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

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

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

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40 **1. Introduction**

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

48 Some portion of food waste, even if waste avoidance measures were to be successful,
49 is unavoidable (Schott et al., 2013); reuse opportunities, through redistribution of edible
50 food to humans or animals probably cannot account for the remainder due to perishability
51 and high transport and distribution costs (Buzby et al., 2014), or the excess food may not
52 meet safety or quality requirements (Salhofer et al., 2008). Furthermore, such prevention
53 activities may not appeal to consumers on aesthetic or cultural grounds (Buzby et al.,
54 2011). About 32 million tonnes (MT) of food waste is disposed annually in the U.S., which
55 is 15% of all disposed municipal solid waste (MSW) (Thyberg et al., 2015). Currently
56 waste planners and managers see diversion of this waste from landfills as a means of
57 enhancing stagnant recycling rates, improving environmental conditions associated with
58 waste management, and ultimately contributing to resource conservation and sustainability.
59 Sound analyses of the environmental impacts of specific food waste treatment options
60 would support the development of better and more successful diversion programs.

61 A life cycle assessment (LCA) is a system assessment tool that quantifies potential
62 environmental exchanges and impacts of system processes. Outputs include indicators
63 which simplify and organize inventory results to make them more understandable (Owens,
64 1999). Waste system LCAs quantify impacts of interconnected waste management
65 technologies, from generation to final disposal/treatment based on a specified waste
66 composition, and so allow for comparisons between options (Manfredi and Pant, 2013).
67 Previous food waste LCAs usually only model the food waste portion of the waste stream
68 and exclude impacts from other residual wastes (e.g., Lundie and Peters, 2005; Lee et al.,
69 2007; Andersen et al., 2012). An evaluation of the entire system is required to determine
70 which changes are needed for system improvement. This holistic approach also enables a
71 more complete understanding of the overall system as additional factors can be included in
72 the model, such as the effects of differing levels of source separation of the targeted
73 materials. Modeling all residual waste is important when considering combustion
74 technologies, too, since net energy production will be quite small for studies looking only at
75 food waste due to high moisture content (Morris et al., 2014).

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

83 The objective of this study was to use LCA to evaluate the environmental impacts of
84 U.S. residential waste disposal to determine if environmental improvement can be achieved
85 by adopting separate food waste recovery and treatment in a suburban municipality (Town
86 of Brookhaven, Long Island, New York). Brookhaven currently disposes of collected
87 wastes using waste-to-energy incineration (WTE) and there is no separation of food waste;
88 this was considered the baseline scenario and alternatives to this baseline were evaluated.
89 The findings were used to determine the conditions under which food waste recovery is
90 beneficial, as well as how LCA analyses can be leveraged to effectively inform decision

91 making focused on sustainable waste management. Emphasis was placed on evaluating the
92 full residual waste stream going to disposal (not only food waste), as impacts and benefits
93 are associated with the entire system of managing wastes, not just the food waste portion.
94 When deciding on approaches for waste system improvements, it is essential to consider the
95 system-wide context rather than just looking at the impacts associated with a single waste
96 fraction. Additionally, determinations of exactly how to aggregate impact categories may
97 affect the interpretation of potential system changes.

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

115

116 **2. Materials and Methods**

117 *2.1. Scope, Functional Unit, Boundaries and Assumptions*

118 The Town of Brookhaven, a suburban New York municipality of 672 km²
119 approximately 100 km east of New York City, was used as a case study. The Town
120 provides residential collection services through municipally-negotiated contracts with
121 private carters to 115,315 households (single-, two-, and three-family houses). There is
122 separate collection for paper and container recyclables, yard waste, and residual waste,
123 resulting in 32% diversion from disposal. The residual wastes are collected curbside twice a
124 week by packer trucks, transported to the Town's transfer station for repacking, and then
125 transported by tractor-trailers to the Town of Hempstead WTE plant (Greene et al., 2012).

