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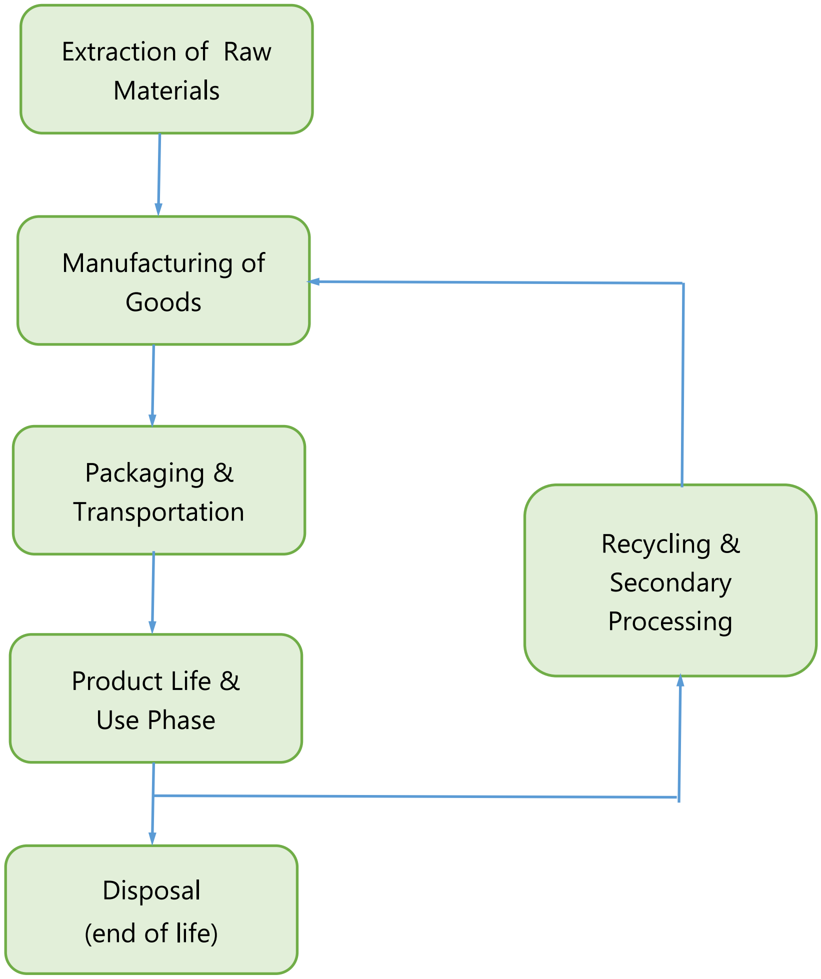

Life-Cycle Assessment (LCA) of Curbside Material Recovery

2022

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EREF's Data & Policy Program

EREF's Data & Policy Program was developed as part of the foundation's effort to expand its mission to advance knowledge and education for sustainable solid waste management. The primary objective of the Data & Policy Program is to aggregate and analyze solid waste data. The program also provides valuable experience to undergraduate and graduate students who assist in data gathering and analysis.

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EXECUTIVE SUMMARY

Life cycle assessment (LCA) can be a valuable tool to evaluate the trade-offs associated with different products and processes. It has been increasingly used to evaluate the environmental impact of solid waste management systems; however, the availability of high-quality studies applicable to the USA are limited. In addition to providing a basic overview of LCA, this report provides information on the greenhouse gas (GHG) emissions and fossil energy use associated with curbside municipal solid waste (MSW) management scenarios including landfilling, recycling, and yard and food waste composting.

Under 4 different collection scenarios, a combination of landfill with gas to energy combined with recycling contributed to the greatest reduction in GHG emissions and energy demand assuming a closed-loop (e.g., bottle to bottle) recycling system. However, the end use of the recyclables are very important and sustainability benefits can be highly variable and even erased under some non-closed loop end use scenarios. For recyclables with marginal or highly variable emissions or energy savings, transport distance to the secondary process can become an important consideration. Emissions and energy savings were found to be different depending on the recyclable being evaluated (e.g. paper, plastic), which suggests that focusing on specific materials that offer the greatest emission reduction when recycled may leader to great overall savings.

Objectives

- Provide an overview of LCA and its application to the waste management industry.
- Compare greenhouse gas emissions and fossil energy use from common municipal solid waste curbside collection strategies including recycling and composting.
- Explore considerations and conditions affecting the sustainability of materials recovery overall, and for specific materials, in North America.

The comparison of these management scenarios is particularly important considering national, state, and local programs commonly use landfill diversion as a metric of sustainable materials management (SMM). A key observation from this study suggests that increased landfill diversion is not directly correlated with lower GHG emissions. Similarly, curbside recycling may not provide emissions or energy savings in all situations. GHG and energy benefits highly depend on the type of material being recovered and the assumption that materials are being reused in a closed-loop remanufacturing process.

GHG impacts associated with waste management activities are influenced by a variety of stakeholders. In scenarios with landfill diversion options (i.e., recycling and composting) the majority (66-70%) of GHG emissions can be attributed to product manufactures and consumer behavior. Of the activities the waste industry can control, landfilling has the largest impact on GHG emissions, and efforts to improve gas capture rates provide the highest GHG benefits. Collection activities are the second largest contributor and the results of this study confirm that companies can continue to reduce emissions by transitioning waste collection vehicle (WCV) fleets to lower-carbon alternatives.

The study results suggest that while composting offers benefits, other endpoints (e.g. backyard composting, anaerobic digestion (AD), bioreactor landfilling) may offer greater reduction in GHG emissions. Organics diversion activities can have other positive local/regional environmental impacts which should be considered in conjunction with any potential reduction in GHG emissions or energy demand. Overall, landfilling, recycling, and composting are important components of the waste management system, but maximizing recovery and closed-loop remanufacturing of materials through recycling has the greatest potential for reducing GHG emissions and energy demand associated with curbside material recovery.

INTRODUCTION & STUDY DESCRIPTION

Recycling efforts in the US can be traced back to early 1900s. Spurred by the environmental movement, linking the concept of reduced environmental burden to recycling evolved in the 1980s as this seemed an obvious way to reduce pollution and natural resource consumption. While the anticipated benefit of recycling (i.e. conserving natural resources) is intuitive, the model of recycling has faced increasing pressure in recent years. Low commodity prices, demand for lower contamination, and poorly developed domestic end markets have created an impetus to take a closer look at when recycling makes sense and how recycling should be done.

In many cases, sustainability goals are based on increasing the recovery of curbside materials for both recycling or composting, and thereby diverting these materials from landfills. These goals are set under the assumption that more materials being processed for recycling equate to the lowest environmental impacts. This assumption may not always be true and, beyond environmental impacts, the business case for material recovery must incorporate the triple bottom line concept of people, planet and profit in order to remain viable long term without government interventions (e.g. subsidies). When material recovery efforts began, tools to better assess sustainability metrics had not been developed. However, the use of life-cycle assessment (LCA) has increasingly been employed to quantify environmental consequences and connect this information to economic metrics. Prior work has suggested that evaluating the sustainability of a particular waste material is far more complex than simply ensuring more tonnage is diverted from landfill, yet little information exists regarding the efficacy of recycling and composting under different operating regimes. Further, the use of LCA has shown that certain manufacturing or discard practices once thought to be sustainable actually may not be the most sustainable option. For example, LCA has been used to demonstrate that multi -layer laminate packaging, even though there is currently no pathway for recycling this material, results in lower environmental impacts for energy use and CO_2 emissions than equivalent recyclable steel or plastic packaging options (Franklin Associates, 2008).

The purpose of this effort is to utilize an LCA to evaluate environmental impacts of residential recycling and composting of specific materials compared to other end points of disposal (e.g. landfill, waste-to-energy, composting). This study contains a number of elements that translate into a more detailed understanding of MSW management and how LCA is used to better understand the environmental impacts associated with recycling and other discard options.

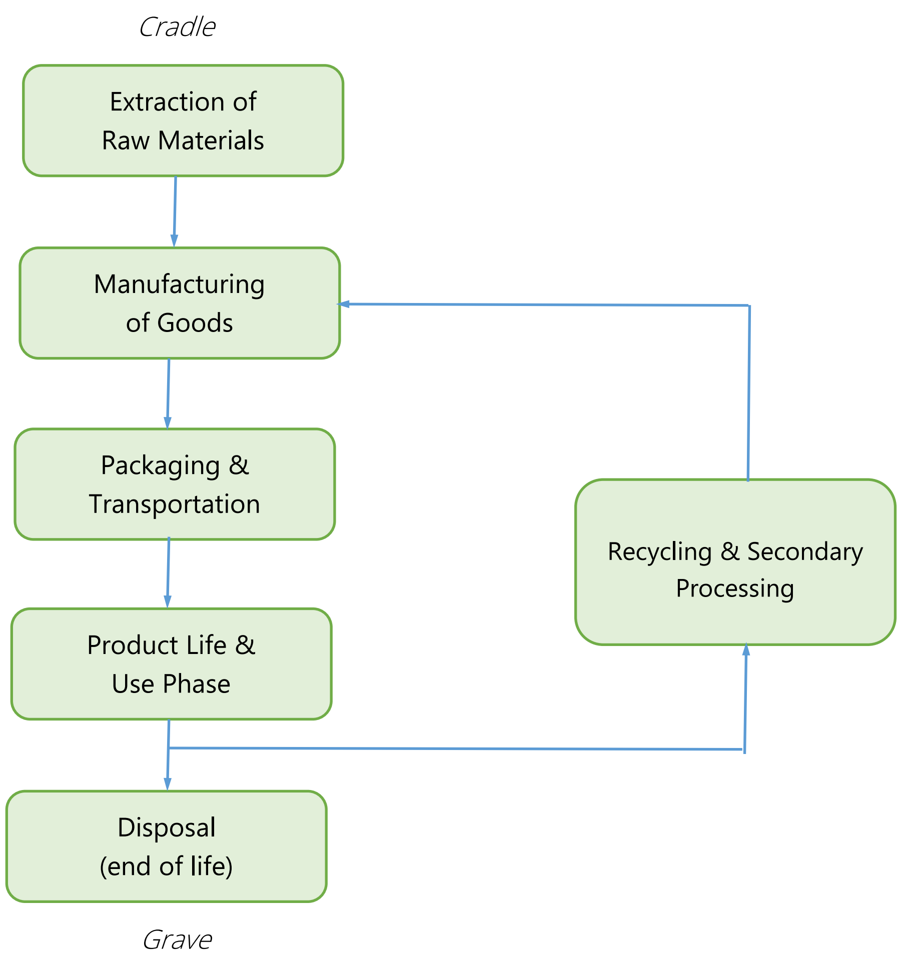
The objectives of this report are to:

1) Provide a general overview of LCA to ensure the reader has a basic understanding of principles of LCA and how it is applied.
2) Present results of an LCA that compares greenhouse gas emissions and fossil energy use from common municipal solid waste curbside collection strategies including recycling and composting.
3) Explore the considerations and conditions affecting the sustainability of material recovery overall, and for specific materials, in North America.

WHY IS LCA IMPORTANT?

What is Life-Cycle Assessment (LCA)? LCA is a technique that evaluates the impact of a product, process, or decision on the environment. LCA can be considered similar to a cost analysis, but focuses on quantifying environmental rather than financial impacts. In LCA, environmental impact is considered for processes throughout a material's life cycle from cradle to grave, including: material acquisition, processing, manufacture, distribution, use and final disposition (e.g. disposal, recycling, re-use) as shown in Figure 1.

Figure 1. Typical process categories for primary and secondary production.



In its simplest form, LCA can be used to quantify and sum the impacts of a single material or product. For example, LCA could be used to sum the environmental impacts of a shirt by quantifying the impacts in each stage of the lifecycle. Examples of potential impacts from each lifecycle stage include:

1) Extraction of Raw Materials—the impacts of growing cotton and producing cloth for the shirt, production of synthetic material for the shirt (e.g. plastic buttons)
2) Manufacturing of Goods—the energy and emissions resulting from manufacturing the shirt

3) Packaging and Transportation—the impacts of the packaging materials and resources (e.g. fuel) used to transport the shirt to the consumer
4) Product life & use phase—the energy and resources used during the shirts useful life (e.g. electricity and hot water to wash the shirt)
5) Recycling & Secondary Processing—impacts associated with collection and reprocessing (if the shirt is source-separated for textile recycling)
6) Disposal—impacts associated with collection and disposal of the shirt, including any emissions produced by the shirt in a landfill (if the shirt is not source-separated for textile recycling)

LCA can also be used to compare alternatives to understand the environmental impacts of various decisions or choices (e.g. less pollution, reduced energy demand). Using the shirt example, consequential LCA could be used in decision-making such as:

- How much energy is saved if the shirt is washed in cold water rather than hot water?
- Does manufacturing the shirt from organic cotton rather than conventional cotton have higher or lower environmental impact?
- Does recovering the shirt for textile recycling increase or decrease environmental impact?

When comparing alternatives, one set of processing steps (referred to as a process flow) is compared versus a 'base case'. Common choices for base case processes are those representing a typical, historical, or worst case scenario. For recycling, in many instances the base case used are processes by which a material or product is made from virgin, non-recycled materials. This process flow is generally referred to as <u>primary production</u>. The recycling process flow, generally referred to as <u>secondary production</u>, would be compared to primary production to determine if there is a reduction in environmental impact from using recycled rather than virgin materials.

It should be noted that the processes shown in Figure 1 are general process categories, and that the specifics of each process may vary based on the type of material. For instance, the extraction of raw materials would include mining silicon for the manufacture of glass but for paper would include the logging of trees. As a result, the environmental impacts for each process step will vary depending on the material. Similarly, the end-of-life management (e.g. collection and disposal) of a particular material represents only a portion of the overall life cycle and the associated impacts or savings will vary in degree depending on the properties of the material (e.g. if the material degrades anaerobically in a landfill). Therefore, it is important to identify which process(es) in a material's life-cycle have the highest impacts and/or offsets when forming strategies for mitigating these impacts since minimizing changes during one process (e.g. end-of-life) could significantly, or minimally, impact the overall environmental assessment of the materials life-cycle.

How LCA Quantifies Environmental Impacts.

Examining and comparing the environmental impacts of individual processes within a material's life cycle requires the ability to quantify impacts. To do this, environmental impact categories such as "global warming" or impacts such as "acid rain" must be represented in measurable terms. For example, greenhouse gas (GHG) emissions are an accepted measure of global warming impacts and are quantified in units of CO_2-equalivent (CO_2-e) emissions. Example impact categories and the accepted metrics for measurement are included in Table 1.

Table 1. Example metrics used to quantify environmental impacts of LCAs.

Impact Category	LCA Metric	End-Point Impact
Global Warming	CO_2 equivalents	Temperature increase/Climate Change
Acidification	H^+ mole equivalents	Acid Rain
Eutrophication	N equivalents	Ecosystem loss
Respiratory Health	Particulate Matter ($PM_{2.5}$)	Human health impacts
Ozone Depletion	Chlorofluorocarbon (CFC-11) equivalents	Human health impacts
Smog	Ozone (O_3) equivalents	Human health impacts
Cancer	Chloroethylene equivalents	Human health impacts
Water Consumption	Water Use	Water Shortage and Drought
Energy Consumption	Energy Use	Varies, depends on energy source

Understanding and Interpreting LCA Results.

Situations where a process or system results in a reduced overall impact can be referred to as an environmental savings, environmental benefit, or burden avoidance. For example, assume primary (e.g. virgin) production of a material results in 20 lbs of carbon dioxide (CO_2) emissions per ton of material and secondary production (e.g. recycling) results in 5 lbs of CO_2 emissions per ton of material. Emissions associated with secondary production could be expressed in a number of different ways, such as:

- the value itself (i.e. "5 lb CO_2/ton gross emissions"),
- as the net emissions avoided (i.e. "15 lbs of avoided CO_2/ton"), or
- as the reduction in emissions (i.e. a 75% reduction in emissions).

