2017

The Environmental Impacts of Alternative Food Waste Treatment Technologies in the U.S.

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Abstract: A Life Cycle Assessment (LCA) was conducted to determine the environmental impacts of several waste treatment scenarios for a suburban New York (U.S.) municipality. The study goal was to determine if separate food waste recovery and management was environmentally sounder than waste-to-energy incineration (the baseline case). Three alternatives, enclosed tunnel composting, enclosed windrow composting, and anaerobic digestion with subsequent enclosed windrow composting of residuals, were examined considering the entire residual waste stream (not just separated food wastes). Impact categories assessed were climate change, environmental eutrophication and acidification, resource depletion, and stratospheric ozone depletion. A normalized, aggregated impact assessment was created to compare the treatments across categories. The anaerobic digestion scenario scored best, followed by the tunnel composting and the baseline waste to energy incineration scenarios, and, last, the enclosed windrow composting scenario. Although it was possible to select an alternative that decreased environmental burdens compared to the business-as-usual case, all modeled scenarios resulted in higher overall environmental burdens than savings, underscoring the need to avoid creating waste to conserve resources and reduce environmental burdens, and ultimately lead to more sustainable waste management practices.

Keywords: food waste; environmental impact; composting; anaerobic digestion; incineration; LCA.

1. Introduction

Food wastage is a complex, interdisciplinary issue which can have profound effects for resource conservation (Thyberg and Tonjes, 2016). Food waste prevention and treatment with technologies that decrease environmental impact are increasingly considered as means to achieve more sustainable global food and waste systems. Policies addressing sustainable food waste management are being proposed and implemented, particularly in the U.S. and Europe. Focus has been placed on food waste due to concerns about the social, environmental, and economic costs of food waste.

Some portion of food waste, even if waste avoidance measures were to be successful, is unavoidable (Schott et al., 2013); reuse opportunities, through redistribution of edible food to humans or animals probably cannot account for the remainder due to perishability and high transport and distribution costs (Buzby et al., 2014), or the excess food may not meet safety or quality requirements (Salhofer et al., 2008). Furthermore, such prevention activities may not appeal to consumers on aesthetic or cultural grounds (Buzby et al., 2011). About 32 million tonnes (MT) of food waste is disposed annually in the U.S., which is 15% of all disposed municipal solid waste (MSW) (Thyberg et al., 2015). Currently waste planners and managers see diversion of this waste from landfills as a means of enhancing stagnant recycling rates, improving environmental conditions associated with waste management, and ultimately contributing to resource conservation and sustainability. Sound analyses of the environmental impacts of specific food waste treatment options would support the development of better and more successful diversion programs.

A life cycle assessment (LCA) is a system assessment tool that quantifies potential environmental exchanges and impacts of system processes. Outputs include indicators which simplify and organize inventory results to make them more understandable (Owens, 1999). Waste system LCAs quantify impacts of interconnected waste management technologies, from generation to final disposal/treatment based on a specified waste composition, and so allow for comparisons between options (Manfredi and Pant, 2013). Previous food waste LCAs usually only model the food waste portion of the waste stream and exclude impacts from other residual wastes (e.g., Lundie and Peters, 2005; Lee et al., 2007; Andersen et al., 2012). An evaluation of the entire system is required to determine which changes are needed for system improvement. This holistic approach also enables a more complete understanding of the overall system as additional factors can be included in the model, such as the effects of differing levels of source separation of the targeted materials. Modeling all residual waste is important when considering combustion technologies, too, since net energy production will be quite small for studies looking only at food waste due to high moisture content (Morris et al., 2014).

Most food waste focused LCA research has been performed in European settings (Laurent et al., 2014), with fewer LCAs performed in the U.S. Table S1 in the Supplementary Materials provides a review of recent food waste focused LCAs, their characteristics, and main findings. Considerable differences between LCA study findings regarding optimal food waste management have been found (Bernstad and Jansen, 2012). However, it is difficult to compare findings from various LCA studies due to differences in modeling approaches, assumptions, and functional units across studies.

The objective of this study was to use LCA to evaluate the environmental impacts of U.S. residential waste disposal to determine if environmental improvement can be achieved by adopting separate food waste recovery and treatment in a suburban municipality (Town of Brookhaven, Long Island, New York). Brookhaven currently disposes of collected wastes using waste-to-energy incineration (WTE) and there is no separation of food waste; this was considered the baseline scenario and alternatives to this baseline were evaluated. The findings were used to determine the conditions under which food waste recovery is beneficial, as well as how LCA analyses can be leveraged to effectively inform decision making focused on sustainable waste management. Emphasis was placed on evaluating the full residual waste stream going to disposal (not only food waste), as impacts and benefits are associated with the entire system of managing wastes, not just the food waste portion. When deciding on approaches for waste system improvements, it is essential to consider the system-wide context rather than just looking at the impacts associated with a single waste fraction. Additionally, determinations of exactly how to aggregate impact categories may affect the interpretation of potential system changes.

Thus, this study is unique because all residual waste was modeled for a suburban U.S. municipality, something previous food waste LCAs have not considered. Four food waste treatments were modeled, including WTE, two types of composting, and anaerobic digestion (AD), to quantify impacts on climate change, eutrophication, acidification, resource depletion, and stratospheric ozone depletion. This assessment indicated conditions where food waste recovery is beneficial and enabled determination of the management scenario with fewest environmental burdens. As mentioned, most prior food waste LCAs only consider food waste in isolation, and so changes in system-wide impacts from alternative food waste treatment are important to examine. Furthermore, no peer-reviewed LCA has been conducted for any of the municipal waste management systems on Long Island to date, although Long Island has been a U.S. pioneer in curbside recyclables collection and long-distance transport of solid waste, banned landfilling altogether in 1990, and sparked policy debates across the U.S. by launching the famous Garbage Barge of 1987 (Tonjes and Swanson, 1994). Ultimately, this investigation can support a discussion regarding effective decision making for sustainable waste management. Food waste is a topic of interest globally, and calls to increase food waste diversion are growing. Therefore, more research is valuable, especially in U.S. settings.