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

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

141 *2.2. Modeling Approach*

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

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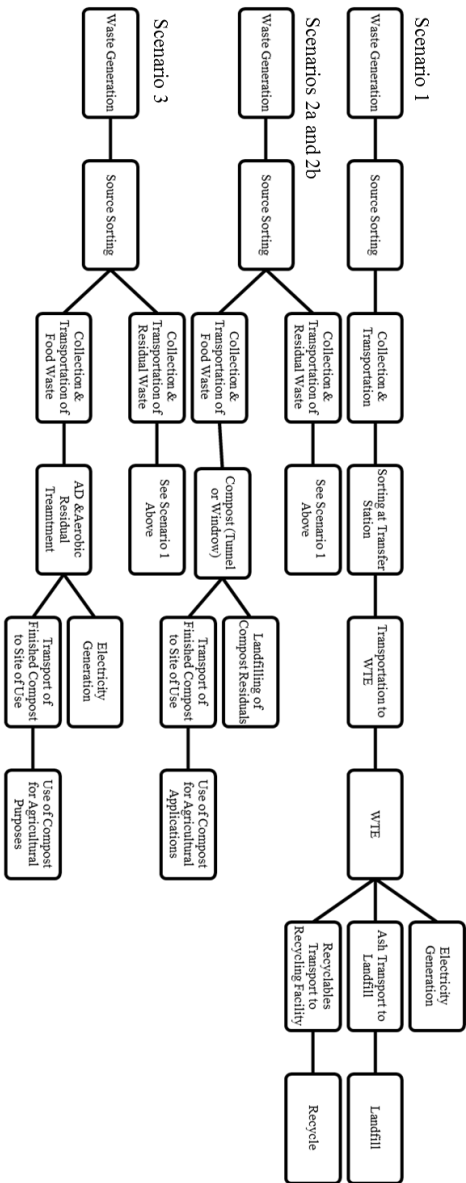


Figure 1. Scenario Outline

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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.

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180 The waste composition of the modeled residual waste was based on the arithmetic mean
 181 of data from a 2012 Brookhaven waste characterization study of three of the Town waste
 182 districts (Aphale et al., 2015). Food waste was 13.4% of the residuals. Animal waste was
 183 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.

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2.3. Inventory and Impact Assessment, Sensitivity Analysis

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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

199 to another, so normalization was not a major priority. However, normalization to person
 200 equivalents was performed to enable comparisons across impact categories. EASTETECH's
 201 default normalization approach was used because it was developed specifically for the ILCD
 202 2013 impact assessment method used here (Blok et al., 2013) (normalization values are
 203 provided in Supplementary Materials). EASETECH normalization factors are based on
 204 global and European emission references, and values for Brookhaven could be somewhat
 205 different. However, the normalization values allow for relative comparisons across impact
 206 categories and construction of an aggregate score for each scenario. The normalized impact
 207 category was also weighted based on perceptions of local public concerns to see how that
 208 affected the analysis.

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

212 3. Results

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

223
 224 **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.)	AR F (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

225 ^a A negative value indicates impact saving/emission reduction

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

229 ^c Scenario 1 = WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow
 230 composting and WTE; scenario 3 = AD and WTE

231

232 The scenarios were ranked with a score of one indicating the best environmental
 233 performance. Ties were ranked as the average of the ranks that they would have otherwise
 234 occupied, and a mean rank was determined (similar to Diggelman and Ham, 2003) (Table
 235 3). This provides a measure of environmental performance relative to the WTE business as
 236 usual scenario. This approach is better for system planning, as decision making based on
 237 the relative performance of alternative policy scenarios under a range of scenarios is
 238 preferred rather than on a single modeled scenario with absolute outputs (Plevin et al.,
 239 2014). The AD scenario performed best (or tied for best) in all impact categories except
 240 marine eutrophication. Generally, the baseline (WTE) and tunnel composting scenarios
 241 performed better than the windrow composting scenario. Although the baseline scenario
 242 performed better than at least one of the alternative scenarios in three impact categories
 243 (climate change, marine eutrophication, depletion of fossil resources), alternatives to the
 244 business as usual scenario appear capable of providing relative environmental benefit in
 245 four of the modeled categories.
 246