In most situations, net emissions are presented as a negative value when making a comparison like the one above (i.e. "-15 lbs CO_2/ton"). Therefore, it is common to see negative values result from an LCA analysis in situations where recycling/manufacturing offsets exist (such as in the recycling example above), or carbon-intensive energy sources like coal are replaced by process-generated energy (e.g. landfill gas-to-energy, waste to energy, or anaerobic digestion) creating an energy emissions off-set or credit.

Use of LCA to Understand MSW Management Systems. While in many ways LCA is still maturing as a tool in terms of its accuracy and use, the technique has significant potential to guide thinking on discards management, circular economy and overall sustainability strategies. For the purposes of this report, the application and discussion of LCA will be specific to its application in waste management. LCA can be used to answer questions related to operational practices during materials recycling, such as:

- What are the most environmentally important processes in the manufacture and discard of a particular material?
- Is using a recycled material always better than using a virgin material?
- Are there certain scenarios where it is better to not divert a material?
- How does transport distance from a recycling facility to re-manufacturer affect global warming potential?

LCA has been increasingly used in countries globally to answers such questions. In 2014, researchers at the Technical University of Denmark published an international review of LCA studies to summarize state of LCA and lessons learned (Laurent et al., 2014). The study identified 222 LCAs that examined the management of one or more solid waste materials (e.g. MSW, C&D, commercial food waste, glass, tires). Of those, Laurent et al. (2014) found that only 39% (87 studies) were of sufficient quality and detail to be deemed reasonably accurate.

Findings from Laurent et al. (2014) showed an increasing trend in the use of LCAs to understand environmental impact of the solid waste management system, with a large spike in the number of published peer-reviewed case studies beginning in 2009. The review asserts there was not definitive agreement regarding the most environmentally favorable endpoint of disposal. This lack of agreement reflects, at least in part, the ability of LCA to be tailored to system- and locale-specific data, and that differences in various factors substantially impact which endpoint is more or less favorable, which include: waste composition, waste collection and management technology, energy grids (e.g. nuclear-based in France versus coal -based in Poland), transportation distances, and end markets. Based on the findings from Laurent et al., key factors when considering LCA studies for use in decision-making include:

- the quality of the study (e.g. how well its design and analysis adheres to best practices),
- what elements are included in the system being analyzed
- the representativeness of the life-cycle data relative to the modeled system or locale, such as whether the study:
  - – examines a waste composition similar to the waste stream of interest
  - – technology/system configuration is similar to the system of interest
  - – uses best-available, current average, or site-specific facility data
  - – considers processes or material end-points not available in the area of interest
  - – assumes an energy grid mix similar to the area of interest, impacting energy offsets

While Laurent et al. highlights the dependence of LCA results on geographical properties and the elements of the waste management system, it also shows the relative lack of USA-based analyses. Of the 222 studies identified by Laurent et al. (2014), only 8 studies (3.6%) modeled MSW management in the United States. Since the Laurent et al. paper was published, 22 additional published LCAs were identified by EREF with relevant information for the U.S. The resulting body of LCA knowledge applied to the U.S. is primarily academic papers (70%), followed by trade or industry publications (e.g. Glass Packaging Institute study on glass recycling), and governmental sources (i.e. supporting documentation from US EPA's Waste Reduction Model, or WARM). The majority of studies examined the full life-cycle from cradle-to-grave or cradle-to-cradle. Others focused specifically on manufacturing, collection or end-of-life exclusively. Previous U.S. studies commonly used a mixed-waste MRF (MWMRF) instead of, or in addition to, single- or dual-stream MRFs and relied on drop-off as well as curbside collection of recyclables (Kaplan et al., 2009; Levis et al., 2014; Cabaraban et al. 2008; Morris, 2005). As such, the LCA modeling work performed as part of this effort provides unique and valuable insight by examining material recovery scenarios that represent more common MSW collection and management practices in the U.S.

Material Recovery Options and the MSW Management System. Three curbside MSW recovery options were examined in this assessment: single-stream recycling, yard waste composting, and mixed organics composting (i.e. food and yard waste). Scenarios were constructed by progressively adding services to a landfill-only MSW management program. First, curbside single-stream recycling was added, creating a 2 bin curbside program comprised of landfilling and recycling. Next, yard-waste only composting was added, creating a 3 bin curbside program. Finally, food waste was added to the yard waste composting program, resulting in a 3 bin program for mixed organics, recycling, and landfilling. These scenarios are described in Table 2.

Table 2. MSW management scenarios and collection frequencies.

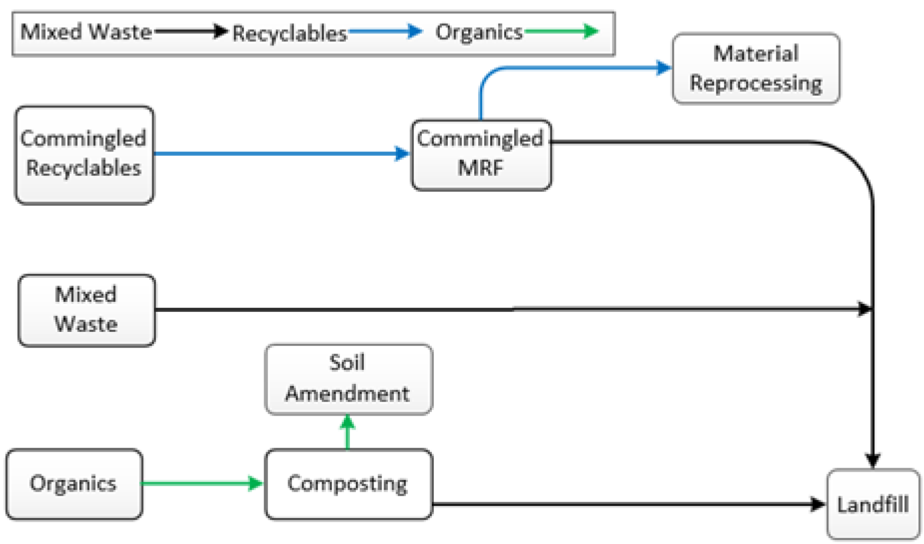
MSW Management Scenario	Number of Bins	Weekly Collection Frequency		
		Landfill	Recycling	Composting
Landfill (with LFGTE) *LF*	1	2x	-	-
Landfill + Recycling *LF + R*	2	1x	1x	-
Landfill + Recycling + Yard Waste *LF + R + YW*	3	1x	1x	1x *(Yard Waste)*
Landfill + Recycling + Yard/Food Waste *LF + R + YW/FW*	3	1x	1x	1x *(Yard & Food Waste)*

Constructing the LCA to examine the MSW management system as a whole, rather than recycling or composting individually, reflects the integrated nature of solid waste management and allows for systemwide impacts to be assessed. This approach also results in comparisons that reflect the decisions facing municipalities when considering expanding MSW recovery programs. Curbside residential programs are tasked with managing a variety of materials, and material recovery does not happen in a vacuum. Facilities associated with material recovery generate discards typically destined for landfill (e.g. contamination, residuals) which much be accounted for in addition to the materials generated by the residents but not recovered (e.g. recyclables placed into the landfill/garbage bin by residents). Additionally, changes to one facet of the program (e.g. addition of food waste to a composting program, removal of materials from a recycling program due to market constraints) will affect the amount and type of materials managed in other facets of the system. These changes can have implications (e.g. changes in landfill gas generation, refuse collection vehicles filling at a different rates during a route), which can be captured in a systemwide analysis.

To represent the integrated nature of MSW management, the analysis conducted as part of this study included the following key attributes (Figure 2):

- curbside collection of the generated MSW, and transport to processing facilities
- processing of recoverable material at a comingled (i.e. single-stream) material recovery facility (MRF) or composting facility
- transportation of material reprocessing, and the associated end use (i.e. bottle-to-bottle, recycling, composting, product used as soil amendment)
- disposal of non-received materials, both generated curbside or as residues at MRF/composting facilities, at a landfill with landfill gas-to-energy (LFGTE) beneficial use.

Figure 2. MSW management system and process flows used in this assessment.



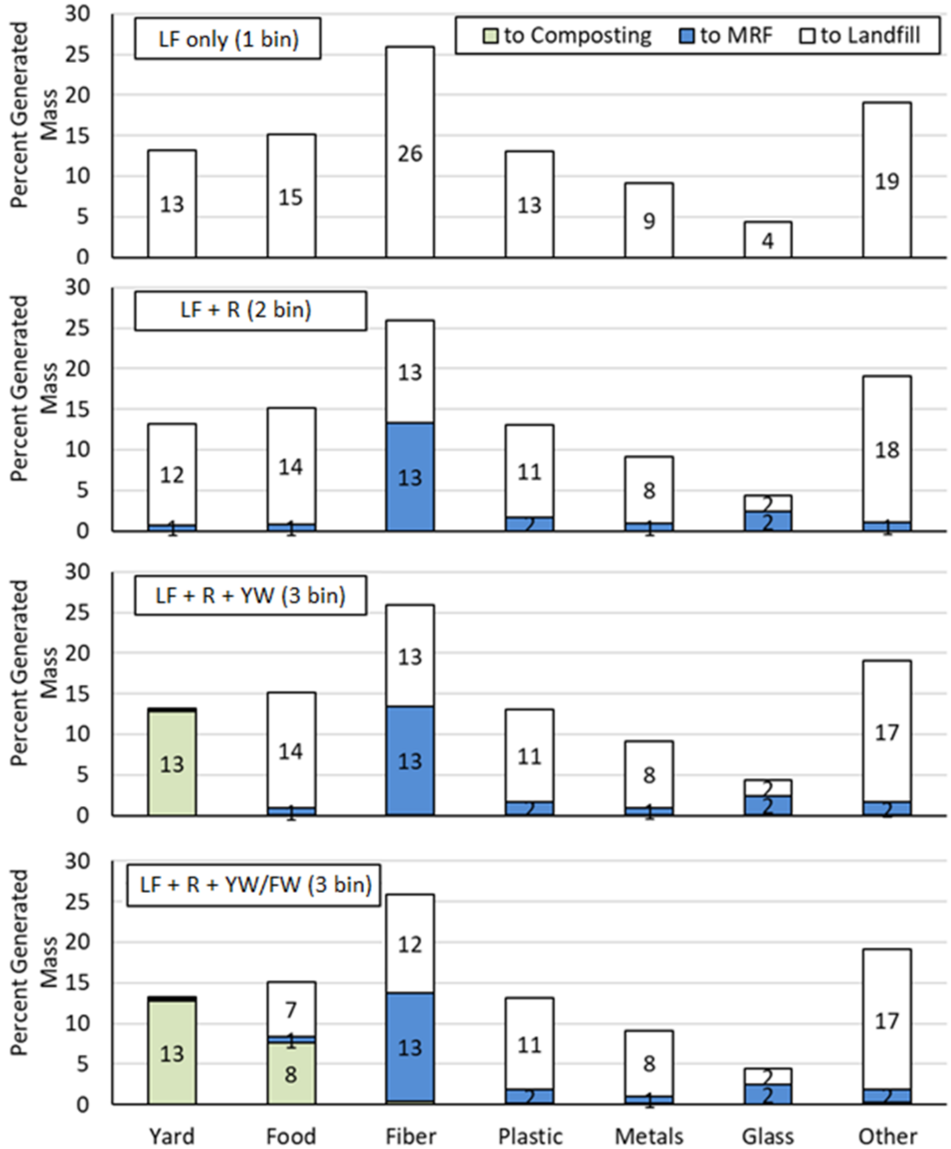
LCA Modeling Approach and Assumptions. To assess the environmental impacts of curbside MSW recovery throughout the MSW management system (Figure 2), LCA modeling was performed using the Solid Waste Optimization Life-cycle Framework (SWOLF), developed by North Carolina State University. The SWOLF LCA framework consists of state-of-the-art life-cycle process models for solid waste collection, recycling, landfilling, composting, anaerobic digestion (AD), waste-to-energy (WTE), and gasification that uniquely facilitate integrated analyses of material recovery strategies. Additionally, factors such as waste composition or contamination rate can be specified within the model, which provides additional functionality compared to more popular, but simplified, tools such as WARM. SWOLF has been used to conduct some of the most recent LCA research (International Institute for Solid Waste Management Life-Cycle Modeling, 2019).

Details on the assumptions, default values, and SWOLF process models are presented in Appendix A. Key assumptions made as part of this analysis include:

- MSW composition based on U.S. EPA 2014 Facts and Figures estimates (Appendix A; Table A1)
- Collection fleet fuel use is 80% diesel and 20% CNG
- Only the most commonly accepted materials were included for recycling:
    – Paper: old corrugated cardboard (OCC) and mixed paper
    – Metals: ferrous containers and aluminum containers
    – Plastics: HDPE containers, PET containers, PP containers
    – Glass: glass containers
- Material-specific capture rates and MRF separation efficiencies (Appendix A; Table A2)
- Contamination rates for MSW recovery pathways:
    – Recycling MRF contamination = 18%
    – Composting contamination = 4%
- Baled recyclables were assumed to go to closed-loop or best case remanufacturing end use.
- Landfill gas-to-energy (LFGTE) system installed at the landfill. The impact of this assumption on the final results is described in the section *Impact of Key Assumptions on LCA Results*, with additional detail in Appendix C.
- Carbon accounting includes long-term biogenic carbon storage (e.g. in the landfill), consistent with accepted practice. A more detailed description of carbon accounting is presented in Appendix B. The impact of this assumption on the final results is evaluated in the section *Impact of Key Assumptions on LCA Results*.

The amount and types of materials collected curbside were based on U.S. EPA estimates of MSW composition, with fiber (all types), yard waste, plastics (all resins), and food waste comprising the largest fractions (Figure 3). For each material recovery scenario, the amount discarded and end of life destination (i.e. landfill, single-stream MRF, or composting facility) of collected material was determined based on material-specific capture rates (Appendix A, Tables 1A and 1B respectively). This includes materials incorrectly sent to recycling (e.g. a small portion of yard and food waste), and recyclable materials residents fail to sort into the recycling bin and are sent directly to the landfill for disposal. The breakdown of waste composition and destination after curbside collection are depicted for all MSW management scenarios in Figure 3.

Figure 3. Waste composition and material destination after curbside collection for each scenario.



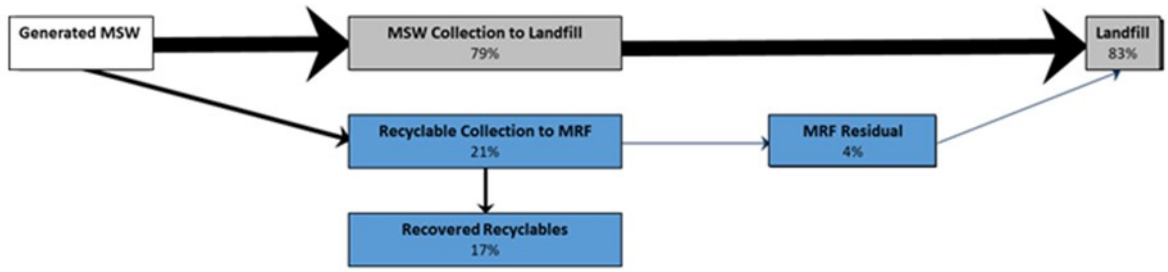
With the addition of yard waste composting (LF + R + YW, 3 bin scenario), 13% of the generated MSW is collected curbside for composting and, as with the previous scenario, 21% of generated MSW is transported to a MRF for processing. Once residuals from the MRF and composting facilities are accounted for, a total of 70% of generated MSW is ultimately disposed of in the landfill and the remaining 30% is recovered for reprocessing or soil amendment (Figure 4B).

When food waste is added to the curbside composting program (LF + R + YW/FW, 3 bin scenario), the fraction of waste collected for composting increases from 13% to 21%. As with the previous scenarios, 21% of generated MSW is transported to a MRF for processing, resulting in 42% of MSW going to recycling or composting facilities for processing. The remaining 58% of generated MSW is transported to landfill. Once residuals from the MRF and composting facilities are accounted for, a total of 62% of generated MSW is ultimately disposed of in the landfill (Figure 4C). This scenario results in the highest diversion rate, with 38% of MSW going to end users (i.e. remanufacturing facilities or use as soil amendment).