2. Materials and Methods

2.1. Scope, Functional Unit, Boundaries and Assumptions

The Town of Brookhaven, a suburban New York municipality of 672 km^2 approximately 100 km east of New York City, was used as a case study. The Town provides residential collection services through municipally-negotiated contracts with private carters to 115,315 households (single-, two-, and three-family houses). There is separate collection for paper and container recyclables, yard waste, and residual waste, resulting in 32% diversion from disposal. The residual wastes are collected curbside twice a week by packer trucks, transported to the Town's transfer station for repacking, and then transported by tractor-trailers to the Town of Hempstead WTE plant (Greene et al., 2012).

The functional unit was one tonne of Brookhaven residential residual MSW collected curbside, with a 100 year emissions time frame. The functional unit excludes wastes that have been separated for recycling and yard waste composting, and those deposited at drop off locations, assumed to be identical in all scenarios and thus mutually excluding (Grosso et al., 2012). A consequential LCA approach was used. Scenarios included system expansions to account for changes outside the waste system, such as the substitution of waste derived energy for fossil fuel energy. All environmental emissions upstream from waste collection, including product manufacture, distribution, and use, were omitted (a "zero burden" LCA) (Table S2) (Gentil et al., 2010).

It was assumed that household food waste source separation efficiency was 70%. It is possible that food waste would be commingled with the source separated yard waste currently collected for composting. However, because the functional unit excluded yard waste, any impacts on recovery processes from commingling food and yard wastes were not addressed. The study was performed in accordance with the International Organization for Standardization (ISO) LCA standard 14044 (2006) (ISO, 2006).

2.2. Modeling Approach

Four food waste treatment scenarios were modeled using EASETECH (Table 1) (Clavreul et al., 2014). Figure 1 outlines the modeled processes. The technological systems modeled were available in the EASETECH database, and were adjusted to the U.S. case. AD and food waste composting, although not widespread in the U.S., are potential alternative technologies for food waste because they have been applied broadly and successfully to other organic wastes. There is a proposal to construct an AD facility near the Brookhaven transfer station; AD plants, especially to treat animal wastes, are becoming more common in the U.S., with biogas being an environmentally desirable fuel (Gomez-Brandon and Podmirseg, 2013). Although there are not any food waste composting plants in the general New York metro region, 7% of 3,285 U.S. composting facilities accept food scraps (Platt et al., 2014). Therefore AD and composting were modeled as alternatives to WTE (Table 1). Co-processing food wastes at sewage sludge AD plants was not modeled to avoid functional unit complications. The assessment only considered enclosed composting facilities due to odor and vector issues in a densely populated suburban setting. Although landfilling is the primary disposal option for residual waste in the U.S. (USEPA, 2015), it was not modeled because landfilling MSW was banned on Long Island as of 1991 to protect its sole source aquifer system. Over half of residual waste on Long Island is treated by WTE (the remainder is shipped to off-Long Island landfills) (Greene et al., 2010).

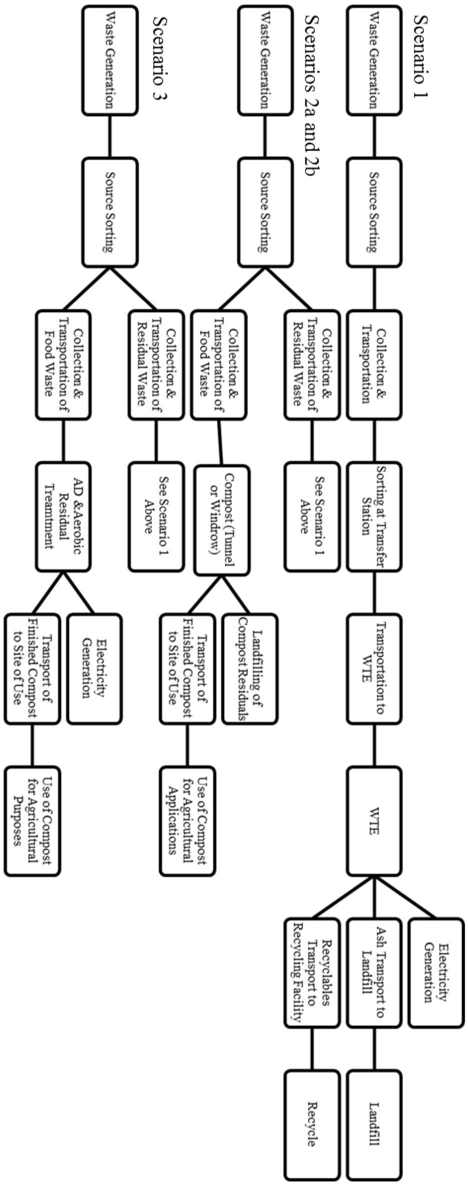

Figure 1. Scenario Outline

Table 1. Scenarios.

Number	Name	Description
1	Waste-to-Energy Incineration (WTE) Disposal	Business as Usual: Current waste management system for Brookhaven. No food waste separation or recovery is performed. All food waste is commingled with residual waste and disposed at a WTE incinerator.
2a	WTE and Enclosed Tunnel Composting	Food waste is composted with an enclosed tunnel composting system (all other residual waste is sent to WTE). Compost is produced by aerobic biodegradation. The compost is applied to facilitate plant growth or soil improvement in agricultural contexts.
2b	WTE and Enclosed Windrow Composting	Food waste is composted with an enclosed windrow system (all other residual waste is sent to WTE). Compost is produced by aerobic biodegradation. The compost is applied to facilitate plant growth or soil improvement in agricultural contexts.
3	WTE and Anaerobic Digestion (AD)	Food waste is digested by AD (all other residual waste is sent to WTE). Biogas is produced by hydrolysis, acid fermentation, and methane fermentation. It is used to generate electricity. Digestate is composted aerobically and the final compost is applied to facilitate plant growth or soil improvement in agricultural contexts.

The waste composition of the modeled residual waste was based on the arithmetic mean of data from a 2012 Brookhaven waste characterization study of three of the Town waste districts (Aphale et al., 2015). Food waste was 13.4% of the residuals. Animal waste was assumed to make up one-third of the total food waste, and vegetable-derived waste the remainder (WRAP, 2013). Specific waste inputs are given in the Supplementary Materials.

