Co-management of domestic wastewater and food waste: A life cycle comparison of alternative food waste diversion strategies

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HIGHLIGHTS
- Co-digestion and AnMBR treatment of food waste maximizes energy recovery.
- Co-digestion minimizes global warming impacts by preventing fugitive gas emissions.
- Composting food waste negatively impacts respiratory effects and eutrophication.
- Energy recovery for AnMBR is sensitive to food waste diversion participation.

GRAPHICAL ABSTRACT

ABSTRACT
Food waste is increasingly viewed as a resource that should be diverted from landfills. This study used life cycle assessment to compare co-management of food waste and domestic wastewater using anaerobic membrane bioreactor (AnMBR) against conventional activated sludge (CAS) and high rate activated sludge (HRAS) with three disposal options for food waste: landfilling (LF), anaerobic digestion (AD), and composting (CP). Based on the net energy balance (NEB), AnMBR and HRAS/AD were the most attractive scenarios due to cogeneration of produced biogas. However, cogeneration negatively impacted carcinogenics, non-carcinogenics, and ozone depletion, illustrating unavoidable tradeoffs between energy recovery from biogas and environmental impacts. Fugitive emissions of methane severely increased global warming impacts of all scenarios except HRAS/AD with AnMBR particularly affected by effluent dissolved methane emissions. AnMBR was also most sensitive to food waste diversion participation, with 40% diversion necessary to achieve a positive NEB at the current state of development.

1. Introduction
Food waste, food discarded or lost in the food supply chain by commercial entities or consumers, is a source of energy that largely goes untapped in the U.S. An estimated 39.7 million tons of food waste are landfilled each year, corresponding to approximately 65% of the amount generated (BSR, 2012). Food waste is thus the largest contributor to municipal solid waste (MSW), comprising 21.1% in the U.S. (U.S. EPA, 2014). Landfilled food waste results in the production of methane gas that contributes to global warming (GW) when uncollected (methane has a GW potential 28 times greater than that of carbon dioxide over 100 years) (Stocker et al., 2013) while also necessitating significant landfill capacity. In response to these concerns, new legislation is motivating proactive implementation of food waste recycling programs. California recently enacted AB 1826 dictating that any business generating 8 cubic yards or more per week must source separate food scraps and yard trimmings for recycling (BioCycle, 2014). It is important to note that minimizing food waste landfilling requires participation...
by both the commercial and residential sectors where 43% and 45% of food waste is generated, respectively (BSR, 2012). Successful diversion programs have been implemented in municipalities such as San Francisco, CA and Seattle, WA where diversion rates of 60–70% have been achieved (MacBride, 2013; Seattle Public Utilities, 2016). However, mandated diversion requirements impose significant pressure on businesses and other organizations to devise new strategies beyond landfiling for food waste disposal.

Composting and anaerobic digestion (AD) are two widely practiced approaches for handling and extracting resources from food waste. Composting of food waste involves piling with other biodegradable MSW and allowing aerobic decomposition to form humus (compost) and has become mandatory in several major U.S. cities (Seattle Public Utilities, 2016; San Francisco, 2009). Conversely, AD of food waste has become an attractive landfill diversion strategy because energy can be recovered in the form of methane-rich biogas which can be subsequently converted to electricity and heat via combined heat and power (CHP) or cogeneration. Further, AD can produce Class A or B biosolids (EPA 40 CFR Rule), which can be used as a soil amendment when land applied (similar to compost). Food waste in the U.S. could power nearly 4.5 million households each year based on U.S. food waste landfill flows, electricity recovery achievable from AD of food waste, and average U.S. household electricity usage. Although energy can be recovered similarly from landfill gas (LFG), currently only 28% is used for energy recovery while 22% is flared and 50% is emitted to the atmosphere (European Commission Joint Research Centre, 2010). AD systems are commonly found at wastewater treatment plants to stabilize sludge produced during aerobic wastewater treatment (e.g., activated sludge processes); therefore, an opportunity exists to co-manage food waste and domestic wastewater using existing activated sludge/AD processes (i.e., co-digestion) or new biotechnologies such as anaerobic membrane bioreactors (AnMBRs).