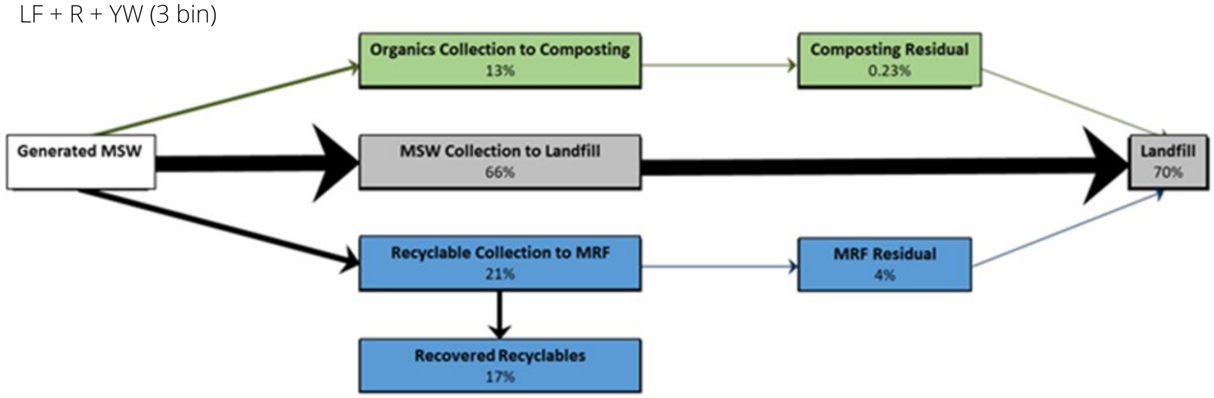
For each of these scenarios, GHG emissions and fossil energy use were examined through LCA using SWOLF. GHG and fossil energy use were chosen as they are common environmental burdens and are applicable globally. Further, GHG emissions, measured as carbon dioxide equivalents (CO_2e), are the generally accepted metric for measuring potential climate impacts (Table 1). It is worth noting that these are only 2 of the possible environmental burdens that could be considered in an LCA model, and other local/regional impacts may also be important to consider in waste management decision-making. For example, water use may be of importance in drought-prone areas of the world and nutrient loading/eutrophication may be of importance in areas near sensitive or protected waters. As a result, the results presented here should be considered thoughtfully.

Figure 4. Mass flow diagrams for material recovery (2 bin and 3 bin) scenarios.

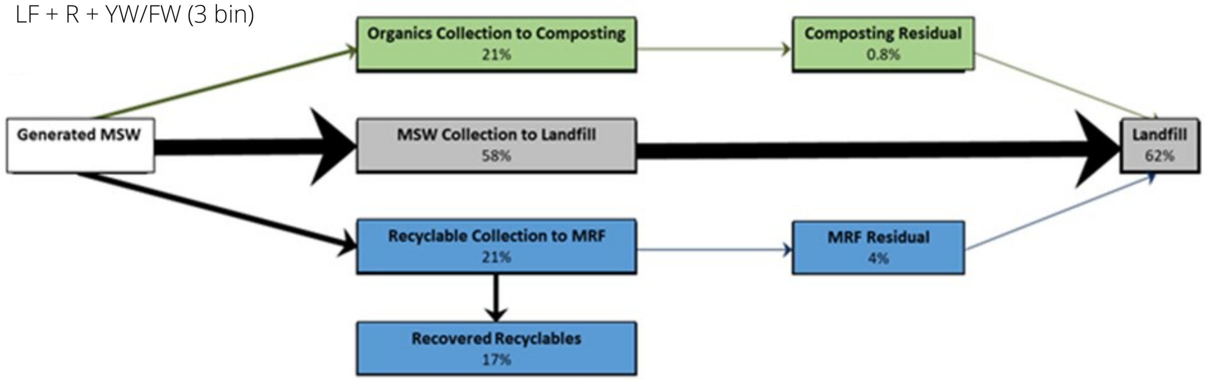
A. LF + R (2 bin)



B. LF + R + YW (3 bin)



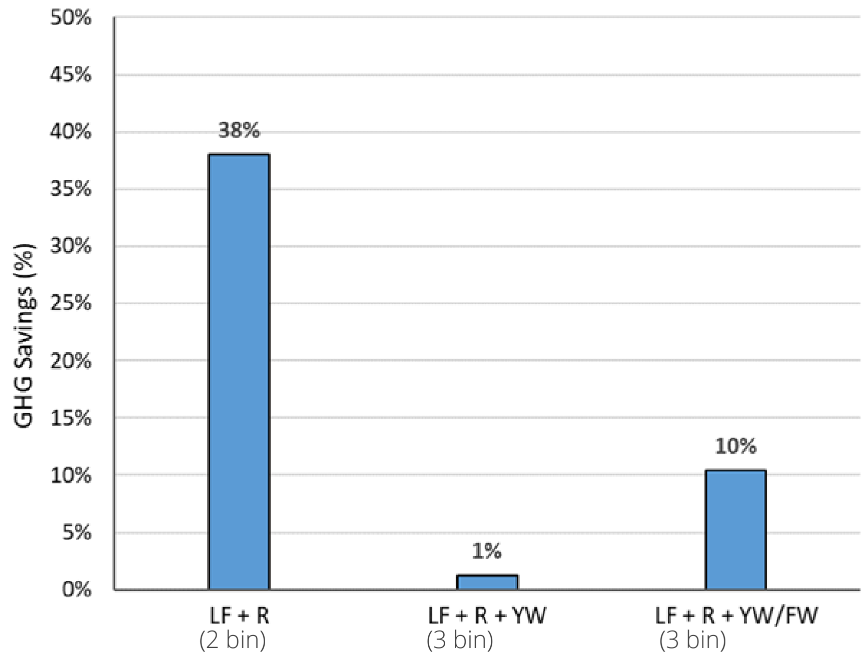
C. LF + R + YW/FW (3 bin)



SWOLF Results for GHG Emissions and Energy Use.

GHG emissions and fossil energy use were estimated for each MSW management scenario and compared to landfill-only MSW management (LF). Results for GHG reduction for each of the material recovery programs are presented in Figure 5, with the LF + R (2 bin) program achieving the highest GHG savings.

Figure 5. Greenhouse gas (GHG) savings for each material recovery program, compared to the landfill (LF) only management scenario[a].



[a] Assumes landfill equipped with gas-to-energy and 1 bin collected twice per week (Table 1).

Adding curbside residential recycling (i.e. LF + R, 2 bin) resulted in a 38% reduction in GHG emissions. The addition of yard waste composting (i.e. LF + R + YW, 3 bin) resulted in GHG emissions roughly equivalent to that of the landfill-only scenario, meaning that the CO_2e emissions savings from implementing curbside recycling are nearly completely negated by the addition of yard waste composting. The addition of food scraps to the composting program (i.e. LF + R + YW/FW, 3 bin scenario) resulted in a 10% GHG emissions savings compared to the landfill-only scenario. A detailed discussion of these results and their interpretation relative to the findings of other studies are discussed subsequently in the following section.

In terms of fossil energy use, all scenarios resulted in energy savings compared to just landfilling the MSW (1 bin scenario). The LF + R (2 bin) scenario resulted in the highest energy savings of 3,787 MJ/metric ton of MSW (1,052 kWh/ metric ton MSW) with the LF + R + YW and LF + R + YW/FW scenarios providing energy savings of 3,600 MJ/metric ton MSW (1,000 kWh/metric ton MSW) and 3,490 MJ/metric ton MSW (969 kWh/metric ton MSW), respectively. The primary reason all scenarios saved energy relates to using recovered recyclable materials in remanufacturing relative to processing virgin materials. Thus, the primary means of energy savings when diversion from landfill scenarios are considered is from recycling. Yard waste or mixed organics composting results in some additional fossil energy use due to activities such as curbside collection and pile aeration; as a result, energy savings are slightly reduced for these scenarios.

Contribution of MSW Management Processes to Overall Impacts. Detailed results for the GHG emissions or savings from individual MSW management activities (i.e. collection, landfilling, recycling, composting, and transportation) and some of the key processes within these management activities (e.g. LFGTE offsets for energy production at the landfill) are presented in Table 3 for each MSW management scenario. Many MSW management activities result in direct emissions as well as offsets or savings, and can result in net negative emissions. For example, emissions from landfilling are offset by credits from landfill gas-to-energy and carbon storage; and, while the magnitude of these emissions and offsets vary based on the types and amount of wastes landfilled, the net landfilling emissions are negative in all 4 scenarios. Similarly, recycling results in emissions at the MRF (e.g. to run equipment) and during remanufacturing of recovered materials; however, emissions savings occur from using recycled rather than virgin inputs to manufacturing and result in net recycling emissions of -123 kg CO_2e/metric ton MSW generated for each scenario that includes recycling (Table 3).

Table 3. Relative contribution of key processes to GHG emissions for each MSW management scenario. ***(Note: Totals may not reconcile exactly with sub-categories due to rounding).***

	GHG Emissions (kg CO_2e/metric ton of MSW managed)			
	LF (1 bin)	LF + R (2 bin)	LF + R + YW (3 bin)	LF + R + YW/FW (3 bin)
Collection	33	33	47	50
Landfilling	-196	-145	-89	-108
LFGTE Offsets	*-44*	*-34*	*-30*	*-24*
Carbon Storage	*-472*	*-341*	*-258*	*-246*
Landfill Operations	*320*	*230*	*200*	*163*
Recycling	0	-123	-123	-123
MRF Emissions	*0*	*1*	*1*	*1*
Remanufacturing Emissions	*0*	*121*	*121*	*121*
Material End Use Offsets	*0*	*-244*	*-244*	*-244*
Composting	0	0	-10	-9
Methane Emissions	*0*	*0*	*9*	*12*
Carbon Storage	*0*	*0*	*-14*	*-19*
Total Emissions	*0*	*0*	*-2*	*1*
Total Offsets	*0*	*0*	*-9*	*-11*
Transport (to End Markets)	0	9	9	10
TOTAL	**-163**	**-226**	**-166**	**-180**

While the landfill scenario results in negative GHG emissions due to credits associated with carbon storage and beneficial use of landfill gas, results show the addition of recycling results in further GHG emissions reductions (-225 kg CO_2e/metric ton MSW for LF + R compared to -163 kg CO_2e/metric ton MSW for LF only). The GHG benefit of adding curbside recycling is primarily due to the large offsets associated with using recycled materials rather than virgin inputs during manufacturing. The addition of yard waste composting (LF + R + YW, 3 bin scenario) results in a 27% increase in GHG emissions, which negates the recycling benefits such that emissions are nearly equivalent to the LF only (1 bin) scenario (Table 3). In contrast, the addition of food waste to the composting program results in a decrease in overall GHG emissions (Table 3).

The opposite performance of YW and FW composting from a GHG emissions standpoint demonstrates that material-specific properties can significantly impact GHG emissions. In this case, the important properties are how fully and how quickly each material degrades in a landfill. When organic material enters a landfill, it degrades under anaerobic conditions to produce landfill gas (meaning carbon in the waste is converted to CO_2 and CH_4). Portions of the waste will not degrade, resulting in some carbon remaining in the landfill for long periods (i.e. carbon storage). Yard waste materials (i.e. leaves, branches) do not degrade as fully as food waste, resulting in more carbon storage in a landfill for yard waste compared to food waste. In addition to how fully a material degrades, it is also important how quickly the degradation occurs. If materials degrade more slowly (e.g. branches), this allows time for the majority of degradation to occur while the landfill gas collection system is installed. Diverting these materials from landfill would reduce the amount of landfill gas collected for beneficial use (if LFGTE is installed). Conversely, if materials degrade quickly (e.g. food waste), the landfill gas collection system will not have been installed in time to capture (and beneficially use) a large portion of the landfill gas. Diverting these materials from landfill would reduce fugitive CH_4 emissions. The impact of landfill gas collection on GHG emissions and degradation rates for specific materials have been well documented in multiple peer-reviewed studies (Eleazer et al., 1997; Barlaz et al., 1998; De la Cruz & Barlaz, 2010). Details on the material properties and carbon balance calculations used in the LCA are provided in Appendix D.

Collection emissions are approximately equal for the LF (1 bin) and LF + R (2 bin) scenarios at 33 kg CO_2e/metric ton MSW since the 2 collection events occur per week in both scenarios: 2x per week garbage collection in the LF (1 bin) scenario and 1x per week garbage and 1x per week recycling in the LF + R (2 bin) scenario (Table 2). The addition of curbside composting requires the addition of another collection event: 1x per week each for garbage, recycling, and composting (Table 2). This additional collection event results in an increase in collection emissions, with the mixed organics (LF + R + YW/FW) scenario having slightly higher collections emissions than the yard waste composting.

Relative contributions to energy use from each process for each scenario are presented in Table 4. Results show that recycling drives the energy savings associated with all material recovery scenarios due to remanufacturing. By comparison, landfilling and composting energy savings are an order of magnitude smaller compared to energy savings from recycling activities.

Table 4. Relative contribution of key processes to fossil energy use for each MSW management scenario.

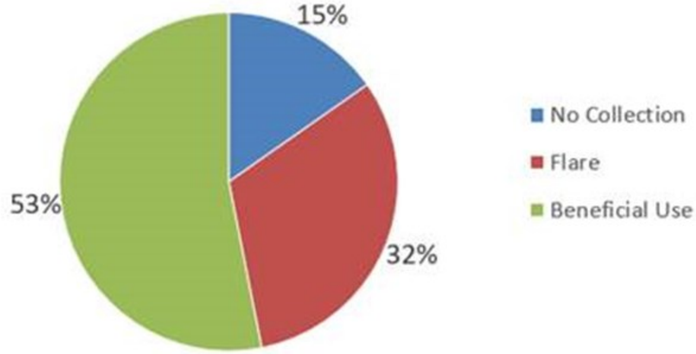
	Fossil Energy Use (MJ/metric ton of MSW managed)			
	LF (1 bin)	**LF + R (2 bin)**	**LF + R + YW (3 bin)**	**LF + R + YW/FW (3 bin)**
Collection	431	429	609	649
Landfilling	-396	-293	-272	-204
Facility Operations	*135*	*112*	*95*	*84*
LFGTE Production	-532	-405	-367	-288
Recycling	0	-4,045	-4,045	-4,045
MRF Energy Use	*0*	*15*	*15*	*15*
Reprocessing Energy	*0*	*1,842*	*1,842*	*1,842*
Remanufacturing Energy Savings	*0*	*-5,902*	*-5,902*	*-5,902*
Composting	0	0	-14	-13
Operations (e.g. aeration)	*0*	*0*	*14*	*23*
Total Offsets	*0*	*0*	*-29*	*-35*
Transport (to End Markets)	0	121	122	122
TOTAL	**34**	**-3,787**	**-3,600**	**-3,490**

For both energy use and GHG emissions, recycling-derived remanufacturing credits are key to realizing environmental benefits, indicating that it is essential to ensure that materials are not only collected for remanufacturing, but that processes and end markets exist to ensure recovered materials actually become inputs for remanufacturing.

Impact of Key Assumptions on LCA Results. LCAs are dependent on the various assumptions and choices made regarding the waste management system (Appendix A), as well as the analytical approach used (Appendix B). It is important to understand how these assumptions may impact the results of the analysis. In solid waste management LCAs, choices and assumptions regarding landfill gas management, CO_2 accounting, and methane global warming potential (GWP) may change the GHG performance and ranking of the program options. To understand how such considerations impact the rankings of program scenarios in this analysis, results were also calculated using alternative assumptions.

In previous work on landfill gas management practices and tonnage data, EREF found roughly 53% of MSW landfilled in 2013 was buried at landfills that beneficially used landfill gas for energy (e.g. heat, electricity), indicating this is the most common management technique for landfilled MSW in the U.S. (Figure 6).