2.3. Inventory and Impact Assessment, Sensitivity Analysis

An inventory of elementary exchanges associated with the functional unit was determined and these exchanges were classified and characterized into impact categories. The International Reference Life Cycle Data System (ILCD) approach (2013), the method recommended in EASETECH, was used for impact assessment (ECJRC, 2010). Seven impact categories were used to ensure consideration of multiple types of environmental burdens. They were: climate change (GW); stratospheric ozone depletion (ODP); terrestrial acidification (TA); terrestrial eutrophication (TE); freshwater eutrophication (FE); marine eutrophication (ME); and depletion of fossil resources (ARF) (details provided in Supplementary Materials). The marginal unit of electricity used by the waste treatment facilities and the electricity displaced by waste-derived electricity was assumed to come from a mixture of natural gas (81%), coal (8%), and oil (11%), in accordance with the marginal fuel sources for the northeast U.S. (Siler-Evans et al., 2012).

After impact assessment, the results can be normalized by comparing outputs to a given reference, typically a regional value. Here focus was on the relative impacts of each scenario to another, so normalization was not a major priority. However, normalization to person equivalents was performed to enable comparisons across impact categories. EASTETECH's default normalization approach was used because it was developed specifically for the ILCD 2013 impact assessment method used here (Blok et al., 2013) (normalization values are provided in Supplementary Materials). EASETECH normalization factors are based on global and European emission references, and values for Brookhaven could be somewhat different. However, the normalization values allow for relative comparisons across impact categories and construction of an aggregate score for each scenario. The normalized impact category was also weighted based on perceptions of local public concerns to see how that affected the analysis.

A sensitivity analysis was performed in which input parameters were varied across a range of possible values (Table S7), including food waste sorting efficiency, transport distances to facilities, and differences in the marginal energy profile.

3. Results

Impacts associated with climate change, terrestrial eutrophication, and marine eutrophication were positive in all scenarios, indicating environmental burdens, while ozone depletion, freshwater eutrophication, terrestrial acidification, and resource depletion scores were negative, indicating avoided impacts (savings) (Table 2). Net savings were observed for these categories because of the inclusion of indirect impacts resulting from the substitution of materials outside the waste management system (e.g., electricity, fertilizers). Because the whole residual waste stream was modeled, nearly all of the waste was treated similarly in the different scenarios (through WTE), so that variation only resulted from food waste (less than 13.4% of the modeled waste), and with only 70% of the food waste being diverted. So, the relative difference between scenarios was small.

Table 2. Modeled environmental impacts (treatment of one tonne residual waste).

Scenario [a, b c]	**GW** (kg CO_2 eq.)	**ODP** (kg CFC-11eq.)	**TA** (AE)	**TE** (AE)	**FE** (kg P eq.)	**ME** (kg N eq.)	**ARF** (MJ)
1	185	-0.0000026	-0.61	2.40	-0.000035	0.22	-911
2a	204	-0.0000026	-0.62	2.23	-0.0072	0.29	-899
2b	206	-0.0000026	-0.61	2.23	-0.0072	0.32	-885
3	185	-0.0000026	-0.67	2.09	-0.0075	0.28	-949

[a] A negative value indicates impact saving/emission reduction

[b] AE: accumulated exceedance; GW: climate change; ODP: stratospheric ozone depletion; TA: terrestrial acidification; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication; ARF: depletion of fossil resources

[c] Scenario 1 = WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow composting and WTE; scenario 3 = AD and WTE

The scenarios were ranked with a score of one indicating the best environmental performance. Ties were ranked as the average of the ranks that they would have otherwise occupied, and a mean rank was determined (similar to Diggelman and Ham, 2003) (Table 3). This provides a measure of environmental performance relative to the WTE business as usual scenario. This approach is better for system planning, as decision making based on the relative performance of alternative policy scenarios under a range of scenarios is preferred rather than on a single modeled scenario with absolute outputs (Plevin et al., 2014). The AD scenario performed best (or tied for best) in all impact categories except marine eutrophication. Generally, the baseline (WTE) and tunnel composting scenarios performed better than the windrow composting scenario. Although the baseline scenario performed better than at least one of the alternative scenarios in three impact categories (climate change, marine eutrophication, depletion of fossil resources), alternatives to the business as usual scenario appear capable of providing relative environmental benefit in four of the modeled categories.

Table 3. Environmental impact rankings.

Scenario [a, b]	GW	ODP	TA	TE	FE	ME	ARF	Average Ranking
1	1.5	2.5	3.5	4	4	1	2	**2.6**
2a	3	2.5	2	2.5	2.5	3	3	**2.6**
2b	4	2.5	3.5	2.5	2.5	4	4	**3.3**
3	1.5	2.5	1	1	1	2	1	**1.4**

[a] AE: accumulated exceedance; GW: climate change; ODP: stratospheric ozone depletion; TA: terrestrial acidification; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication; ARF: depletion of fossil resources

[b] Scenario 1= WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow composting and WTE; scenario 3 = AD and WTE

3.1. Process-Specific Impacts

The contribution of each waste system process was assessed for each impact category (see the Supplementary Materials Figures S1-S7). Generally, collection and transport contributed relatively moderately to the life cycle impacts in all impact categories. Fuel consumption during collection and transportation yielded NO_x and SO_x emissions, which affected terrestrial eutrophication and acidification, and marine eutrophication. Fuel use also contributed to depletion of fossil resources and climate change emissions.

WTE had mixed results. Environmental burdens in climate change occurred due to stack emissions (primarily of CO_2), which were partially offset by waste-derived energy substituting for fossil fuels. Burdens were observed in marine and terrestrial eutrophication, primarily due to NO_x emissions, with slight offsets due to waste substituting for fossil fuels. Savings were derived for terrestrial acidification savings due to SO_2 and NO_x offsets from replaced fossil fuel use. Savings also were observed in freshwater eutrophication due to reductions in phosphate emissions.

Recycling impacts occurred from the recovery of scrap aluminum and steel from WTE ash, with savings observed for climate change, primarily due to CO_2 reductions from offsets of virgin material use. Minimal stratospheric ozone depletion savings were observed due to CFC-11 savings, but these impacts were small and carry little importance.

Landfilling WTE residuals had small burdens across all categories; the effects were small because of the mass reduction associated with WTE, and because WTE ash is inert since organic matter is consumed, resulting in no methane or CO_2 generation in the landfill (Papageorgiou et al., 2009).

