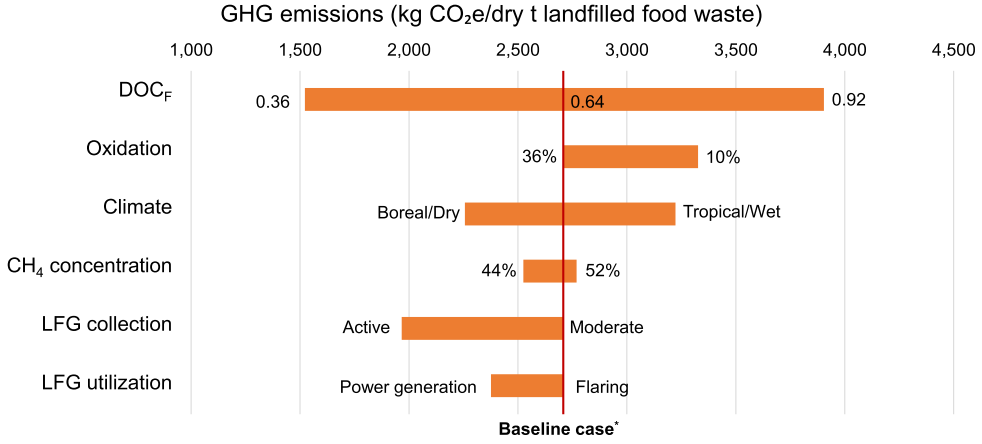


Fig. 6. Sensitivity analysis of GHG emissions (kg CO_2e) from 1 dry t landfilled food waste (baseline case: estimated emissions using updated DOC_F and oxidation values under boreal and wet climate conditions with moderate LFG collection and LFG flaring).

electricity using collected LFG reduces the GHG emissions by displacing electricity that would otherwise be generated with conventional power sources. Using both active LFG collection and power generation, GHG emissions reductions range from 28% for wood to 44% for food waste relative to the case that flares LFG collected through the moderate LFG collection strategy.

The method used and the data collected in this study improve the quality of the GHG estimates of landfills. This study can support transparent decision-making on the sustainable treatment, management, and utilization of MSW for GHG mitigation. It is important to note that each landfill operates differently. The results presented in this study were specific to two landfills this study examined, and should not be taken as representative results for other landfills. To estimate the LFG emissions from a specific landfill, the analysis method of this study can be followed, using the parameters (e.g., DOC_F, CH_4 concentration, oxidation factor, LFG collection, and LFG utilization) estimated for the specific landfill.

In order for WTE technologies to have GHG reduction benefits, they should generate fewer life-cycle emissions compared to current waste management practices. Each type of feedstock under different landfill operating conditions leads to different LFG emissions. This study provides a method that can be used to assess the GHG consequences of various WTE pathways that displace waste landfills and avoid LFG emissions. The method helps identify opportunities for large GHG reductions using WTE pathways. For example, landfilled food waste under poor LFG collection currently generates a lot of LFG emissions, and therefore has a large potential for GHG benefits.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2017.08.016.

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