Figure 6. Landfill gas management practices for MSW accepted in 2013, on a tonnage basis.



Currently, the majority of tonnage to landfills goes to land fills where the gad is used beneficially for heat or energy (Figure 6). Landfilling with LFGTE is considered the best case landfilling scenario since this practice reduces on-site carbon emissions compared to simply flaring landfill gas or passive venting. Additionally, energy produced from LFGTE can replace more carbon-intensive energy production from fossil energy sources elsewhere (e.g. coal, natural gas, diesel), resulting in potential emissions credits. To understand how much the assumption of LFGTE impacts the relative performance of landfilling compared to material recovery, an alternative scenario where LFG is managed using a flare was assumed.

The primary LCA results include a carbon storage credit for biogenic carbon that is stored, or sequestered, for long periods (e.g. 100 years) such as during landfill containment. This credit is based on the assumption that biogenic carbon in waste is part of the short-term carbon cycle. This means that emitting biogenic CO_2 has no climate impact, while storing biogenic carbon over long periods instead of returning it to the atmosphere provides a climate benefit. This assumption is the accepted approach consistent with most solid waste management LCAs and GHG reporting (Gentil et al., 2009). To understand the impact of this assumption, an alternative scenario without biogenic storage credits was considered. This alternative method of carbon accounting treats biogenic carbon as already stored in the waste material. Consequently, keeping the carbon stored over long periods (e.g. in landfill) does not provide climate benefit, since it was already stored as the waste material. Further, any biogenic CO_2 emissions are treated as a release of CO_2 since the carbon has gone from being stored in the waste material to being released into the atmosphere.

Many LCAs use a methane global warming potential (GWP) of 25 kg CO_2e/kg CH_4, derived from the Intergovernmental Panel on Climate Change's (IPCC) Fourth Assessment Report (2007). This value was used in the primary LCA analysis because it is currently the recommended value for use in national inventories for consistency with past analyses. However, the IPCC Fifth Assessment Report (2013) recommends a higher GWP value of 28 to 36 kg CO_2/kg CH_4 depending on the assumptions related to climate carbon feedbacks and the oxidation of CH_4 to CO_2. As a result, a GWP value of 34 kg CO_2/kg CH_4 was considered as an alternative.

Rankings for GHG emissions from the primary LCA results and each alternative are presented in Table 5. When emissions were within 5% for two program types, they were considered to be tied. Results show that assumptions regarding LFG management, carbon accounting, and methane GWP had minimal impacts to LCA outcomes.

Table 5. Primary LCA result rankings and alternative assumptions (based on GHG emissions).[a]

MSW Program Type	Primary LCA Results[b]	Alternative Analyses			
		LFG Flare[c]	No Biogenic C Storage Credit[d]	CH_4 GWP 34 kg CO_2/kg CH_4	Average
Landfill with LFGTE *LF (1 bin)*	3 (Tie)	4	2 (Tie)	4	3 (Tie)
Landfill + Recycling *LF + R (2 bin)*	1	1	1	1	1
Landfill + Recycling + Yard Waste *LF + R + YW (3 bin)*	3 (Tie)	3	4	3	3 (Tie)
Landfill + Recycling + Yard/Food Waste *LF + R + YW/FW (3 bin)*	2	2	2 (Tie)	2	2

[a]If GHG emissions varied by less than 5% between two scenarios, they were considered tied.
[b]Primary LCA assumes landfill gas is used for energy, biogenic carbon storage credits and a methane GWP of 25 kg CO_2/kg CH_4.
[c]Appendix C contains results and additional discussion of the alternative modeling scenario assuming landfill gas is flared.
[d]Appendix B contains additional discussion of carbon accounting and the alternative modeling scenario

In the primary LCA, the LF + R (2 bin) management scenario ranked first with the lowest systemwide GHG emissions, followed by the LF + R + YW/FW (3 bin) scenario. The LF (1 bin) and LF + R + YW (3 bin) scenario tied for third with the highest systemwide GHG emissions. When LFGTE is replaced with gas flaring, LFGTE offsets are lost and landfill emissions increase. In the LF (1 bin) scenario, switching from LFGTE to landfill gas flaring increased systemwide GHG emissions by 30%. As a result, the LF (1 bin) scenario surpasses the yard waste composting scenario (i.e. LF + R + YW) in the ranking for highest GHG emissions. When the biogenic carbon accounting does not consider storage, long-term biogenic carbon storage no longer results in a carbon storage credit. In this alternative scenario, total landfill emissions increase in all scenarios since landfilling no longer provides a carbon storage credit. Since biogenic CO_2 emissions from waste are now counted in total CO_2 emissions, this impacts GHG emissions not just for landfilling, but for composting as well (see Appendix B, Table B1). The shift in carbon accounting results in the LF + R + YW (3 bin) management scenario performing the worst, and the LF + R + YW/FW (3 bin) scenario being roughly equivalent to the landfill-only (LF) scenario. When the GWP for methane is increased from 25 to 34 kg CO_2/kg CH_4, GHG emissions increase for all scenarios, and GHG emissions increase the most for management options with higher CH_4 emissions (i.e. landfilling), and less for options with lower CH_4 emissions (i.e. composting). In this alternative scenario, the LF (1 bin) management scenario results in the highest GHG emissions, followed by the LF + R + YW (3 bin) scenario, the LF + R + YW/FW (3 bin) scenario, and the LF + R (2 bin) scenario.

Although assumptions regarding LFG management, carbon storage, and methane GWP did alter the rankings in certain instances, the average rankings of the program options were unchanged from the initial results. In all analyses, the LF + R (2 bin) management scenario ranked first with the lowest systemwide GHG emissions, followed by the LF + R + YW/FW (3 bin) scenario. On average, the LF (1 bin) and LF + R + YW (3 bin) scenarios were tied for the highest GHG emissions overall.

Additional details on alternative assumptions made for LFGTE/flaring and carbon storage are provided in Appendix B and C.

IMPLICATIONS FOR SUSTAINABLE MATERIALS MANAGEMENT

LCA studies, including the one performed as part of this study, provide insight into MSW management options and their relative environmental impacts. While results from this study are valuable for comparing specific MSW management scenarios, additional insight can be derived by examining these results in context with other relevant LCAs for the U.S. and, where applicable, elsewhere.

Implications for Curbside Recycling. The LCA performed as part of this study, as well as previous U.S. focused research studies, found MSW recycling to be favorable relative to landfilling for GHG and energy use (Figure 5; Morris, 2005; ICF International, 2018; Kang et al., 2017; Franchetti and Kilaru, 2012; Kaplan et al., 2009). The level of environmental benefit realized from recycling can be influenced by a number of factors, such as:

- the materials included in recycling programs (plastics, glass, fiber, metals)
- how successfully residents source-separate materials for recycling (i.e. "curbside capture rate")
- the type of curbside recycling program (e.g. single-stream, dual-stream)
- the location and type of end-uses markets (e.g. bottle-to-bottle) for each recovered material

These factors suggest that maximizing the environmental benefit of recycling is dependent on variables that waste management entities can control (e.g. type of program) and variables that are out of their control (e.g. source separation by residents, end market viability). Further, the type(s) of materials included in a program may be influenced by what processing technologies the MRF has, available end markets, and economic factors. These observations highlight that recycling materials is complex and involves a variety of stakeholders and decision makers to be successful economically and environmentally.

Materials Accepted for Curbside Recycling. Recycling programs across the U.S. may include a variety of materials in their residential curbside program depending on factors such as material markets, MRF design, commodity prices, and recovery mandates. Material-specific LCA results, including those performed in this study, suggest that curbside recycling can result in GHG and energy savings for common curbside recyclable materials. Because each type of recyclable material has different properties, production activities and remanufacturing processes (e.g. paper production vs. metal mining and smelting), the level of GHG or energy savings from recycling rather than landfilling varies depending on the material (Tables 6 and 7). For example, results from the U.S. EPA WARM LCA model demonstrate that the magnitude of the emissions vary widely by the type of material. Avoided emissions associated with recycling rather than landfilling are highest for aluminum cans (9,130 kg CO_2 avoided/ton aluminum recycled), a value that is roughly 30 times higher than glass which has the lowest avoided emissions (300 kg CO_2 avoided/ton glass recycled). Similarly, energy savings per ton of recycled material are highest for aluminum cans (153 BTU/ton aluminum recycled), which is roughly 64 times higher than glass which has the lowest energy savings per ton of material recycled (2.4 BTU/ton glass recycled).

Table 6. GHG emissions from recycling and landfilling common materials using WARM (ICF International, 2018b). Units are kg CO_2-eqivalents/ton recycled material.

Material	Net Emissions from Recycling[a]	Net Emissions from Landfilling[b]	Avoided Emissions from Recycling versus Landfilling
Aluminum Cans	-9,110	20	9,130
Mixed Metal	-4,340	20	4,360
Mixed Paper (office)	-3,590	170	3,760
Mixed Paper (residential)	-3,530	70	3,600
Corrugated Container	-3,120	230	3,350
Steel Cans	-1,810	20	1,830
PET	-1,120	20	1,140
Mixed Plastic	-1,020	20	1,040
HDPE	-870	20	890
Glass	-280	20	300

[a]Values include: collection, transportation, and credits for either closed- or open-loop recycling depending on the material.
[b]Values based on a default mix of landfills with gas-to-energy, landfills with gas flaring, and landfills without gas collection.

Table 7. Energy use from recycling and landfilling common materials using WARM (ICF International, 2018b). Units are million Btu/ton recycled material.

Material	Net Energy from Recycling[a]	Net Energy from Landfilling[b]	Energy Savings from Recycling versus Landfilling
Aluminum Cans	-152.76	0.27	153.03
Mixed Metal	-65.99	0.27	66.26
HDPE	-50.20	0.27	50.47
Mixed Plastic	-38.84	0.27	39.11
PET	-31.87	0.27	32.14
Mixed Paper (office)	-20.85	-0.18	20.67
Mixed Paper (residential)	-20.45	-0.19	20.26
Steel Cans	-19.97	0.27	20.24
Corrugated Container	-15.07	-0.25	14.82
Glass	-2.13	0.27	2.4

[a]Values include: collection, transportation, and credits for either closed- or open-loop recycling depending on the material.
[b]Values based on a default mix of landfills with gas-to-energy, landfills with gas flaring, and landfills without gas collection.

One caveat to interpreting and using results from recycling LCAs is that generally the values provided examine a "best case" recycling scenario and, therefore, provide maximal GHG and energy savings. This is due to assumptions about material end use/remanufacturing. In the case of WARM, recycled material end use is assumed to be:

- a closed-loop recycling system, where recovered material is used in the remanufacturing of similar product (e.g. aluminum cans recycled into new aluminum cans), or
- an open-loop recycling to the most suitable good in the case of materials that decrease in quality (e.g. high quality paper recycling to lower-quality due to decrease in fiber length).

While these end use assumptions suggest that the values in Tables 6 and 7 represent best-case GHG and energy savings, they demonstrate an important consideration for recycling programs. While recycling 1 ton of aluminum cans and 1 ton of glass count equally in a tonnage-based recycling metric (i.e. they both count as 1 ton of material recycled), the environmental benefits of recycling 1 ton of aluminum cans and 1 ton of glass are very different from a GHG and energy savings perspective.

Another consideration in using numeric results from Tables 6 and 7 for recycling program planning is that these values are presented per ton of each specific material recycled. As a result, while they demonstrate the difference of recycling 1 ton of glass compared to 1 ton of aluminum cans, these results do not provide the relative systemwide impacts of including a specific material (e.g. aluminum cans) in a curbside recycling program since, by weight, the recycling stream contains significantly more of some materials (e.g. paper) than others (e.g. aluminum) (Table 8). Therefore, to calculate the overall systemwide impact of including various materials in a recycling program, factors such as waste composition and curbside capture rate must also be included as part of the analysis. These factors were included in the LCA performed as part of this study using SWOLF (Appendix A), allowing the impact of including specific materials in the recycling program to be explored.

Table 8. Composition of the recycling stream used in this LCA study.

Waste Material	Portion of the Recycling Stream, by weight (%)
Fiber (OCC, mixed paper)	54.6
Glass Containers	11.3
Plastic (HDPE and PET) Containers	5.0
Ferrous Containers	2.3
Aluminum Containers	1.2
Other/Contamination	18.0
TOTAL	**100**

The materials included in this LCA study were: glass containers, metal containers (ferrous and non-ferrous), plastic containers (PET and HDPE), and fiber (OCC, mixed paper). To understand how each material affects the sustainability of curbside recycling, the impacts of including or removing each material from the program were evaluated. LCA results suggest that, under base case assumptions (e.g. closed-loop or best-use remanufacturing), there are systemwide GHG and energy benefits associated with including each material type in the curbside recycling program (Figures 7 and 8). The largest impact on overall GHG emissions was the inclusion of aluminum containers, at -24.6 kg CO_2e/metric ton generated. This means that when aluminum cans are included in the curbside recycling program, overall GHG emissions for the MSW management system are reduced by 24.6 kg CO_2e for each metric ton of MSW managed. Glass containers had the smallest impact on systemwide GHG emissions, at -5.9 kg CO_2e/total metric ton. This means that if glass previously going to closed-loop recycling is removed from the curbside program, the impacts of managing each metric ton of generated MSW would increase by 5.9 kg CO2e. Fiber recycling resulting in the largest program-wide energy savings, at –2,562 MJe/total metric ton (-1711 kwh/metric ton). Ferrous containers resulted in the lowest energy savings benefit of –112 MJe/total metric ton (-31 kwh/metric ton).

Figure 7. The net change in GHG emissions, and the range of potential values, associated with including each material in a recycling program.

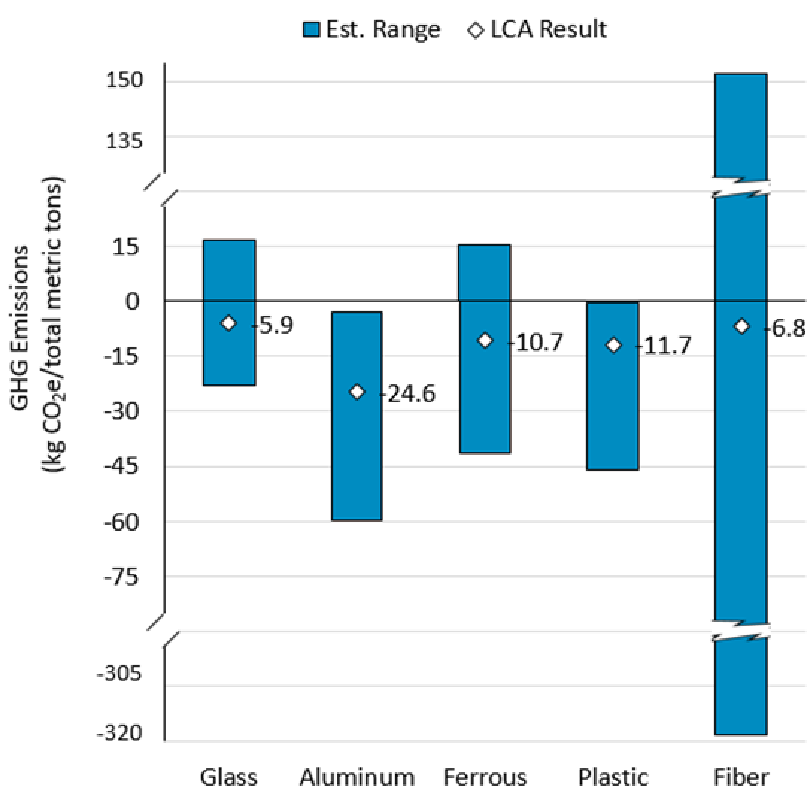
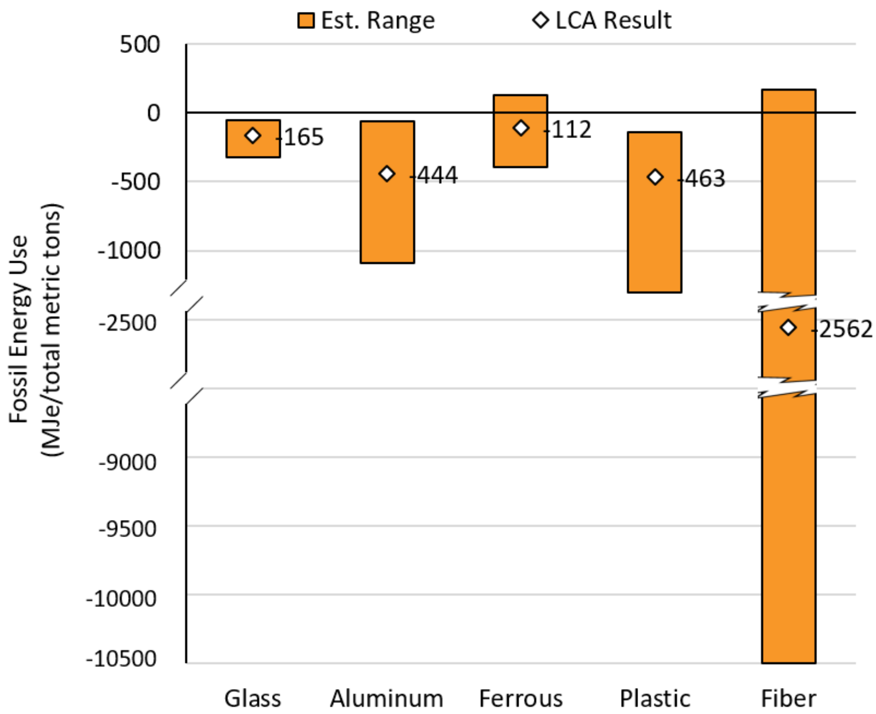


Figure 8. The net change in fossil energy use, and the range of potential values, associated with including each material in a recycling program.