Burdens from AD and composting operations were small because only 70% of the total amount of food waste was involved. Food waste made up 13.4% of the total MSW stream; if 70% of this food waste was source separated and treated differently than residual waste, 93.3 kg. of food waste was subject to the alternative treatment and therefore, treated differently across scenarios. Savings accruing from compost use were also minimal. However, relative differences for the three alternative treatments for these 93.3 kg compared to WTE provided the differences among the ratings of the scenarios, so these small absolute differences are relatively important (Table 4).

Table 4. Modeled environmental impacts (treatment of 93.3 kg. of residual food waste).

Scenario [a, b c]	**GW** (kg CO_2 eq.)	**ODP** (kg CFC-11eq.)	**TA** (AE)	**TE** (AE)	**FE** (kg P eq.)	**ME** (kg N eq.)	**ARF** (MJ)
1	-12.5	-2.1×10^{-8}	0.03	0.30	-1.2×10^{-6}	0.0029	-9.21
2a	8.59	4.6×10^{-10}	0.04	0.16	3.0×10^{-7}	0.0025	7.09
2b	12.9	8.3×10^{-10}	0.05	0.16	7.3×10^{-7}	0.0039	13.31
3	-9.25	-6.3×10^{-9}	-0.023	-0.014	-1.6×10^{-6}	-0.0013	-73.9

[a] A negative value indicates impact saving/emission reduction

[b] AE: accumulated exceedance; GW: climate change; ODP: stratospheric ozone depletion; TA: terrestrial acidification; TE: terrestrial eutrophication; FE: freshwater eutrophication; ME: marine eutrophication; ARF: depletion of fossil resources

[c] Scenario 1 = WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow composting and WTE; scenario 3 = AD and WTE

Table 4 provides impacts for the alternative treatment of source separated food waste (93.3 kg. of food waste resulting from the source separation of 70% of the total food waste in the 1,000 kg. total MSW); the impacts of treating this waste with WTE were also provided for comparison. This table only indicates results from the waste treatment processes (WTE, AD, composting), not other system components (e.g., transport). Composting operations yielded net climate change burdens rather than benefits because composting requires energy expenditures but generates no electricity (echoing findings in Khoo et al. 2010 and Morris et al. 2014). N_2O and CO_2 emissions, partially from energy consumption, drove composting climate change burdens. C and N compound emissions were reduced with indoor composting due to assumed biofilter usage (the same filter efficiencies were assumed for both composting scenarios). Emissions of SO_2, NO_x, and NH_3 from daily operations (e.g., electricity requirements of facilities) and fugitive emissions which escaped through the biofilter, contributed to the terrestrial acidification, terrestrial eutrophication, and marine eutrophication burdens. Electricity use and the operation of mechanical equipment in the composting facilities caused depletion of fossil resources. The differences between the two composting technologies largely resulted from the one third lower electricity requirements for tunnel composting.

For AD, the greatest savings for climate change provided net benefits in all impact categories, due to the replacement of fossil fuel energy by AD-generated energy (savings resulted primarily from CO_2 offsets). Although environmental emissions from AD were reduced due to a biofilter, some fugitive emissions and facility operation emissions occurred. However, direct emissions of NO_x, NH_3, SO_2, and CH_4 emissions from AD were entirely offset by the replacement of fossil fuels, which also led to savings in the depletion of fossil resources category.

Compost use, comprised of land application, fertilizer substitution, and soil C and N sequestration for compost and composted AD residuals, yielded benefits in four impact categories but not ozone depletion, marine eutrophication, and depletion of fossil resources (Table S6). Burdens resulted from the use of a diesel manure spreader, but were relatively small compared to other aspects of the LCA. Savings resulted from attributed carbon sequestration in soils from compost use and substituting compost nutrient inputs that displace commercial fertilizers. It is not surprising that the two compost scenarios rank better than AD for compost use in all impact categories, as only AD compost residuals are composted, and AD compost is of lower quality because AD consumes organic matter during the digestion phase to create energy gases (Andersen et al., 2012).

Composting offers additional benefits that are difficult to quantify through LCA, including weed suppression, increased soil productivity, and water conservation. The LCA literature does not currently have an impact category directly assessing soil quality and productivity, although soil carbon sequestration and synthetic fertilizer displacement are typically included (as they were here) (Morris et al., 2014). It is necessary to qualitatively recognize the additional benefits of compost to soils when examining composting options, and future efforts to formally quantify them are necessary to improve the performance of composting relative to other technologies.

3.2. Normalized Environmental Impacts

The impact category with the highest normalized effects under all scenarios was marine eutrophication (ME); climate change (GW) and terrestrial eutrophication (TE) also had high impacts (Figure 2). These categories showed the greatest differences across scenarios. The smallest differences across scenarios occurred for ozone depletion (ODP). Fossil resource depletion (ARF) showed the highest normalized impact reductions. Overall, all scenarios had higher environmental burdens than savings, as indicated by the aggregated total of normalized impacts. The concept of person equivalents essentially gives each impact category the same importance. If this is reasonable, then the overall burden from AD was about 0.01 normalized impact factors less than the tunnel composting scenario, which in turn was about 0.003 impact factors less than the baseline scenario, which was again slightly less than the windrow composting scenario (Figure 2).

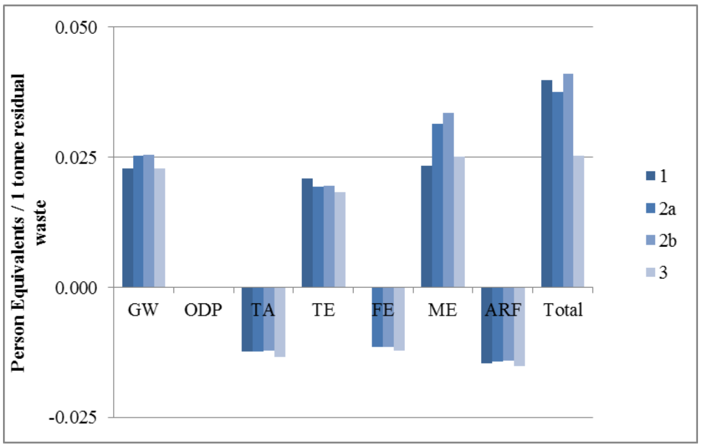

Figure 2. Normalized impact profiles.