247 **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

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

251 ^b Scenario 1= WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow
 252 composting and WTE; scenario 3 = AD and WTE
 253

254 3.1. Process-Specific Impacts

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

261 WTE had mixed results. Environmental burdens in climate change occurred due to
 262 stack emissions (primarily of CO₂), which were partially offset by waste-derived energy
 263 substituting for fossil fuels. Burdens were observed in marine and terrestrial eutrophication,
 264 primarily due to NO_x emissions, with slight offsets due to waste substituting for fossil fuels.
 265 Savings were derived for terrestrial acidification savings due to SO₂ and NO_x offsets from
 266 replaced fossil fuel use. Savings also were observed in freshwater eutrophication due to
 267 reductions in phosphate emissions.

268 Recycling impacts occurred from the recovery of scrap aluminum and steel from WTE
 269 ash, with savings observed for climate change, primarily due to CO₂ reductions from
 270 offsets of virgin material use. Minimal stratospheric ozone depletion savings were observed
 271 due to CFC-11 savings, but these impacts were small and carry little importance.

272 Landfilling WTE residuals had small burdens across all categories; the effects were
 273 small because of the mass reduction associated with WTE, and because WTE ash is inert

274 since organic matter is consumed, resulting in no methane or CO₂ generation in the landfill
 275 (Papageorgiou et al., 2009).

276 Burdens from AD and composting operations were small because only 70% of the total
 277 amount of food waste was involved. Food waste made up 13.4% of the total MSW stream;
 278 if 70% of this food waste was source separated and treated differently than residual waste,
 279 93.3 kg. of food waste was subject to the alternative treatment and therefore, treated
 280 differently across scenarios. Savings accruing from compost use were also minimal.
 281 However, relative differences for the three alternative treatments for these 93.3 kg
 282 compared to WTE provided the differences among the ratings of the scenarios, so these
 283 small absolute differences are relatively important (Table 4).
 284

285 **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 x 10 ⁻⁸	0.03	0.30	-1.2 x 10 ⁻⁶	0.0029	- 9.21
2a	8.59	4.6 x 10 ⁻¹⁰	0.04	0.16	3.0 x 10 ⁻⁷	0.0025	7.09
2b	12.9	8.3 x 10 ⁻¹⁰	0.05	0.16	7.3 x 10 ⁻⁷	0.0039	13.3 1
3	-9.25	-6.3 x 10 ⁻⁹	- 0.023	- 0.014	-1.6 x 10 ⁻⁶	-0.0013	- 73.9

286 ^a A negative value indicates impact saving/emission reduction

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

290 ^c Scenario 1 = WTE; scenario 2a = tunnel composting and WTE; scenario 2b = windrow
 291 composting and WTE; scenario 3 = AD and WTE

292
 293 Table 4 provides impacts for the alternative treatment of source separated food waste
 294 (93.3 kg. of food waste resulting from the source separation of 70% of the total food waste
 295 in the 1,000 kg. total MSW); the impacts of treating this waste with WTE were also
 296 provided for comparison. This table only indicates results from the waste treatment
 297 processes (WTE, AD, composting), not other system components (e.g., transport).
 298 Composting operations yielded net climate change burdens rather than benefits because
 299 composting requires energy expenditures but generates no electricity (echoing findings in
 300 Khoo et al. 2010 and Morris et al. 2014). N₂O and CO₂ emissions, partially from energy
 301 consumption, drove composting climate change burdens. C and N compound emissions
 302 were reduced with indoor composting due to assumed biofilter usage (the same filter
 303 efficiencies were assumed for both composting scenarios). Emissions of SO₂, NO_x, and
 304 NH₃ from daily operations (e.g., electricity requirements of facilities) and fugitive
 305 emissions which escaped through the biofilter, contributed to the terrestrial acidification,
 306 terrestrial eutrophication, and marine eutrophication burdens. Electricity use and the
 307 operation of mechanical equipment in the composting facilities caused depletion of fossil
 308 resources. The differences between the two composting technologies largely resulted from
 309 the one third lower electricity requirements for tunnel composting.