As mentioned previously, the environmental benefits realized during the remanufacturing process account for the majority of the GHG and energy savings of recycling (Tables 3 and 4). The level of these benefits can be impacted by a variety of factors, such as: the remanufacturing process, energy efficiency, transportation distances, and electrical energy grid mix. These factors are discussed in more detail later in this report. For materials with small GHG or energy savings, an increase in remanufacturing emissions or energy use could potentially negate the benefits of recycling and result in a situation where recycling a material results in higher emissions or energy use than if the material were not recycled.

To explore the potential for this situation to occur for each material, GHG and energy savings were calculated based on the best and worst case values for reprocessing, creating conservative ranges for each material (Figures 7 and 8). Ranges that include positive values indicate that there are potential situations where environmental impacts are higher if a material is recycled versus if a material is not recycled. The GHG emissions analysis shows that there are potential situations where excluding glass, ferrous, and fiber from single-stream recycling programs could improve overall performance by reducing CO_2e emissions (Figure 7). For fossil energy savings, both ferrous containers and fiber showed the potential for situations where not recycling these materials could result in fossil energy savings (Figure 8).

Material Capture and Separation. Results from this LCA indicate there is significant potential to decrease program-wide GHG emissions by improving how well residents correctly sort recyclables into the recycling bin (i.e. "curbside capture rate"). In the primary LCA analysis, it was assumed that residents place 25% to 83% of recyclable materials into the recycling bin, depending on the material (Appendix A). This leaves significant room for improvement through activities such as education, outreach, and labeling. For example, a 10 percentage-point increase in curbside capture for each material (i.e. an increase in the above range of 35% to 93%, depending on the material) could decrease program-wide GHG emissions by nearly 25 kg CO2e per metric ton of MSW managed. Given the varying level of offsets for recyclable materials (Figure 7), greater emissions savings could be achieved by improving the capture rate of materials with the greatest GHG benefits (i.e. aluminum cans) rather than targeting all materials equally.

Over the past decade, curbside residential recycling programs have transitioned to single-stream (i.e. co-mingled) recycling programs, partially in an effort to increase curbside capture. Programs that transition to single-stream carts anticipate an increased amount of recyclable materials. Fitzgerald et al. (2012) performed a LCA comparison of dual-stream and single-stream collection and processing of curbside recyclables, to understand the sustainability implications of making this change. Researchers used data from U.S. MRFs and collection programs. Results showed that MRFs and cities that switched to single-stream recycling (collection and processing) experienced "considerable GHG emissions benefits", reducing net emissions by approximately 51% (Table 9). Although the single-stream system had higher contamination rates (requiring landfilling or WTE incineration) and MRFs required more electricity due to additional sortation equipment, these increased environmental impacts were outweighed by the offsets generated due to the increased recyclable commodities sent to remanufacturing.

Table 9. GHG emissions for dual- and single-stream recycling (from Fitzgerald et al., 2012). Units are kg CO_2-equivalents/metric ton recyclables processed.

Process	Dual-Stream	Single-Stream	Difference	% Difference
Remanufacturing	-1451.62	-2142.53	-690.91	-48 %
Separation	11.15	11.07	-0.08	-0.7%
Collection	58.31	37.89	-20.42	-35 %
TOTAL	**-1382.16**	**-2093.57**	**-711.41**	**-51 %**

Transport from MRFs to End Users. While GHG emissions associated with the transportation of recovered materials (e.g. finished bales) from the MRF to end users was a relative small portion of total emissions, they do have the potential to affect systemwide GHG benefits. This is especially true for materials with more marginal emissions benefits associated with their recycling such as fiber and glass (Figure 7). The relative impact of transporting recovered material (i.e. MRF output) was explored by examining how far transport vehicles travel from MRF to end user, and how closely to maximum weight the vehicles are loaded. Breakeven distance from the MRF to the end user is the distance where there are no net GHG benefits associated with recovering the material for remanufacturing compared to landfilling. Thus, transporting materials farther than the breakeven distance means landfilling the material would achieve lower GHG emissions than recycling the material. Results for over the road (OTR) transport suggest that fiber has the shortest breakeven distance at 360 miles by truck. Glass has the second shortest OTR breakeven distance for truck transport (1,150 miles) assuming bottle-to-bottle glass remanufacturing. Both plastics and metals could be transported over-road from coast-to-coast (~3,200 miles) without exceeding the breakeven distance. The important caveat to these results is that the assumption made is closed loop (e.g. bottle to bottle) or best case recycling. Therefore, end uses that are not closed loop or best case will result in breakeven distances that are lower, and in some cases this could be significant.

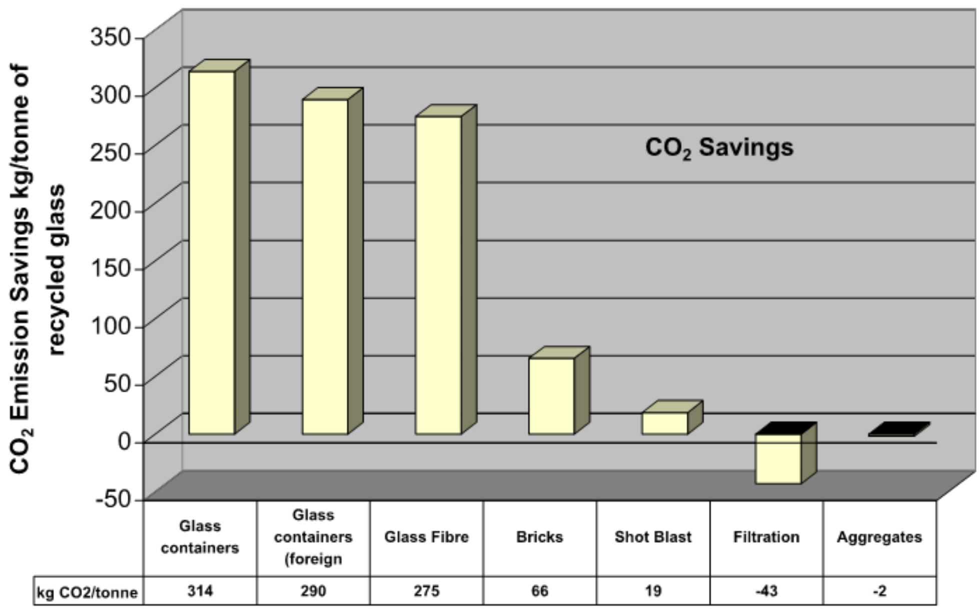
In addition to OTR transport, recyclables are also commonly transported across the U.S. via rail and exported overseas through shipping ports. A recent LCA performed on the transportation of goods compared OTR, rail and ocean-going vessels on the basis of energy use, GHG emissions, and other conventional air pollutants (Nahlik et al., 2015). Results indicate that OTR vehicles have the highest energy use and GHG emissions of all options. Of the OTR vehicles, heavy-duty transport (e.g. tractor-trailers) performed slightly better than medium-duty (e.g. single-unit box truck). Energy use and GHG emissions via rail and ocean-going vessels were significantly lower than OTR transport. Compared to heavy-duty diesel trucks, diesel train transport provides an 80% reduction in both GHG emissions and energy use while transport via container ship results in a 97% reduction in GHG emissions and energy use (Nahlik et al., 2015).

Another factor in transportation efficiency from MRF to end user relates to how full the vehicles are, by weight. LCA results demonstrate that recycling is favorable to landfilling as long as the vehicles are at least 8% full, which is quite low. Naturally, the closer the transport vehicle is to max weight, the lower the program-wide GHG emissions. For example, an increase from 30% of maximum weight to 60% results in GHG savings of 10 kg CO_2e per metric ton of MSW managed. This relationship is not linear, however, and these savings become incrementally lower as the vehicle approaches maximum weight.

End Use and Remanufacturing Using Recovered Materials. As mentioned previously, most LCA studies assume processed materials are being used for closed-loop recycling, or best case open-loop for materials such as paper or plastics. This assumption of best case end use provides results from LCAs that yield, in most cases, the lowest possible emissions or energy use for recycling of specific materials.

This is an important consideration because not all recyclable material recovered via MRF will be sent to an end-user for the best case recycling process. In these cases, the potential offsets for recycling may be significantly smaller. A key example of this is in the recycling of glass. A study by Enviros Consulting (2003) used LCA to evaluate the GHG benefits from different end uses for recycled glass in the United Kingdom compared to landfilling glass (Figure 9). While emissions savings from using glass containers for closed-loop recycling to new glass containers were estimated to be as much as 314 kg CO_2/metric ton, the study found savings were significantly smaller if recycled glass replaces virgin inputs for the manufacturing of bricks or shot blast. Further, when used for filtration or aggregates the savings were negative, indicating that landfilling glass would result in lower CO_2 emissions for these end uses.

Figure 9. Remanufacturing emissions savings due to various end-use options for glass, from Enviros Consulting (2003).[a]

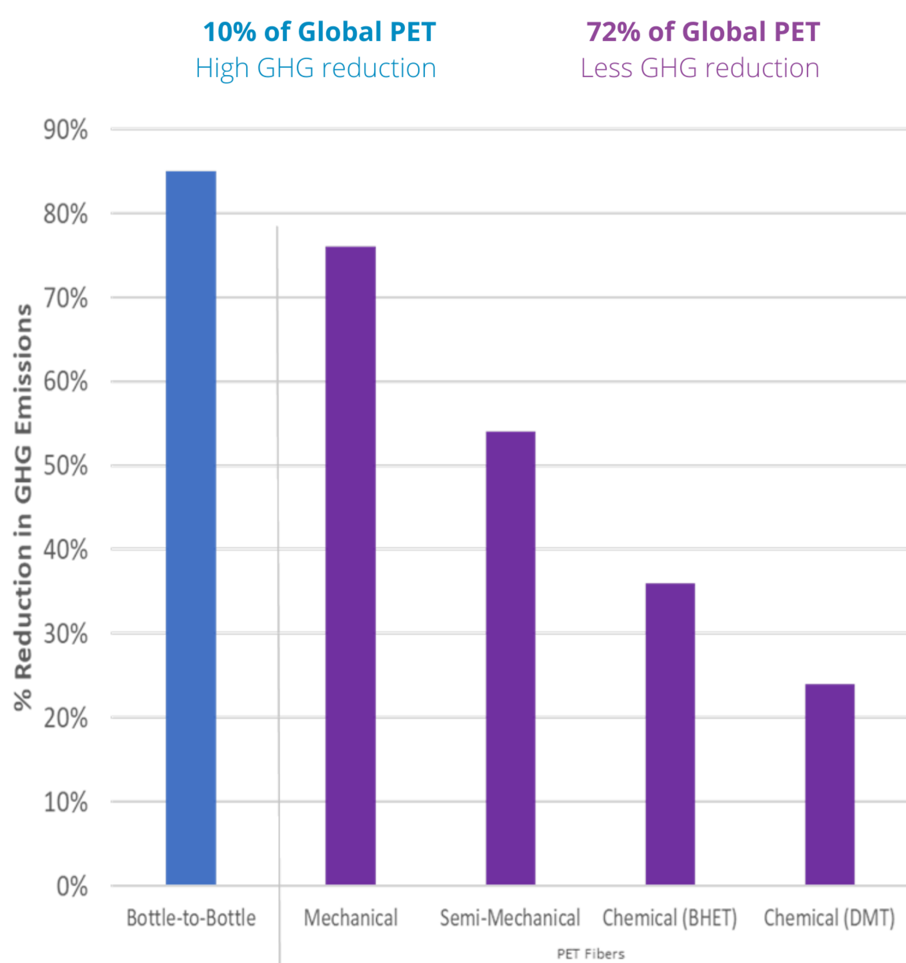


[a]1 tonne = 1 metric ton.

In practice, geographical location can impact LCA findings and given this study was conducted in the UK the findings may not be fully applicable to the US. Nonetheless, the primary implication of the Enviros Consulting (2003) study is the end use of recycled materials is of large importance. Lower end-use emissions or energy savings will cause other variables (e.g. transportation distance from MRF to end-user, energy efficiency of MRF equipment) to have more impact on the results or even flip the results such that recycling is no longer environmentally favorable compared to other options such as landfilling or WTE. For example, glass has relatively small savings even from recycling when assuming closed-loop remanufacturing (Table 6). Based on an OTR breakeven distance of 1,150 miles for glass, the results from Enviros Consulting (2003) suggest break even distances for recycled glass going to bricks or shot blast of approximately 240 miles and 70 miles, respectively.

A key reason that 'best case' or closed-loop assumptions are being made in LCAs, including in the results presented in this report, are that end use data are not widely or publicly available. Such data is critically important. For example, if end market demand for a particular material is limited and environmental benefits for recovery are negligible or do not exist, this may suggest the most sustainable short term option is landfilling the material until more sustainable options for this material can be designed by upstream manufacturers. Consider PET recycling, which typically is assumed in LCA modeling to be bottle-to-bottle remanufacturing. While data is limited, a study looking at end use data for PET indicated that only 10% of global production of PET is recycled bottle-to-bottle while 72% is remanufactured into fibers for products such as carpet (Noone, 2008). This suggests that GHG benefits of recycling PET to fiber can be as much as two-thirds lower than bottle-to-bottle recycling (Figure 10). PET bottles that go to fiber will greatly reduce the number of times PET might be recycled because recovering PET fiber (e.g. from carpet) is currently limited in the U.S. Further, the lack of parity between the assumption made by LCAs and the actual end uses of recycled materials could result in significant over estimates of environmental benefits. While the Enviros Consulting (2003) and Noone (2008) studies suggests the results could be drastically different, more research is necessary to confirm how end use impacts LCAs.