3.3. Sensitivy Analysis

A sensitivity analysis was performed to examine the effects of altering several input parameters on climate change (see SI Tables S6 and S7); this impact category was selected because it is of particular interest in the waste management field (Vergara et al., 2011) and it had one of the highest normalized impacts. Sorting efficiency represented a major source of uncertainty (see Yoshida et al. 2012, where capture efficiency was key for modeling greenhouse gas emissions from several organic waste management options). Waste sorts we have reviewed (see Thyberg et al., 2015) indicate that even with robust recycling programs, considerable amounts of targeted recyclables remain in disposal streams. For Brookhaven, up to one-third of discarded residual waste is recyclable, and overall capture efficiencies range from one-quarter to one-half (Aphale et al., 2015). It may be that a 70% separation efficiency is optimistic. In any case, because the baseline scenario of all WTE for residuals had the lowest climate change burden, increased sorting efficiencies for food waste non-intuitively increased climate change burdens. However, these increases were not substantial and they did not change the rank ordering of scenarios.

Transportation and collection are the most commonly tested parameters for sensitivity assessments in waste LCAs, although several studies have shown that impacts of waste transport rarely has a large influence on overall system environmental impacts (Laurent et al., 2014; Grosso et al., 2012). Distance from facilities is an issue in recent food waste legislation, in that several New England regulations base diversion requirements on the distance waste generators are from available treatment facilities. Climate change burden increased with increased distance from treatment facilities, but not substantially (relative to overall system impacts). An increase of approximately three kg CO_2-eq. per tonne of waste managed was observed for all three alternative scenarios as distance increased from 11 to 400 km; similar findings hold when increasing the distance from management facility to the compost use site. The rank ordering of scenarios did not change. Although the relative effects are not great, cumulative impacts from transportation with regard to thousands or millions of tonnes of food waste could be substantial.

Here waste-derived energy was substituted for energy from other sources. Others have found the exact manner in which this substitution is quantified can be important, especially relating to climate change impacts (Bernstad et al., 2012). Changing from northeast to mid-Atlantic marginal energy mixes made a considerable difference in climate change effects (Table S8). Northeast energy is dominated by natural gas, a relatively clean fossil fuel; the mid-Atlantic relies primarily on hard coal, which has more climate change impacts. Each scenario switched from having climate change impacts to having climate change benefits under mid-Atlantic marginal energy, although the relative ranking of the scenarios did not change (considering only that climate change impacts altered). The relative difference between AD and the other scenarios might increase when using another marginal energy mix (more dependent on coal). Across the U.S., marginal CO_2 emissions vary from 486 kg/MWh (west) to 834 kg/MWh (midwest), SO_2 emissions vary from 0.2 kg/MWh (west) to 3.3 kg/MWh (mid-Atlantic), and NO_x emissions from 0.32 kg/MWh (west) to 1.07 kg/MWh (midwest) (Siler-Evans et al., 2012). Waste derived energy will show high impact savings when substituting for marginal energy in regions with high emissions; if it substitutes for renewable, non-polluting energy sources, perceived benefits are reduced. The benefits of waste derived energy substituting for fossil energy are likely to decrease in the future as more energy is created from cleaner, non-fossil sources. In addition, there is much talk of a changing residual waste composition due to the loss of paper in the waste stream, increased use of plastics, and the potential for loss of organics in the disposal waste stream, all of which will decrease non-fossil fuel waste energy benefits. Thus, the impact assessment of alternative food waste treatment will differ by location and will likely change over time.

4. Discussion

The best management approach for food waste can be selected in two ways: through rankings (Table 3), or using the aggregated totals of normalized effects (Figure 2). Both results indicated that diverting food waste from WTE to AD reduced environmental burdens, and the AD scenario performed the best relative to the other scenarios. In the aggregated total approach, the tunnel composting scenario performed marginally better than the WTE scenario. The windrow composting scenario performed the worst. The ranking approach showed WTE and tunnel composting being equivalent in impact, with windrow composting worse. Some important aspects of compost use (weed suppression, increased soil productivity, water conservation) are not included in EASETECH and in LCAs generally (Buzby et al., 2011), and so overall benefits of composting are likely underestimated. Additionally, in this iteration toxicity indicators were not included. Generally, other waste LCAs have determined that AD and composting have fewer potential impacts on human toxicity, human carcinogenicity, human respiratory effects, and ecotoxicity than WTE (Morris et al., 2013). Therefore, the benefits of AD and composting are likely to be even more underestimated relative to WTE.

Diverting food waste to AD in Brookhaven provides the greatest potential for environmental benefit. It is not clear if the un-included factors for composting choices would outweigh the considerable advantage from energy offsets that accrue due to the business as usual WTE option. The difference in the impact factors we examined in the LCA tended not to be too great; marine eutrophication was the only impact category where any of the scenarios were as much as 0.01 impact factors different from each other. So toxicity factors and the unaccounted for compost benefits would need to score very high to change the order of scenarios as depicted here.

All scenarios yielded greater environmental burdens than savings. This suggests that the best way to improve environmental performance and contribute to global sustainability is through waste prevention. Waste prevention also eliminates upstream impacts of food production (Hamilton et al., 2015). This can be compared to more traditional recycling efforts, which generally are found to create net environmental benefits. This suggests that if funds are limited, trying to energize Brookhaven citizens to recover more paper and containers might be a better expenditure of public monies, because it would create environmental benefits rather than burdens. However, overall system burdens could be reduced by adopting AD; furthermore, trying to increase recycling while also diverting food wastes to AD would reduce the overall impact of managing wastes in the Town of Brookhaven.

Although it is unlikely that the Town would switch to landfilling MSW instead of incineration in the future, it is interesting to think about how such a switch would be affected by alternative food waste treatment technologies. If the Town landfilled its wastes, the impacts of a switch to alternative food waste treatment would be greater. Landfilling is almost always found to have more environmental burden than WTE (due to methane emissions) (Guereca et al., 2006; Lee et al., 2007), and since food waste degrades more thoroughly and quickly than other organic wastes, its removal from a landfill would result in much lower environmental burden for the system as a whole (Morris et al., 2014; Bernstad and Jansen, 2012).