310 For AD, the greatest savings for climate change provided net benefits in all impact
 311 categories, due to the replacement of fossil fuel energy by AD-generated energy (savings
 312 resulted primarily from CO₂ offsets). Although environmental emissions from AD were
 313 reduced due to a biofilter, some fugitive emissions and facility operation emissions
 314 occurred. However, direct emissions of NO_x, NH₃, SO₂, and CH₄ emissions from AD were
 315 entirely offset by the replacement of fossil fuels, which also led to savings in the depletion
 316 of fossil resources category.

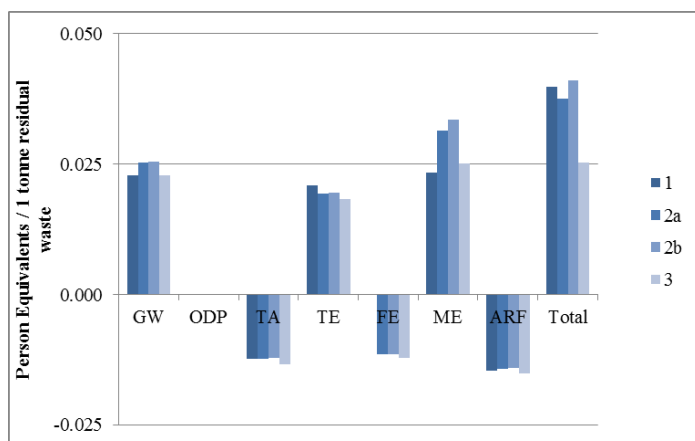
317 Compost use, comprised of land application, fertilizer substitution, and soil C and N
 318 sequestration for compost and composted AD residuals, yielded benefits in four impact
 319 categories but not ozone depletion, marine eutrophication, and depletion of fossil resources
 320 (Table S6). Burdens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

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

335 3.2. Normalized Environmental Impacts

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

347



348

349 **Figure 2.** Normalized impact profiles.

350

351 3.3. Sensitivity Analysis

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

366 Transportation and collection are the most commonly tested parameters for sensitivity
367 assessments in waste LCAs, although several studies have shown that impacts of waste
368 transport rarely has a large influence on overall system environmental impacts (Laurent et
369 al., 2014; Grosso et al., 2012). Distance from facilities is an issue in recent food waste
370 legislation, in that several New England regulations base diversion requirements on the
371 distance waste generators are from available treatment facilities. Climate change burden
372 increased with increased distance from treatment facilities, but not substantially (relative to
373 overall system impacts). An increase of approximately three kg CO₂-eq. per tonne of waste
374 managed was observed for all three alternative scenarios as distance increased from 11 to
375 400 km; similar findings hold when increasing the distance from management facility to the
376 compost use site. The rank ordering of scenarios did not change. Although the relative effects
377 are not great, cumulative impacts from transportation with regard to thousands or millions of
378 tonnes of food waste could be substantial.

379 Here waste-derived energy was substituted for energy from other sources. Others have
380 found the exact manner in which this substitution is quantified can be important, especially
381 relating to climate change impacts (Bernstad et al., 2012). Changing from northeast to mid-
382 Atlantic marginal energy mixes made a considerable difference in climate change effects
383 (Table S8). Northeast energy is dominated by natural gas, a relatively clean fossil fuel; the
384 mid-Atlantic relies primarily on hard coal, which has more climate change impacts. Each
385 scenario switched from having climate change impacts to having climate change benefits
386 under mid-Atlantic marginal energy, although the relative ranking of the scenarios did not
387 change (considering only that climate change impacts altered). The relative difference
388 between AD and the other scenarios might increase when using another marginal energy mix
389 (more dependent on coal). Across the U.S., marginal CO₂ emissions vary from 486 kg/MWh
390 (west) to 834 kg/MWh (midwest), SO₂ emissions vary from 0.2 kg/MWh (west) to 3.3
391 kg/MWh (mid-Atlantic), and NO_x emissions from 0.32 kg/MWh (west) to 1.07 kg/MWh
392 (midwest) (Siler-Evans et al., 2012). Waste derived energy will show high impact savings
393 when substituting for marginal energy in regions with high emissions; if it substitutes for
394 renewable, non-polluting energy sources, perceived benefits are reduced. The benefits of
395 waste derived energy substituting for fossil energy are likely to decrease in the future as more
396 energy is created from cleaner, non-fossil sources. In addition, there is much talk of a
397 changing residual waste composition due to the loss of paper in the waste stream, increased