Figure 10. Percent reduction of replacing virgin PET inputs with recycled PET for both bottle-to-bottle and PET fiber remanufacturing ***(Noone, 2008; Shen, 2010; Brogaard, 2013).***



Implications for Organics Management. Of the MSW management scenarios considered in this analysis, results indicate that program-wide GHG emissions increase when either yard waste or mixed organics composting is added to a curbside recycling and landfilling program. For the LF + R + YW (3 bin) scenario, the increase in GHG emissions roughly canceled out GHG benefits associated with curbside recycling (Figure 5). While LCA results for composting scenarios (i.e. LF + R + YW and LF + R + YW/FW) may seem to go against conventional wisdom regarding the sustainability of composting, many other studies support the finding that composting yard waste organics does not always have clear GHG benefits over landfilling (Laurent et al., 2014; Levis and Barlaz, 2011; van Haaren et al., 2010).

GHG emissions of landfilling and composting, however, are highly dependent on the type of system being considered. Landfills with gas collection and/or gas-to-energy systems will have lower GHG emissions as CH_4 emissions into the environment will be reduced through gas collection and conversion, as well as the potential for credits for energy production. Composting systems will have different impacts depending on the level of aeration (to minimize CH_4 production) and the method for aeration (e.g. in-vessel, aerated pile, etc.). When landfilling and composting are compared, results are mixed as to which endpoint has the least GHG emissions due to factors such as:

- the type of organic waste (food, yard, leaves, grass, etc.)
- the type of composting system
- how well-managed/aerated the compost system is
- waste composition (e.g. % food waste, yard waste) and resulting LFG production rate
- the collection efficiency of landfill gas systems
- the moisture content of the composting feedstock
- whether compost product is offsetting the use of peat or chemical-derived fertilizer

Differences in Yard Waste and Food/Yard Waste Collection Scenarios. When the yard waste collection scenario (LF + R + YW) is compared to the mixed organics scenario (LF + R + YW/FW), overall GHG emissions decrease by 15 kg CO_2/ metric ton MSW when food waste goes to composting (Table 3), due to differences in yard and food waste properties. For example, some organics (e.g. branches) degrade more slowly, which allows the landfill gas collection system to capture these emissions. In contrast, other organics such as food waste degrade quickly such that the biogas is typically not captured, which results in higher GHG emissions.

Type of Composting System. Previous LCA studies suggest that GHG emissions for compost can vary greatly depending on the type of composting system. Different commercial organics management strategies were compared Levis and Barlaz, (2011) who found that composting emissions can range from -150 to -60 kg CO_2/metric ton (Table 10).

Windrow composting had the lowest GHG emissions, followed by Gore cover technology, aerated static pile (ASP), and in -vessel configurations. Morris et al. (2013) reviewed 82 organic waste management LCAs and found GHG emissions for backyard composting ranged from -690 kg CO_2/metric ton (nearly as low as anaerobic digestion) to 290 kg CO_2/metric ton (nearly as high as landfilling without gas collection). This wide range of values is due, in part, to the level of mixing. A well-maintained and well-aerated backyard compost pile will maintain aerobic conditions, greatly reducing methane (CH_4) emissions from the pile. A poorly managed compost pile, on the other hand, can become anaerobic and produce CH_4, which has significantly higher global warming potential compared to CO_2. As such, the type of composting system and its proper management can impact the relative performance of MSW recovery programs that include composting.

Table 10. GHG emissions and relative ranking of organic waste management options.

	Levis and Barlaz		Morris et al.[a]	
Management Option	**Ranking**	**kg CO_2/tonne**	**Ranking**	**kg CO_2/tonne**
Anaerobic Digestion	1	-400	1	-740 to -60
Commercial Composting	(3 – 6)	-150 to -60	2	-260 to +60
Windrow	3	-150		
Gore	4	-100		
ASP	5	-75		
In-Vessel	6	-60		
Backyard Composting			5	-690 to +290
Conventional Landfilling				
w/out LFG collection	9	+1,150		
w/ LFG flaring	7	-25	3	-60 to -50
w/ LFG to Energy	2	-230	6	-310 to +550
Bioreactor Landfilling	8	-24		
Waste-to-Energy Incineration			4	-240 to +20

[a]Morris et al. (2013) was a review of values in 82 LCAs of organic waste management. Ranking shown is based on the average values removed of outliers. Range in result also presented without outliers.

Composting Benefits. It is worth noting that, as with any LCA, the relative ranking of the scenarios is a function of the environmental impact categories examined. In this study, environmental impacts evaluated were GHG emissions (i.e. CO_2-equivalents) and fossil energy use. By nature, the aerobic decomposition process of composting converts biogenic carbon, in part, into CO_2 emissions.

The composting process also requires aeration to maintain aerobic conditions, and aeration strategies such as turning or air injection can rely on fossil energy. Although generally less favorable from the perspectives of GHG emissions and energy use, composting has other significant environmental benefits. The use of compost has been shown to improve soil quality and structure, control erosion and sedimentation, and improve water retention (ILSR, 2014). These benefits would not be captured in impact assessments focused solely on GHG and energy use, such as the one performed in this study. These benefits could be reflected in LCA results if different impact categories are chosen (Table 11), especially those that align with the potential benefits of composting such as water use, nutrient management (i.e. eutrophication) and soil health. Thus, results from this study relative to organics MSW management should be considered carefully in light of other potential benefits beyond GHG emissions and energy use.

Other Organics Management Options. While composting is the dominant end point for organic waste diversion, others exist. LCAs examining organic waste management have considered a variety of endpoints and uses for these materials, such as: commercial composting, backyard composting, anaerobic digestion (AD), conventional landfilling, bioreactor landfilling, in-sink disposal to publicly-owned treatment works (POTW) with AD, and use as alternative daily cover (ADC) at landfills. Prior studies suggest that no single option had the lowest impact across all categories of environmental burdens (Morris et al., 2013; Laurent, 2014; ODEQ, 2019). When considering only greenhouse gas emissions, studies suggest anaerobic digestion through either stand-alone AD facilities or sewer conveyance to POTWs is favorable compared to landfilling or composting of MSW food waste (Morris et al., 2013; Levis and Barlaz, 2011; ODEQ, 2014). For yard waste, a study by van Haaren et al. (2010) indicated using it as landfill ADC resulted in less GHG emissions compared to composting. These findings suggest that organics diversion needs to be carefully considered in light of local infrastructure. Composting facilities are the most common large-scale organics infrastructure in the U.S., with nearly 3,500 operating facilities accepting MSW organics (EREF, 2016). However, infrastructure exists in many states to manage potentially significant portions of the MSW organic stream through other pathways (Table 11).

Table 11. Suitability and Availability of MSW Organics End Use Options.

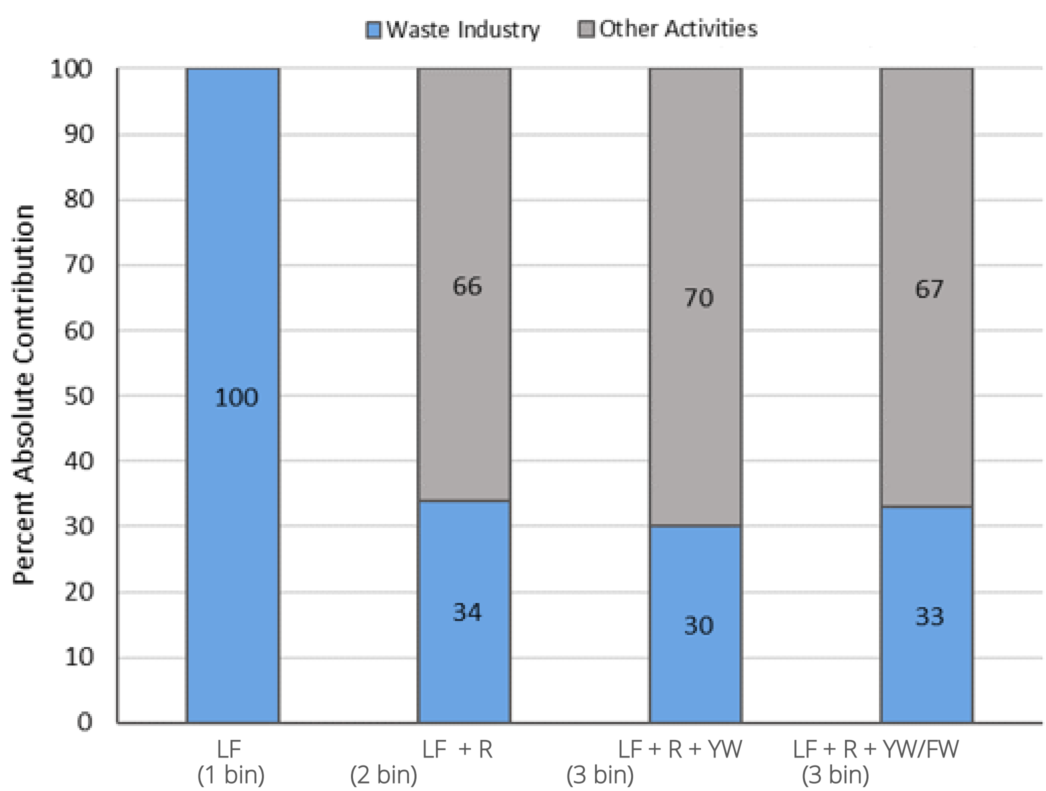
End Use Option	Suitable Wastes	Number of States	Number of Facilities
Composting[a]	Yard, Food	49	3,494
Anaerobic Digestion[b]	Food, Yard (limited)	32	181
In-sink to POTW-AD	Food	N/A	104[e]
Landfill ADC[c]	Yard	17[d]	699

[a]From EREF, 2016
[b]From EREF, 2015
[c]Estimated based on the number of MSW landfills in states without landfill yard-waste bans (EREF, 2015).
[d]An additional 5 states allow for yard waste disposal only in landfills with LFGTE systems, resulting in 23 states if included.
[e]From Willis et al., 2012

In addition to food waste diversion options, the prevention of food waste has substantial GHG emissions benefits. As part of an LCA-based approach to materials management, the Oregon Department of Environmental Quality examined local food waste management options, and estimated the environmental impacts of recovering food waste and reducing food waste generation. Results showed that reducing food waste generation by 40% had nearly double the GHG benefits of recovering 100% of food waste (Brown, 2018).

GHG Impacts Attributed to Waste Management Activities. It is important to note that GHG impacts from waste management are influenced by decisions from a variety of stakeholders, including product manufacturers, consumers, waste management entities and others. In a landfill-only (1 bin) MSW management scenario, the primary influence associated with GHG emissions and credits is from waste collection and landfill operations. With the addition of landfill diversion options (i.e. recycling and composting), however, a portion of the GHG emissions (e.g. reprocessing recycled feedstock) and/or credits (e.g. virgin feedstock offsets) are influenced by decisions outside the waste industry. In these scenarios, only 30% - 34% of GHG emissions and credits can be attributed to waste management activities (Figure 11). This means that other than the LF scenario, the majority of GHG emissions and energy use are outside the direct influence/control of the waste industry. The implication is that while decisions within the waste industry can certainly impact sustainability, the greater contribution to sustainable materials management (66 – 70%) is borne primarily by consumer behavior and decisions made by product manufacturing sector.

Figure 11. Contribution to total GHG emissions and credits falling within or outside of the waste industry for each scenario.



While the waste industry may not have direct control over the majority of GHG emissions in recycling and composting systems, there are still many opportunities for meaningful reduction in emissions. Additional findings of this study suggest:

- Landfilling has the largest impact on GHG emissions within the industry's influence, as anaerobic degradation of waste produces CO_2 and CH_4. Efforts to improve gas capture rates and minimize downtime of landfill gas management equipment provides the highest GHG benefits for landfilling waste materials.
- Collection activities (e.g. waste, recycling, organics) are the second largest contributor to waste management emissions. Many waste management companies are targeting collection emissions in their sustainability goals, including transitioning waste collection vehicle (WCV) fleets from diesel to CNG RNG, or electric vehicles. Based on the LCA analysis, for every 1% of collection activity (e.g. tonnage transported per mile) that was switched from diesel to CNG, there was a systemwide GHG emission reduction of 0.34 kg CO_2e/metric ton of MSW.
- Offsets such as recycling remanufacturing credits are key to realizing recycling benefits. While decisions regarding remanufacturing processes are not directly within the industry's influence (i.e. waste and recycling entities cannot control processing/remanufacturing decisions affecting emissions), elements such as MRF design and operation can impact the types and quality of bales produced and which may allow for improvement or refinement of end uses and markets.

Landfill Diversion as a Measure of SMM. Sustainable materials management (SMM) is an approach to managing waste as a resource, with goals that include increasing material reuse and reducing "environmental impacts throughout the material life cycle" (US EPA, 2019). While there are not universally defined metrics for SMM success, a common measure used in state SMM programs is landfill diversion rate. To date, 24 states have SMM goals focused on measuring landfill diversion, either solely or primarily as the percent of MSW going to recycling and composting (Kantner, 2019).

One key observation from the LCA modeling performed is that increased landfill diversion is not directly correlated with lower GHG emissions (Table 12). The LF + R (2 bin) scenario consistently had the lowest GHG emissions (Table 12; Table 5), but resulted in the lowest landfill diversion rate of 21%. While the LF + R+ YW (3 bin) scenario increased the diversion rate 9 percentage points (from 21% to 30%), it also increased emissions to approximately equivalent to the LF (1 bin) scenario, which has a 0% diversion rate. As such, a landfill diversion rate provides insight into material capture and potential reuse, however it is not a suitable measure for SMM from a GHG emissions perspective.

Table 12. Rankings, GHG emissions, and landfill diversion rates for modeled scenarios in this study.

Scenario	Ranking	Systemwide GHG Emissions (kg CO2e/metric ton MSW generated)	Landfill Diversion (% of MSW generated)
Landfill + Recycling (LF + R)	1	-225	21%
Landfill + Recycling + YW/FW (LF + R + YW/FW)	2	-180	38%
Landfill + Recycling + Yard Waste (LF + R + YW)	3-Tie	-165	30%
Landfill Only (LF)	3-Tie	-163	0%

KEY FINDINGS

A re-examination of material recovery is occurring within the solid waste management industry. This is driven, in part, by the challenges faced in the recycling industry and the emergence of sustainable materials management (SMM) concepts. Specifically, low commodity prices, a demand for lower contamination and poorly developed domestic end markets have created an impetus to better understand when recycling makes sense and, when it does, how recycling should be done. To remain viable long term, the success of recycling and composting rests upon the ability to demonstrate that it can achieve the triple-bottom line of people, planet and profit. Based on the results from this study and review of prior studies, a number of key observations can be made.

1. **The combination of landfill with recycling results in the highest GHG and energy benefits, but there are caveats.** The landfilling + recycling management scenario (LF + R, 2 bin) had the lowest GHG emissions and the lowest energy demand (Tables 3 and 4) and resulted in a 38% savings in emissions compared to landfill only (Figure 5). In many cases recycling offers significant benefits, but this result is not universally true due to factors such as assuming closed-loop recycling/end use, material type, energy grid, capture rate, etc. may result in different outcomes.
2. **Increased landfill diversion does not always correlate to lower GHG emissions.** While the LF + R (2 bin) scenario performed the best in terms of GHG and energy savings, it had the lowest diversion rate from landfill (Table 12). Further, while the addition of yard waste composting increased the diversion rate 9% (from 21% to 30%), it also increased GHG emissions to approximately equal to the landfill-only (1 bin) scenario. While computing landfill diversion provides insight into material capture and potential reuse, it is not a suitable measure for sustainable materials management from a GHG perspective.
3. **Curbside recycling may not provide emissions or energy savings in all situations.** The GHG and energy benefits of recycling vary based on the material, primarily due to differences in production and remanufacturing activities. Under closed-loop or best case remanufacturing, curbside recycling does offer GHG and energy savings benefits for the recyclable materials considered in this study (i.e. glass containers, aluminum cans, ferrous cans, HDPE and PET plastics, and fiber). However, glass, ferrous cans, and fiber had lower or marginal GHG or energy benefits, on average, and a wide range of potential emissions or energy usage outcomes, based on factors such as energy grid, production processes and end use (Figures 7 and 8). As a result, there are likely circumstances where it could be more favorable, from a GHG emissions or energy use savings standpoint, to landfill these materials rather than recycle them. A more detailed analysis of the prevalence and frequency of when recycling may not make sense requires additional data that is currently not readily or widely available.
4. **End uses of recovered materials appear to be critically important.** Sustainability benefits could be significantly diminished or even erased if recovered materials go to end uses that are not closed loop or best use. For example, glass recycled into new glass-bottles (i.e. closed-loop recycling) results in emissions savings of roughly 300 kg CO_2/metric ton recycled (Figure 9). If that glass is used for filtration or aggregate, the benefit appears to be greatly reduced and may result in more emissions than if it had been landfilled (Figure 9). However, such findings need to be verified with additional research since end use data in the U.S. is not widely available.