Using rankings to determine the best management approach ignores the scale of differences among the choices. However, using the aggregated totals to determine the better management choices means relying on the many assumptions used to generate the aggregation process, and further assumptions regarding the relative importance of each impact category. Adding quantitative sophistication to the decision process does not ensure better decision-making (Plevin et al., 2014), although a comparison of more refined data appears to have more certitude.

4.1. Weighting Results

A rough weighting of the impact categories was also made using our perceptions of the relative importance of the seven impact categories to the local environment (Table S9). Weighting criteria included the level of public awareness of the impact category, as well as their emphasis in local environmental legislation. Weighted impacts appeared to have less of an environmental burden, and reduced the relative normalized difference between scenarios. Weighting also caused the WTE baseline scenario to perform better than the tunnel composting scenario; the windrow composting scenario still performed the worst (Figure 3). It is recognized that LCA weighting is controversial because it is subjective, yet has the ability to greatly influence study results and conclusions. This rough weighting approach was performed to provide a general indication of weighted impacts, but a more formal panel approach may be undertaken in the future. By eliciting participation and feedback from a diverse panel of expertise (such as waste managers, stakeholders, general public, and partners in the waste field), the panel could substantiate the weights, thus refining the LCA results.

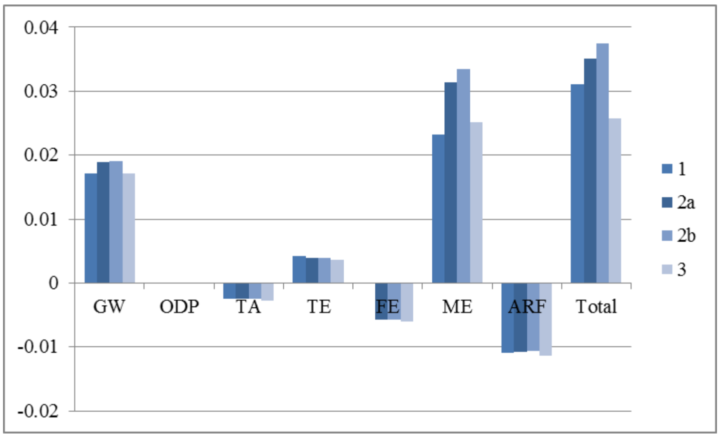

Figure 3. Weighted normalized impact profiles.

4.2. Limitations

Although LCA is useful as a decision support tool for policy development because it can indicate the technologies with fewest environmental burdens, the subjective nature of outputs to modeling choices and the inability to account for social and economic factors limit its utility. Factors important to decision-making for sustainable waste management, such as local environmental impacts (e.g., odor, noise), working environment factors (e.g., safety), investment costs, maintenance costs, and stakeholder concern are generally not included in LCAs (Thyberg and Tonjes, 2015). Political goals (e.g., resource recovery, reduced emissions, energy recovery) will also affect which technological option appears to be the most beneficial, although these can be accounted for through factor weighting. Cost is always an issue; separate management of food wastes will require extra collection effort, and most likely higher disposal fees. So, it is clear that selecting the most sustainable waste management practices requires additional information and evaluation besides that presented by traditional LCAs. The inability of LCAs to account for important parameters other than environmental impacts make them too one-dimensional to be used as a sole means to select sustainable waste treatments (Morris et al., 2014). Therefore, LCAs are important elements for sustainable decision-making, but they should be used in conjunction with other tools (e.g., social LCA, life cycle cost evaluations) (Thyberg and Tonjes, 2015). An area of future research includes capturing these other factors in our analyses.

It is difficult to make a direct comparison of waste treatment alternatives across LCA studies (Lundie and Peters, 2005; Bernstad and Jansen, 2012), so that findings tend to be case specific (Vandermeersch et al., 2014). This is due to functional units often being not equivalent and differences in modeling assumptions, impact categories, technologies being assessed, and geographical settings. Our findings appear reasonable for the Town of Brookhaven, although they may not hold elsewhere.

Food waste prevention was not included as an option. Only limited prior LCA work has included waste prevention (e.g., Oldfield and Holden, 2016; Schott and Andersson, 2015, Gentil et al., 2010, Hamilton et al., 2015). The existing quantitative work that has been conducted on food waste prevention indicates it results in the greatest impact reductions, primarily from avoided food production (Gentil et al., 2010); prevention also achieves certain economic and social benefits (Thyberg and Tonjes, 2015). Technically, prevention alters the functional unit, thus making it challenging to compare results between scenarios. Waste prevention can liberate treatment capacity at disposal facilities. For WTE, this can result in higher energy value in residual waste due to lesser food waste. These effects are not typically accounted for in waste LCAs. Upstream impacts, such as those from agricultural and industrial food production, may be substantial, and their inclusion is necessary for analysis of waste prevention effects (Oldfield and Holden, 2014).

5. Conclusion

A LCA of the environmental impacts of four waste system scenarios was conducted for the Town of Brookhaven, New York, to determine the effect of changes in food waste treatment. This allowed for the inclusion of local specifics in the model, such as waste composition and transport distances, and provided insight into potential improvements for the current system. The objective of the study was to evaluate the environmental impacts of U.S. residential waste disposal in a suburban municipality to determine if environmental improvement could be achieved by adopting separate food waste recovery and treatment. Results indicated that overall environmental burdens can be reduced by source separating food waste and treating it by AD, and then composting the AD residuals, or treating it with tunnel composting. Results also indicated, however, that in some impact categories, the business as usual scenario (WTE of residuals including food wastes) is a better choice from an environmental perspective. Sensitivity analysis found marginal energy portfolios have considerable effects on the size of impacts.

These findings can be used to inform decision making focused on sustainable waste management in the U.S. Although our findings are, strictly speaking, limited to the location and technologies we studied, our results suggest that food waste diversion may be considerably more beneficial in other regions, particularly those that landfill wastes and burn coal to make electricity. Shifting to waste treatment technologies that minimize the environmental impacts of waste systems can contribute to more sustainable waste management practices, and the use of LCAs to identify those more advantageous approaches can be beneficial. However, we do recognize that LCAs can sometimes overcomplicate environmental impact studies by presenting a plethora of impact categories, and also oversimplify effects when results are reduced to single values. In the latter situation, care must be taken to assign weightings to categories that fit local conditions, as well as social and policy goals.