398 use of plastics, and the potential for loss of organics in the disposal waste stream, all of which
399 will decrease non-fossil fuel waste energy benefits. Thus, the impact assessment of
400 alternative food waste treatment will differ by location and will likely change over time.

401 **4. Discussion**

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

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

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

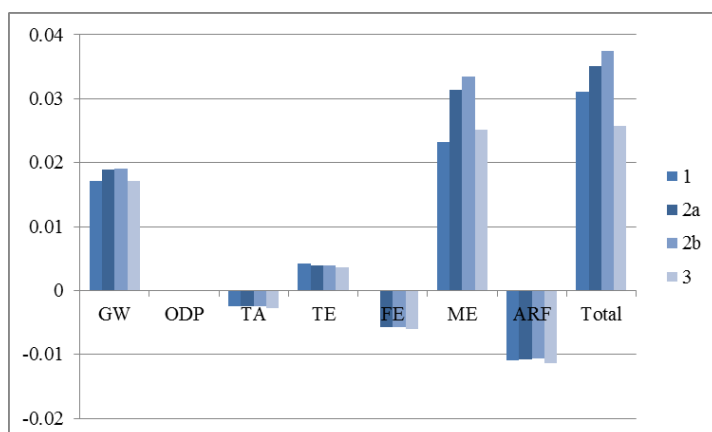
436 Although it is unlikely that the Town would switch to landfilling MSW instead of
437 incineration in the future, it is interesting to think about how such a switch would be
438 affected by alternative food waste treatment technologies. If the Town landfilled its wastes,
439 the impacts of a switch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

445 Using rankings to determine the best management approach ignores the scale of
446 differences among the choices. However, using the aggregated totals to determine the better
447 management choices means relying on the many assumptions used to generate the

448 aggregation process, and further assumptions regarding the relative importance of each
449 impact category. Adding quantitative sophistication to the decision process does not ensure
450 better decision-making (Plevin et al., 2014), although a comparison of more refined data
451 appears to have more certitude.

452 4.1. Weighting Results

453 A rough weighting of the impact categories was also made using our perceptions of the
454 relative importance of the seven impact categories to the local environment (Table S9).
455 Weighting criteria included the level of public awareness of the impact category, as well as
456 their emphasis in local environmental legislation. Weighted impacts appeared to have less
457 of an environmental burden, and reduced the relative normalized difference between
458 scenarios. Weighting also caused the WTE baseline scenario to perform better than the
459 tunnel composting scenario; the windrow composting scenario still performed the worst
460 (Figure 3). It is recognized that LCA weighting is controversial because it is subjective, yet
461 has the ability to greatly influence study results and conclusions. This rough weighting
462 approach was performed to provide a general indication of weighted impacts, but a more
463 formal panel approach may be undertaken in the future. By eliciting participation and
464 feedback from a diverse panel of expertise (such as waste managers, stakeholders, general
465 public, and partners in the waste field), the panel could substantiate the weights, thus
466 refining the LCA results.
467