5. **Depending on end use, transport of recyclables to secondary processors can influence net GHG emissions and energy consumption.** For materials with lower or marginal GHG or energy benefits (e.g. glass, fiber), the location of remanufacturers or end-user relative to the MRF can impact sustainability. Results for fiber, for example, suggest that the benefits of recycling could be negated after over the road transport beyond 360 miles (see subsection Transport from MRF to End Users). This distance is impacted substantially by the type of end use. Estimates from a UK-based study (Enviros Consulting, 2003) suggest that if glass is used for shot blast rather than bottle remanufacturing, it can only be transported less than 10% of the distance before negating its recycling benefits (i.e. 70 miles vs. 1150 miles). This suggests that for some materials local end-use markets can be of high importance (i.e. fiber, glass). However, if alternative transport options are used (e.g. by rail or water) then transport distance becomes less influential on emissions or energy use.
6. **Human behavior can significantly impact GHG emissions.** How well residents correctly sort recyclables into the recycling bin versus the landfill bin (aka. 'capture rate') can reduce emissions. For example, a 10% improvement in capture rate could decrease emissions by 25 kg CO_2-e per metric ton of MSW managed. Further, given the level of offsets for recyclable materials varies based on the specific material, greater savings could be achieved by improving the capture rate of materials with the greatest environmental benefit (e.g. aluminum) rather than targeting all materials equally. This suggests efforts that can influence or direct human behavior, such as changes to product manufacturing, may be important drivers leading to maximized emissions savings.
7. **The effect of composting programs on GHG emissions differs depending on the types of materials accepted in the program.** While composting as a singular activity does indeed provide emissions savings, when analyzed as part of an integrated system less savings are realized. When the landfill + recycling scenario includes yard waste composting (LF + R + YW, 3 bin), yard waste is diverted from the landfill. Due to this, carbon storage offsets in the landfill from more slowly degrading yard waste materials (e.g. branches) are lost which results in emissions for the LF + R + YW scenario being similar to the landfill only scenario (LF) (Table 3, Figure 5). In contrast, the addition of food waste to composting programs (LF + R + YW/FW, 3 bin scenario) improved environmental performance compared to composting only yard waste. Composting both yard and food waste resulted in slightly higher emissions and energy use than the LF + R (2 bin) program, but still had 10% emission savings compared to landfill-only (1 bin) (Figure 5). The primary reason for this shift is because the majority of food waste rapidly degrades in a landfill which makes it difficult to capture for landfill gas to energy (since it degrades prior to the landfill gas system being installed) and offers little to no carbon storage benefit. Thus, when food waste is diverted this results in reduced fugitive emissions at the landfill.
8. **Anaerobic digestion is the most favorable option from a GHG emissions standpoint for food discards.** While this study indicated that composting food waste resulted in lower emissions and energy use than landfilling it, if food waste is managed by anaerobic digestion, it provides lower emissions compared to composting or landfilling (Morris et al., 2013; Levis and Barlaz, 2011; ODEQ, 2014). However, as described in this report, there may be other reasons to consider composting beyond GHG emissions mitigation and decisions made should consider such factors (e.g. soil improvement, water use).

9. **The most influential variable that impacts GHG emissions and energy demand for waste diversion strategies is offsets from recycling (by not using virgin materials).** It is essential to ensure that recycled materials are not only collected for remanufacturing, but that processes and end markets exist to ensure recovered material actually becomes inputs to the manufacturing process. The benefit of recycling comes primarily from using recycled feedstock to avoid mining or extracting virgin material for manufacturing. These manufacturing benefits can dwarf MRF emissions by nearly 200 times in the case of single-stream recycling (Table 9).
10. **Curbside waste collection activities are a minor portion of overall emissions and fossil energy demand.** While the transport of waste from the point of generation to disposal is considered one of the largest facets of the waste industry, they generally comprise less than 14% of gross GHG emissions and less than 24% of gross fossil energy demand. The exception is the fossil energy demand for the landfill only scenario (LF, 1 bin) in which waste collection comprises roughly 76% of total energy demand. The reason collection activities are generally a smaller proportion is due to the GHG and energy demand of other waste management activities, such as those related to diversion activities (Tables 4 and 5).
11. **LCAs likely overestimate recycling benefits because they typically assume recovered materials go to closed-loop or best case remanufacturing (e.g. bottle-to-bottle).** This assumption results in LCAs computing, in most cases, the highest possible environmental benefits for recycling since remanufacturing and end use drives the majority of systemwide offsets (Tables 3 and 4). This suggests that most LCA studies (including this one) may overestimate emissions and energy savings from recycling simply because it is highly unlikely to achieve situations where 100% of recovered materials are going to closed-loop or best case remanufacturing. Data from Noone (2008) that show 72% of global PET is not closed loop, which suggests assuming closed-loop or best case manufacturing may be unreasonable. However, more research is needed and data regarding end uses of recycled materials in the U.S. are currently not readily available.
12. **Locale-specific data can significantly influence LCA results.** Brogaard et al. (2014) demonstrated that LCA results vary substantially based on the lifecycle inventory datasets used. Thus, one cannot assume results from an LCA are applicable to a specific situation unless it can be confirmed that the data used in the LCA is relevant to that location. Only a small fraction of LCA studies focused on integrated waste management have been performed using U.S. data. Large differences in results can occur if the lifecycle inventory data used is based on a geographic locale where waste management practices and other infrastructure characteristics (e.g. energy grid, waste composition) differ substantially from the location of interest. This means that LCA results from other countries or different localized regions may not be applicable.
13. **Only 1/3 of GHG emissions and credits are attributed to waste management activities conducted by traditional waste industry entities (e.g. waste haulers, facility owners).** While activities by companies that collect, haul and process waste are critical to sustainable materials management, roughly 2/3 of emissions and credits for the landfill + recycling (2 bin) scenario are attributed to activities of secondary processors/product remanufacturers (Figure 11). This means that entities within the waste industry proper have less influence on overall GHG emissions. However, there are a number of opportunities for the waste industry to still reduce emissions by implementing strategies such as: improved landfill gas capture rates, converting waste collection vehicles from diesel to CNG, reduced contamination and technological improvements.

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Appendix A. Modeling Details and Assumptions

Process Modeling Description. Life-cycle process models exist to estimate GHG emissions and energy use from each of the processes shown in Figure 2. The LCA will use components of the Solid Waste Optimization Lifecycle Framework (SWOLF) as described by Levis et al. (2013 and 2014), with further information available at go.ncsu.edu/swolf. For each solid waste process, the SWOLF tool calculates the costs and emissions as a function of the mass and composition of the influent waste.

The collection process model was described by Jaunich et al. (2016a; 2016b) and estimates fuel, labor, and vehicle use for MSW collection systems. Transportation related emission factors were developed from the ecoinvent v3.01 database for medium-heavy duty trucks and heavy-heavy duty trucks (Weidema 2013).

The composting process model is based on an updated version of that described by Levis and Barlaz (2011, 2013a) and Hodge et al. (2016). The composting model assumes the use of aerated static piles for active composting followed by windrows for curing. After curing, the materials are screened, and the final soil amendment is transported, applied to land. It is assumed to offset peat use based on the data developed by Boldrin et al. (2010); no offset, and nutrient offsets will be explored in the sensitivity analyses. Soil carbon storage was estimated for the finished compost material.

The default single stream MRF is highly automated and was described by Pressley et al. (2014). The MRF uses disc screens to separate old corrugated cardboard (OCC) and other paper from the containers stream, while a vacuum is used to remove plastic film. Glass bottles are then manually sorted by color. Plastics are separated using optical sorters and manual sorting, while ferrous is separated with a magnet, and aluminum is separated using an eddy current separator. All of the separated materials except glass are then baled. The model also includes pickers that negatively sort dangerous or contaminating materials from the recyclable streams at the front end of the MRF. The materials that are not recovered from the residual, which is sent to the landfill.

The landfill process model has been described previously by Levis and Barlaz (2011) and Hodge et al. (2016). Values for landfill gas collection and oxidation were updated as described by Levis and Barlaz (2014). Landfill gas generation was modeled using a first-order decay model with material-specific decay rates and methane yields. The landfill was assumed to have a bulk decay rate of 0.04 yr^{-1}, which was used to scale the material-specific decay rates as described by De La Cruz and Barlaz (2010). Temporally averaged waste-age collection efficiencies were then applied to each material. In the Electricity Generation sub-scenario, the collected methane is assumed to be combusted in an internal combustion engine to generate electricity while the gas flowrate was great enough to do so (i.e., >350 ft^3/min). Gas collected prior to reaching the minimum flow rate is combusted in a flare.

A fraction of the uncollected methane is oxidized to CO_2 as it passes through the landfill cover soil. Temporally averaged waste-age oxidation efficiencies were developed based on guidance from the U.S. EPA (Title 40 CFR §98 Subpart HH) and ranged from 10% to 35% (Levis and Barlaz 2014). Landfills under final cover will generally have a relatively low flux through the cover which justifies the upper end of the range (35%). Similarly, landfills without a gas collection system will have a relatively high flux, suggesting that 10% is more appropriate. Landfills with a gas collection system in place but prior to final cover placement were assigned an oxidation of 20%. The biogenic carbon that is not released after 100 years is considered stored.

Recyclables that are recovered at the MRF are reprocessed into new materials. The life-cycle offsets associated with material recovery were developed by RTI (2003) with updates from Franklin and Associates (2011).

Table A1. Generated MSW Composition Used in SWOLF Analysis, based on US EPA (2016).

Category	Material	Composition (%)
Organics	Yard Trimmings, Leaves	5.3
	Yard Trimmings, Grass	4.0
	Yard Trimmings, Branches	3.9
	Food Waste - Vegetable	11.9
	Food Waste - Non-Vegetable	3.0
	Wood	6.2
	Textiles	6.3
	Rubber/Leather	3.2
Paper and Paperboard	Newsprint	2.9
	Corrugated Cardboard	11.8
	Office Paper	1.7
	Magazines	0.5
	3rd Class Mail\Other Commercial Printing	2.4
	Folding Containers	2.1
	Paper Bags	0.3
	Paper - Other	4.7
Plastic	PET (#1) – Containers	1.5
	HDPE (#2) - Translucent Containers	0.4
	HDPE (#2) - Pigmented Containers	0.8
	PVC (#3) – Containers	0.1
	LDPE (#4) - Containers	1.6
	Polypropylene (#5) - Containers	0.5
	Polystyrene (#6)	0.1
	Mixed plastic bags, sacks, and wraps	1.6
	Plastic - Other	6.4
Metals	Ferrous Cans	0.6
	Ferrous Metal - Other	6.2
	Aluminum Cans	0.5
	Aluminum - Foil	0.2
	Aluminum - Other	0.7
	Metals - Other	0.8
Glass	Glass - Brown	2.1
	Glass - Green	0.9
	Glass - Clear	0.6
	Glass - Other	0.9
	Other	3.4

Table A2. Waste composition and material capture rates.

Waste Materials	Percent Generated Mass	Recycling Capture Rates (%)	Composting Capture Rates (%)	SSMRF Separation Efficiency (%)
Yard Trimmings, Leaves	5.3	5.6	96.4	
Yard Trimmings, Grass	4.0	5.6	96.4	
Yard Trimmings, Branches	3.9	5.6	96.4	
Food Waste - Vegetable	12.1	5.6	50.0[b]	
Food Waste - Non-Vegetable	3.0	5.6	50.0[b]	
Wood	6.2	5.6	1.4	
Textiles	6.1	5.6	1.4	
Rubber/Leather	3.2	5.6	1.4	
Newsprint[a]	2.6	73.0	1.4	98.7
Corr. Cardboard[a]	11.9	67.8	1.4	98.7
Office Paper[a]	1.7	25.6	1.4	98.7
Magazines[a]	0.5	26.0	1.4	98.7
3rd Class Mail/Other Commercial Printing	2.3	70.4	1.4	98.7
Folding Containers[a]	2.1	41.5	1.4	98.7
Paper Bags [a]	0.4	25.0	1.4	98.7
Paper - Non-recyclable	4.5	5.6	1.4	98.7
HDPE (#2) - Translucent[a]	0.3	52.4	1.4	
HDPE (#2) - Pigmented[a]	0.6	52.4	1.4	98.0
PET (#1)[a]	1.1	53.9	1.4	98.0
Polypropylene (#5)	0.1	5.6	1.4	98.0
PVC (#3)	0.01	5.6	1.4	88.0
LDPE (#4)	0.02	5.6	1.4	
Polystyrene (#6)	0.03	5.6	1.4	
Mixed plastic bags, sacks, and wraps	1.6	5.6	1.4	81.0
Plastic - Other	9.4	5.6	1.4	
Ferrous Cans[a]	0.7	70.7	1.4	98.0
Ferrous Metal - Other	6.3	1.5	1.4	98.0
Aluminum Cans[a]	0.5	52.0	1.4	97.0
Aluminum - Foil	0.2	2.5	1.4	97.0
Aluminum - Other	0.7	5.6	1.4	97.0
Metals - Other	0.8	5.6	1.4	
Glass - Brown[a]	2.0	63.8	1.4	94.1
Glass - Green[a]	0.9	82.7	1.4	94.1
Glass – Clear[a]	0.6	60.4	1.4	94.1
Glass - Non-recyclable	0.9	5.6	1.4	
Misc. Organic	1.0	5.6	1.4	
Misc. Inorganic	2.5	5.6	1.4	

[a]Material included in the modeled recycling program
[b]Only included in the LF + R + YW/FW (3 bin) scenario

Appendix B. Carbon Accounting

Organic wastes are typically defined as those containing carbon from biological sources (i.e., biogenic C) (e.g., food, paper, and wood). During anaerobic degradation, some biogenic C is released in the biogas as CO_2 and CH_4, while some biogenic C remains stored in the landfill. When material is composted it aerobically degrades during composting and after land application. A relatively small fraction of the initial C remains stored in the soil for long periods of time. The GWP assigned to these flows will significantly affect the reported global warming impact from a facility (Christensen et al., 2009). There are two primary methods for estimating the global warming impact of biogenic C flows in solid waste management life-cycle assessments (LCAs). A summary of each method is shown in Table B1. If CO_2-biogenic (CO_2-b) (CO_2 containing biogenic C) is considered to have a GWP of 0, then stored biogenic C should have a GWP of -1 in CO_2 equivalents. This accounting method assumes that biogenic C will remain in the short-term carbon cycle, such that emitting it as CO_2-b results in no net impact on global warming, while removing it from the carbon cycle by storing it results in a global warming benefit. This accounting method is common in waste LCAs, but is unsuitable when system boundaries extend further upstream to biomass growth. The alternative accounting method assumes that biogenic C and fossil C (i.e., C from fossil sources such as coal or oil) are both stored when received by the waste management system. As such, the release of either CO_2-b or CO_2-fossil is equivalent, and keeping either stored has no net impact on global warming.

In end-of-life LCAs, both methods will generally result in consistent rankings of alternatives. However, if the LCA is going to be used in a broader product LCA that includes biomass growth, then the latter approach should be used to avoid double-counting (Christensen et al., 2009).

Table B1. Description of internally consistent GWP assignments for end-of-life biogenic C flows.