So, in order to increase the sustainability of waste systems, other factors that influence decisions, including economic costs, social priorities, and stakeholder concerns, should also be considered. Because our analysis was conducted on the entire waste stream, results can be compared to the system-wide economic effects of changes in food waste management, as well as the broader social and policy impacts of addressing food waste disposal issues. Because previous food waste LCAs look only at food waste, it is difficult to integrate their findings into system-wide economic and stakeholder analyses.

In conclusion, food waste must be responsibly managed for societies to be sustainable. Key aspects of sustainable food waste strategies will include food waste prevention policies, as well as its treatment with the most environmentally sound technologies. This study indicated that treating food waste with certain technologies will provide greater environmental impact reductions than others. Sustainable food waste management will become even more important over time as populations grow, and urbanization, economic growth, and globalization lead to differing food waste generation and disposal trends.

Supplementary Materials: The supplementary material (SM) describes the Life Cycle Assessment (LCA) case study. Section 1 describes previous LCA work focused on food waste. Sections 2 and 3 further describe the model and the case study. Sections 4-6 expand on the results presented in the main section of the paper. Specifically, the following are available online: Table S1. LCAs Focused on Food Waste; Table S2. LCA Boundaries; Table S3. Material Characteristics of Waste Fractions; Table S4. Environmental Impact Categories Included in LCA; Table S5. Process Groups in the LCA; Table S6. Compost Use Process Impacts; Table S7. Sensitivity Analyses; Table S8. Marginal Energy Sensitivity Analysis Results; Table S9. Weighting Criteria; Figure S1. Climate Change (GW) - Process Specific Impacts; Figure S2. Stratospheric Ozone Depletion (ODP) - Process Specific Impacts; Figure S3. Terrestrial Acidification (TA) - Process Specific Impacts; Figure S4. Terrestrial Eutrophication (TE) - Process Specific Impacts; Figure S5. Freshwater Eutrophication (FE) - Process Specific Impacts; Figure S6. Marine Eutrophication (ME) - Process Specific Impacts; and, Figure S7. Depletion of Fossil Resources (ARF) - Process Specific Impacts.

Acknowledgments: KLT was supported by the Town of Brookhaven under a Professional Services Agreement; DJT received some support from the Town similarly. Although the Town of Brookhaven supported this research, it does not necessarily reflect the view of the Town and no official endorsement should be inferred. The Town makes no warranties or representations as to the usability or suitability of the materials and the Town shall be under no liability whatsoever for any use made thereon. We would like to acknowledge Anders Damgaard of the Technical University of Denmark for his assistance with developing the modeling approach for the study. Comments made by three anonymous reviewers improved this work appreciably.

Author Contributions: KLT conceived and designed the study, performed analyses, and evaluated results. DJT assisted with study design, and the evaluation and discussion of results.

Conflicts of Interest: The authors declare no conflict of interest.

References

Andersen, J. K., A. Boldrin, T. H. Christensen, and C. Scheutz. Home composting as an alternative treatment option for organic household waste in Denmark: An environmental assessment using life cycle assessment modelling. *Waste Management.* **2012,** *32* (1), 31-40.

Aphale, O., K. L. Thyberg, and D. J. Tonjes.. Differences in waste generation, waste composition, and source separation across three waste districts in a New York suburb. *Resources, Conservation and Recycling.* **2015,** *99*, 19-28.

Bernstad, A., and J. L. Jansen. Review of comparative LCAs of food waste management systems - Current status and potential improvements. *Waste Management***.** **2012,** *32* (12), 2439-2455.

Blok, K., M. Huijbregts, L. Roes, B. van Haaster, M. Patel, E. Hertwich, and M. Harmelink. Development and application of a standardized methodology for the PROspective SUstaInability assessment of TEchnologies (PROSUITE). Report prepared within the EC 7th Framework Project, n: 227078, European Commission CORDIS, Netherlands. **2013.** Available online: http://cordis.europa.eu/project/rcn/92592_en.html.

Buzby, J. C., J. Hyman, H. Stewart, and H. F. Wells. The value of retail- and consumer-level fruit and vegetable losses in the United States. *Journal of Consumer Affairs*. **2011,** *45* (3), 492-515.

Buzby, J. C., H. F. Wells, and J. Hyman. The Estimated Amount, Value and Calories of Postharvest Food Losses at the Retail and Consumer Levels in the United States. USDA Economic Research Service, Washington, D.C., USA. **2014.** Available online: http://www.ers.usda.gov/media/1282296/eib121.pdf.

Clavreul, J. H. Baumeister, T.H. Christensen, and A. Damgaard. An environmental assessment system for environmental technologies. *Environmental Modelling and Software*. **2014,** *60* (October 2014), 18-30.

Diggelman, C., and R. K. Ham. Household food waste to wastewater or to solid waste? That is the question. *Waste Management & Research*. **2003,** *21* (6), 501-514.

ECJRC (European Commission Joint Research Centre). Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment. Institute for Environment and Sustainability, Ispra, Italy. **2010.** Available online: http://eplca.jrc.ec.europa.eu/uploads/ILCD-Handbook-LCIA-Background-analysis-online-12March2010.pdf.

Gentil, E. C., A. Damgaard, M. Hauschild, G. Finnveden, O. Eriksson, S. Thorneloe, P. O. Kaplan, M. Barlaz, O. Muller, Y. Matsui, R. Ii, and T. H. Christensen. Models for waste life cycle assessment: Review of technical assumptions. *Waste Management*. **2010,** *30,* (12): 2636-2648.

Gomez-Brandon, M., and S. M. Podmirseg. 2013. Biological waste treatment. *Waste Management & Research* . **2012**, *31* (8), 773-774.

Greene, K., O. Aphale, G. Ntshalintshali, P. Nayak, L. Swanson, and D. Tonjes. Recycling on Long Island 2009. Waste Reduction and Management Institute, Stony Brook University, Stony Brook, NY, USA. **2011.** Available online: http://www.stonybrook.edu/est/research/FINAL%20Recycling%20Report%20(4-16-11)%20PDF.pdf.

Grosso, M., C. Nava, R. Testori, L. Rigamonti, and F. Vigano. The implementation of anaerobic digestion of food waste in a highly populated urban area: an LCA evaluation. *Waste Management & Research.* **2012**, *30* (9), 78-87.