468
469 **Figure 3.** Weighted normalized impact profiles.
470

471 4.2. Limitations

472 Although LCA is useful as a decision support tool for policy development because it
473 can indicate the technologies with fewest environmental burdens, the subjective nature of
474 outputs to modeling choices and the inability to account for social and economic factors
475 limit its utility. Factors important to decision-making for sustainable waste management,
476 such as local environmental impacts (e.g., odor, noise), working environment factors (e.g.,
477 safety), investment costs, maintenance costs, and stakeholder concern are generally not
478 included in LCAs (Thyberg and Tonjes, 2015). Political goals (e.g., resource recovery,
479 reduced emissions, energy recovery) will also affect which technological option appears to
480 be the most beneficial, although these can be accounted for through factor weighting. Cost
481 is always an issue; separate management of food wastes will require extra collection effort,
482 and most likely higher disposal fees. So, it is clear that selecting the most sustainable waste
483 management practices requires additional information and evaluation besides that presented
484 by traditional LCAs. The inability of LCAs to account for important parameters other than
485 environmental impacts make them too one-dimensional to be used as a sole means to select

486 sustainable waste treatments (Morris et al., 2014). Therefore, LCAs are important elements
487 for sustainable decision-making, but they should be used in conjunction with other tools
488 (e.g., social LCA, life cycle cost evaluations) (Thyberg and Tonjes, 2015). An area of
489 future research includes capturing these other factors in our analyses.

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

496 Food waste prevention was not included as an option. Only limited prior LCA work
497 has included waste prevention (e.g., Oldfield and Holden, 2016; Schott and Andersson,
498 2015, Gentil et al., 2010, Hamilton et al., 2015). The existing quantitative work that has
499 been conducted on food waste prevention indicates it results in the greatest impact
500 reductions, primarily from avoided food production (Gentil et al., 2010); prevention also
501 achieves certain economic and social benefits (Thyberg and Tonjes, 2015). Technically,
502 prevention alters the functional unit, thus making it challenging to compare results between
503 scenarios. Waste prevention can liberate treatment capacity at disposal facilities. For WTE,
504 this can result in higher energy value in residual waste due to lesser food waste. These
505 effects are not typically accounted for in waste LCAs. Upstream impacts, such as those
506 from agricultural and industrial food production, may be substantial, and their inclusion is
507 necessary for analysis of waste prevention effects (Oldfield and Holden, 2014).

508 **5. Conclusion**

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

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

534 So, in order to increase the sustainability of waste systems, other factors that influence
535 decisions, including economic costs, social priorities, and stakeholder concerns, should also

536 be considered. Because our analysis was conducted on the entire waste stream, results can
537 be compared to the system-wide economic effects of changes in food waste management,
538 as well as the broader social and policy impacts of addressing food waste disposal issues.
539 Because previous food waste LCAs look only at food waste, it is difficult to integrate their
540 findings into system-wide economic and stakeholder analyses.

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

548 **Supplementary Materials:** The supplementary material (SM) describes the Life Cycle
549 Assessment (LCA) case study. Section 1 describes previous LCA work focused on food
550 waste. Sections 2 and 3 further describe the model and the case study. Sections 4-6 expand
551 on the results presented in the main section of the paper. Specifically, the following are
552 available online: Table S1. LCAs Focused on Food Waste; Table S2. LCA Boundaries; Table
553 S3. Material Characteristics of Waste Fractions; Table S4. Environmental Impact Categories
554 Included in LCA; Table S5. Process Groups in the LCA; Table S6. Compost Use Process
555 Impacts; Table S7. Sensitivity Analyses; Table S8. Marginal Energy Sensitivity Analysis
556 Results; Table S9. Weighting Criteria; Figure S1. Climate Change (GW) - Process Specific
557 Impacts; Figure S2. Stratospheric Ozone Depletion (ODP) - Process Specific Impacts; Figure
558 S3. Terrestrial Acidification (TA) - Process Specific Impacts; Figure S4. Terrestrial
559 Eutrophication (TE) - Process Specific Impacts; Figure S5. Freshwater Eutrophication (FE)
560 - Process Specific Impacts; Figure S6. Marine Eutrophication (ME) - Process Specific
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575

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