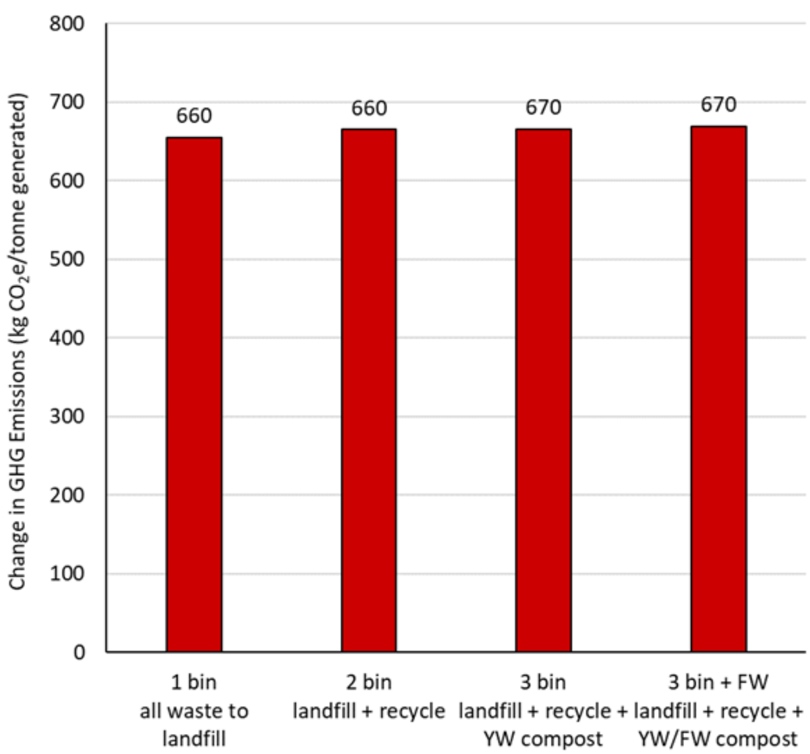
Name	**Base Assumption**	**Fossil CO_2 Emissions (kg CO_2e/kg CO_2-f)**	**Biogenic CO_2 Emissions (kg CO_2e/kg CO_2 -b)**	**Biogenic C Storage as CO_2 (kg CO_2e/kg CO_2-s)**
Neutral CO_2-b	Biogenic C is part of the short-term carbon-cycle.	1	0	-1
Neutral Stored Biogenic C	Biogenic C arrives stored and has the same impact as fossil C.	1	1	0

Alternative Scenario Modeling. As described above, most LCAs of SWM systems assume that the biogenic C in waste materials is part of the short-term carbon cycle. This means that emitting biogenic CO_2 has no climate impact, while keeping biogenic C stored for a long period of time (e.g., 100 years) instead of returning it to the atmosphere leads to a climate benefit.

The alternative biogenic C accounting method treats the biogenic C in waste as already stored, therefore keeping it stored has no climate impact, and releasing the stored biogenic C as CO_2 is treated the same as releasing fossil CO_2.

The net change in GHG emissions based on using the alternative biogenic C accounting is shown in Figure B1. The relative differences between the scenarios do not significantly change because biogenic C is balanced in and out of the system. CO_2-biogenic emissions are a direct result of a loss of stored biogenic C, so scenarios that emit a large amount of CO_{-2}-biogenic (e.g., Scenarios 3 and 4) necessarily do not store much biogenic.

Figure B1. The net change in GHG emission values when switching biogenic CO2 accounting from post-disposal storage to pre-disposal storage.

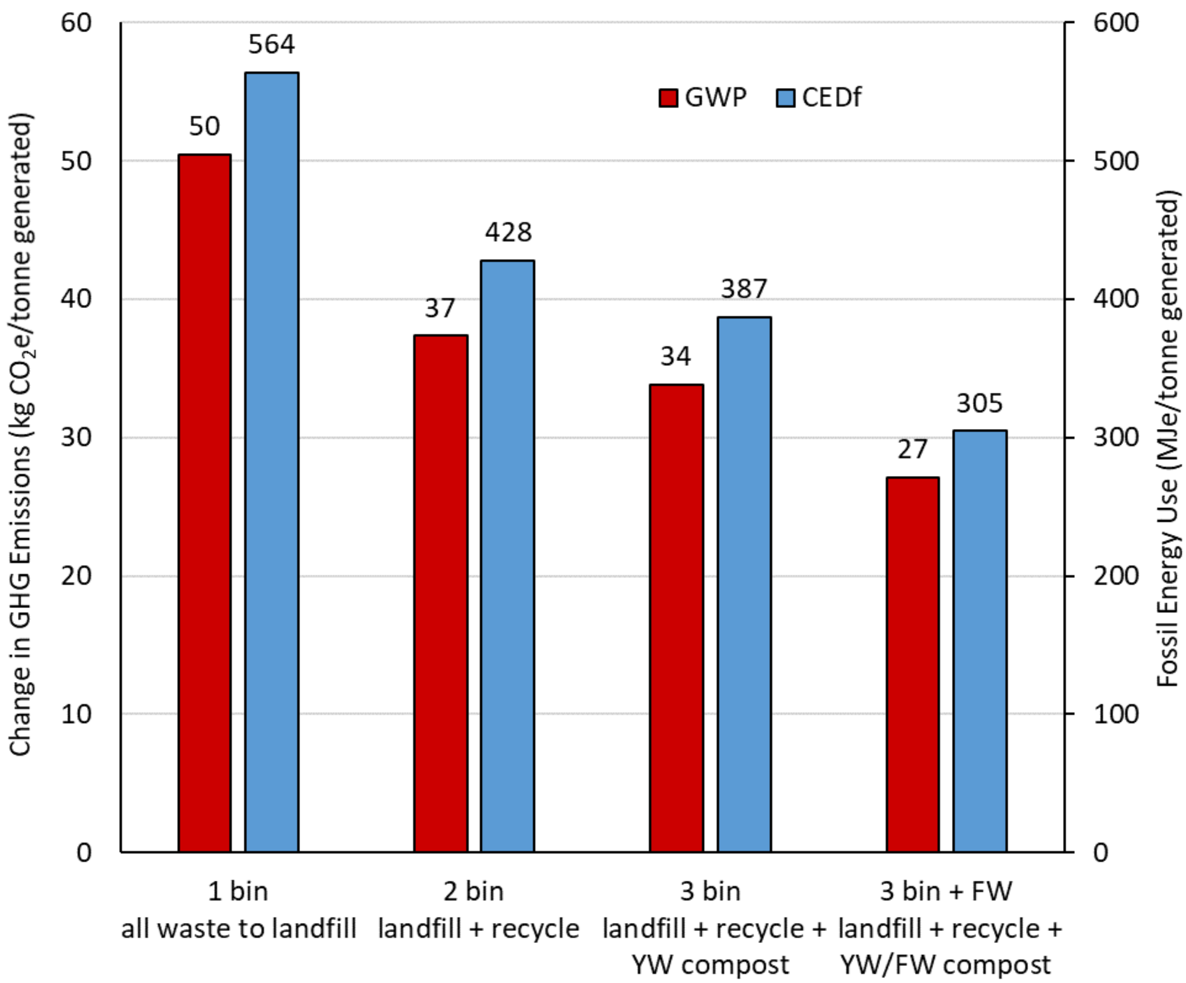


The only relative differences among the scenarios is due to stored biogenic C being emitted as CH_4 since all CH_4 emissions are treated the same in both scenarios (i.e., GWP = 25 kg CO_2e/kg CH_4). This means that the scenarios that rely more on landfill change by slightly less than other scenarios due to the significant CH_4 emissions from landfills. The relative rankings of the scenarios change slightly as the 3 bin scenario with yard waste composting now leads to most GHG emissions, and 1 bin and 3 bin + FW are effectively equivalent (Table 5).

Appendix C. Landfill Gas Flaring

In this alternative scenario, collected landfill gas is flared instead of being converted to electricity. The net change in GHG emissions and fossil energy use due to switching to landfill gas flaring from energy recovery are shown in Figure C1. GHG emissions and fossil energy use increase in all of the scenarios, and the increase is directly correlated with the mass be landfilled in each scenario. However, the rankings of the scenarios only change slightly from the base scenarios, as the increase in GHG emissions causes the LF (1 bin) and LF + R+ YW (3 bin) scenarios to no longer be tied (Table 5).

Figure C1. The net change in GHG emissions from switching to landfill gas flaring from landfill gas to energy recovery.



Appendix D. Documentation of Methane Yields Used in SWOLF

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March 19, 2016

The objective of this note is to document the updated methane yields and carbon balance material properties used in this analysis and the Solid Waste Optimization Life-cycle Framework (SWOLF). Eleazar et al. (1997) estimate the methane yields from several waste components, and Barlaz (1998) reported the carbon loss associated those methane yield results. A second CH_4 yield estimate was thus calculated by assuming that 50% of the organic C lost was converted to CH_4 and 50% to CO_2 as is typical for carbohydrates (cellulose and hemicellulose) based on the carbon loss reported by Barlaz (1998). It is recognized that food waste will contain some protein and fats which produce a slightly higher CH_4 to CO_2 ratio. However, the difference is small given the uncertainty of this work. In most cases the methane yield calculated from carbon loss was greater than the methane yield reported by Eleazer et al. (1997) However, in the case of newsprint and magazines, the original reported methane yields were greater, so those values were used. Updated carbon storage values were then calculated for these materials based on the reported methane yield. The percent of carbon converted to biogas was then calculated based on the stored carbon and the initial biogenic carbon content of the material reported by Barlaz (1998). A similar process was performed for wood based on data reported by Wang et al. (2011) The methane yield of food waste was developed from the mean of the 37 studies shown in Table D2, while the biogenic carbon content was developed from Riber et al (2009) The adopted methane yields, biogenic carbon content, carbon storage factor, and percent carbon conversion for each material are summarized in Table D1.

The methane yield calculation is illustrated below for branches. The branches sample tested was 49.4% C and the carbon storage factor is 0.38 kg C stored/kg dry substrate (de la Cruz and Barlaz, 2010)

0.494 - 0.38 = 0.114 kg C loss per dry kg

Methane Yield = 0.114 x 0.5 kg C to CH_4/ kg C x 1000 gm/kg / (12 gm/mole) * 22.4 liters/moles = 106.4 L CH_4/dry kg

Percent C Conversion = 100 x (49.4 – 380/10)/49.4 = 23.1%

Methane yields of 204 to 576 m3 CH_4/dry Mg of food waste have been reported (Table 2) and this uncertainty can lead to large impacts in the estimated performance of landfills and anaerobic digestion facilities.

Table D1. Waste component anaerobic degradation related carbon balance material properties.

Waste Component	Methane Yield (m^3/dry Mg)[a]	Biogenic C Content (%TS)[b]	Carbon Storage Factor[d] (kg C/dry Mg)[c]	Anaerobic Biogenic C Conversion[d]
Yard Trimmings, Leaves[e]	65.3	45.5	385	15.4
Yard Trimmings, Grass	194.8	44.9	240	46.5
Yard Trimmings, Branches	106.4	49.4	380	23.1
Food Waste – Vegetable[f]	369.0	47.7	81	82.9
Food Waste - Non-Vegetable[f]	369.0	56.5	169	70.0
Wood[g]	14.5	43.4	418	3.6
Textiles[h]	86.1	10.5	13	87.6
Rubber/Leather[i]	0.0	30.9	309	0.0
Newsprint[j]	74.3	49.2	412	16.2
Corr. Cardboard	195.1	46.9	260	44.6
Office Paper	263.6	32.2	40	87.6
Magazines[j]	84.4	34.3	253	26.4
3rd Class Mail[k]	76.3	46.3	381	17.7
Folding Containers[l]	159.1	46.9	298	36.4
Paper Bags[l]	192.0	46.9	263	43.9
Mixed Paper[m]	148.7	45.2	293	35.2
Paper - Non-recyclable[m]	148.7	45.2	293	35.2
Miscellaneous Organics[n]	179	42.2	231	45.4

a. Methane yields calculated from carbon loss reported by Barlaz (1998) except where noted.
b. Biogenic carbon content from Barlaz (1998) except where noted.
c. Reported by Barlaz (1998) except where noted.
d. Biogenic C Conversion = 100 x (Biogenic C – CSF/10)/Biogenic C
e. Leaves Biogenic C content is average of two measurements.
f. Food waste methane yield is the average of 37 studies shown in Table 2. Biogenic C content developed from Riber et al. (2009)
g. Used weighted average of lumber, plywood, oriented strand board, and medium-density fiber board reported by Wang et al. (2011) to calculate methane yield and biogenic C content.
h. Biogenic C content is based on 23.7% of textiles being cellulose with carbon content of 44.4 %TS. Biogenic C conversion is based on office paper.
i. Biogenic C content from Riber et al. (2009) with no biodegradation.
j. Directly used methane yield reported by Eleazar et al. (1997) because it was greater than the methane yield calculated based on C loss reported by Barlaz.[3] CSF = 1000 x (Biogenic C/100) – 2 x Methane Yield x (12.01/44.01) which assumes emitted C is equally split between CH_4 and CO_2.
k. Used weighted average of newsprint and magazines based on U.S. EPA (2015) discarded composition.
l. Assumed the same as cardboard.
m. Used weighted average of papers based on U.S. EPA (2015) discarded composition.
n. Used weighted average of all materials based U.S. EPA (2015) discarded composition.

Table D2. Existing literature values for food waste methane yields.

	Moisture Content (%ww)	VS Content (%TS)	Methane Yield m^3/Mg VS	m^3/Mg TS	m^3/Mg ww	Reference
	47	97	482	468	248	Cho et al. (1995)
	65	99	294	291	102	Cho et al. (1995)
	95	84	277	233	12	Cho et al. (1995)
	74	95	472	448	117	Cho et al. (1995)
	85	88	489	430	65	Heo et al. (2004)
	75[a]	93.8	320	300	75	Eleazer et al. (1997)
	70	83	445	369	111	Zhang et al. (2007)
	80.3	86	531	457	90	Qiao et al. (2011)
	90.9	84	281	236	22	Qiao et al. (2011)
	70.6	95.3	415	395	116	Browne and Murphy (2013)
	70.6	95.3	357	340	100	Browne and Murphy[9] (2013)
	75	93	338	314	79	Mohan and Bindhu (2008)
	75	93	288	268	67	Mohan and Bindhu (2008)
	75	93	219	204	51	Mohan and Bindhu (2008)
	71.5	89	344	306	87	Davidsson et al. (2007)
	72	87	349	304	85	Davidsson et al. (2007)
	71	92	275	253	73	Davidsson et al. (2007)
	72	92	289	266	74	Davidsson et al. (2007)
	68	87	284	247	79	Davidsson et al. (2007)
	75[a]	92[a]	630	572	144	Agyeman and Tao (2014)
	75[a]	92[a]	560	509	128	Agyeman and Tao (2014)
	75[a]	92[a]	470	427	107	Agyeman and Tao (2014)
	67	89.9	334	300	99	Gray et al. (2008)
	75[a]	92[a]	225	204	51	Staley et al. (2006)
	87.1	94.2	512	483	62	Lopez (2015)
	86.9	94.0	394	371	49	Lopez (2015)
	71.1	94.3	363	342	99	Lopez (2015)
	71.9	94.3	406	383	108	Lopez (2015)
	72.7	96.1	516	496	135	Lopez (2015)
	73.2	96.1	508	488	131	Lopez (2015)
	84.5	90.9	427	388	60	Lopez (2015)
	85.3	92.2	455	419	62	Lopez (2015)
	72	88.0	420	370	103	U.S. EPA (2008)[13]
	92.6	88.1	300	264	20	Lin et al. (2011)
	77.9	92.5	560	518	114	Lin et al. (2011)
	72	86.1	657	565	158	El-Mashad and Zhang (2010)
	88.3	92.3	464	428	50	Chu et al. (2008)
Min	47	83	219	204	12	
Max	95	99	657	576	248	
Mean	76	91	404	369	90	
Median	73	92	406	370	87	
StDev	9.7	4.1	111.9	103.3	43.2	

[a] Not reported. Mean value used.

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