Guereca, L. P., S. Gasso, J. M. Baldasano, and P. Jimenez-Guerrero. Life cycle assessment of two biowaste management systems for Barcelona, Spain. *Resources Conservation and Recycling*. **2006,** *49* (1), 32-48.

Hamilton, H. A., M. S. Peverill, D. B. Muller, and H. Brattebo. Assessment of Food Waste Prevention and Recycling Strategies Using a Multilater Systems Approach. **2015,** *49* (24), 13937-13945.

International Organization for Standardization. ISO 14040: Life Cycle Assessment - Principles and Framework. Geneva, Switzerland. **2006.**

Khoo, H. H., T. Z. Lim, and R. B. H. Tan. Food waste conversion options in Singapore: Environmental impacts based on an LCA perspective. *Science of the Total Environment*. **2010**, *408* (6), 1367-1373.

Laurent, A., J. Clavreul, A. Bernstad, I. Bakas, M. Niero, E. Gentile, T. H. Christensen, and M. Z. Hauschild. Review of LCA studies of solid waste management systems - Part II: Methodological guidance for a better practice. *Waste Management*, **2014**, 34 (3), 589-606.

Lee, S. H., K. I. Choi, M. Osako, and J. I. Dong. Evaluation of environmental burdens caused by changes of food waste management systems in Seoul, Korea. Science of the *Total Environment*. **2007**, *387* (1-3), 42-53.

Lundie, S., and G. M. Peters. Life cycle assessment of food waste management options. *Journal of Cleaner Production*. **2005**, *13* (3), 275-286.

Manfredi, S., and R. Pant. Improving the environmental performance of bio-waste management with life cycle thinking (LCT) and life cycle assessment (LCA). *International Journal of Life Cycle Assessment*. **2013,** *18* (1), 285-291.

Morris, J., H. S. Matthews, and C. Morawski. Review and meta-analysis of 82 studies on end-of-life management methods for source separated organics. *Waste Management.* **2013,** *33* (3), 545-551.

Morris, J., S. Brown, H.S. Matthews, and M. Cotton. Evaluation of Climate, Energy, and Soils Impacts of Selected Food Discards Management Systems. State of Oregon Department of Environmental Quality, Portland, OR, USA. **2014.** Available online: http://www.oregon.gov/deq/LQ/Documents/SWdocs/FoodReport.pdf.

Oldfield, T. and N.M. Holden. An environmental analysis of options for utilising wasted food and food residue. *Journal of Environmental Management*. **2016,** *(183)* (3), 826-835.

Oldfield, T. and N.M. Holden. An evaluation of upstream assumptions in food-waste life cycle assessments. in Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector, Schneck, R. and D. Huizenga, editors. San Francisco, CA, USA. **2014.** Available online: http://lcafood2014.org/papers/23.pdf.

Owens, J. W. Why life cycle impact assessment is now described as an indicator system. *The International Journal of Life Cycle Assessment.* **1999,** *4* (2), 81-86.

Papageorgiou, A., J. R. Barton, and A. Karagiannidis. Assessment of the greenhouse effect impact of technologies used for energy recovery from municipal waste: A case for England. *Journal of Environmental Management*. **2009.** *90* (10), 2999-3012.

Platt, B.; Goldstein, N.; Brown, S. State of Composting in the U.S.. Institute for Local Self Reliance, Washington, DC, USA, **2014**. Avaialble online: http://ilsr.org/wp-content/uploads/2014/07/state-of-composting-in-us.pdf.

Plevin, R. J., M. A. Delucchi, and F. Creutzig. Using attributional life cycle assessment to estimate climate-change mitigation benefits misleads policy makers. *Journal of Industrial Ecology*. **2014,** 18 (1), 73-83.

Salhofer, S., G. Obersteiner, F. Schneider, and S. Lebersorger. Potentials for the prevention of municipal solid waste. *Waste Management*. **2008,** *28* (2), 245-259.

Schott, A.B.S. and T. Andersson. Food waste minimization from a life-cycle perspective. *Journal of Environmental Management*. **2015,** *147* (1), 219-226.

Schott, A. B. S., S. Vukicevic, I. Bohn, and T. Andersson. Potentials for food waste minimization and effects on potential biogas production through anaerobic digestion. *Waste Management and Research.* **2013,** *31* (8), 811-819.

Siler-Evans, K., I. L. Azevedo, and M. G. Morgan. Marginal emissions factors for the U.S. electricity system. *Environmental Science & Technology*. **2012,** *46* (9), 4742-4748.

Thyberg, K. L. and D. J. Tonjes. Drivers of Food Wastage and their Implications for Sustainable Policy Development. *Resources Conservation and Recycling.* **2016**, *106*, 110-123.

Thyberg, K.L., D. J. Tonjes, and J. Gurevitch. Quantification of food waste disposal in the United States: A meta-analysis. *Environmental Science and Technology*. **2015**, *49* (24), 13946-13953.

Thyberg, K. L. and D. J. Tonjes. A management framework for municipal solid waste systems and its application to food waste prevention. *Systems.* **2015**, 3, 133-151.

Tonjes, D. J. and R. L. Swanson. Long Island's solid waste perplexities. *Journal of Urban Technologies*. **1994**, 1(3), 21-46.

USEPA. MSW in the United States: 2013 Facts and Figures- Fact Sheet. USEPA, Washington, D.C., USA. **2015.** Available online: https://www.epa.gov/smm/advancing-sustainable-materials-management-facts-and-figures.

Vandermeersch, T., R. A. F. Alvarenga, P. Ragaert, and J. Dewulf. Environmental sustainability assessment of food waste valorization options. *Resources, Conservation and Recycling*. **2014,** *87*, 57-64.

Vergara, S. E., A. Damgaard, and A. Horvath. Boundaries matter: Greenhouse gas emission reductions from alternative waste treatment strategies for California's municipal solid waste. *Resources, Conservation and Recycling*. **2011,** *57*, 87-97.

WRAP. Household Food and Drink Waste in the United Kingdom 2012. WRAP, Banbury, U.K. **2013.** Available online: http://www.wrap.org.uk/content/household-food-and-drink-waste-uk-2012.

Yoshida, H., J. J. Gable, and J. K. Park. Evaluation of organic waste diversion alternatives for greenhouse gas reduction. *Resources, Conservation and Recycling*. **2012,** *60* (0), 1-9.