High-rate activated sludge (HRAS) combined with AD of produced sludge and AnMBR are two competing biotechnologies that maximize energy recovery from wastewater. HRAS is a well-established approach to domestic wastewater treatment whereas AnMBR is relatively new with only bench- and pilot-scale systems operated to date for domestic wastewater treatment. Relative to conventional activated sludge (CAS), HRAS uses a short hydraulic retention time (HRT) of 1.5–3 h and solids retention time (SRT) of <2 d, decreasing aeriation energy demand (Tchobanoglous et al., 2003). The short HRT and SRT maximizes sludge production providing increased energy recovery in AD. AnMBR combines anaerobic treatment with membrane separation to provide energy recovery via biogas production directly from organics in domestic wastewater rather than sludge produced during aerobic treatment. AnMBRs also produce a high-quality effluent comparable to HRAS. One concern is that energy recovery than current practices. Further, HRAS and AnMBR produce a similar effluent quality with regards to carbon and nutrients allowing for environmental impacts associated with effluent discharge or reuse to be excluded from the system boundary. Therefore, three food waste management strategies (landfilling, AD, and composting) were paired with HRAS and compared to AnMBR.

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We previously highlighted the environmental tradeoffs between HRAS and AnMBR in a life cycle comparison for domestic wastewater treatment compared to CAS (Smith et al., 2014). Namely, AnMBR GW impacts were significantly greater due to release of fugitive methane emissions from effluent released to the environment. Others have similarly noted the criticality in addressing dissolved methane management in AnMBR (e.g., Liu et al., 2013). Further, energy requirements for membrane fouling control diminished energy recovery and made achieving a positive net energy balance (NEB) challenging. However, the benefits of AnMBR were more apparent for higher strength domestic wastewater where AnMBR was able to recover more net energy and had comparable environmental impacts to HRAS. One concern is that high strength domestic wastewater is relatively rare in the U.S. and strength is largely outside of our control apart from implementing water conservation efforts and new collection systems that limit infiltration. This provided motivation for the current systems-level study in which we modeled potential life cycle benefits of domestic wastewater and food waste co-management at wastewater treatment plants relative to landfiling or composting of food waste.

This paper is the first to comprehensively explore the viability of co-managing domestic wastewater and food waste via life cycle assessment (LCA). Five scenarios were evaluated: (1) CAS of domestic wastewater and landfiling of food waste (CAS/LF), (2) HRAS of domestic wastewater and landfiling of food waste (HRAS/LF), (3) HRAS and AD of food waste (HRAS/AD), (4) HRAS and composting of food waste (HRAS/CP), and (5) AnMBR co-management of domestic wastewater and food waste (AnMBR). CAS with AD represents current domestic wastewater management and serves as a reference scenario. HRAS with AD and AnMBR both represent technologies that could achieve greater energy recovery than current practices. Further, HRAS and AnMBR produce a similar effluent quality with regards to carbon and nutrients allowing for environmental impacts associated with effluent discharge or reuse to be excluded from the system boundary. Therefore, three food waste management strategies (landfilling, AD, and composting) were paired with HRAS and compared to AnMBR, domestic wastewater and food waste co-management.

2. Methods

2.1. System boundary and functional unit

The international standard for LCA, ISO 14040 (Finkbeiner et al., 2006) was used to compare the environmental impacts of the five scenarios. CAS/LF was considered a reference scenario representing the most conventional practices compared for domestic wastewater treatment and food waste disposal. Only the use-phase environmental impacts were included for CAS, HRAS, and AnMBR as construction phase impacts have been shown to be negligible in comparison to the operational phase of the life cycle for wastewater treatment systems (Renou et al., 2008). A wastewater temperature of 15°C was assumed as representative of average U.S. domestic wastewater (Tchobanoglous et al., 2003). The system boundary of the study for the HRAS and AnMBR scenarios is shown in Fig. 1 (the CAS/LF process flow (not shown) is identical to HRAS/LF).

A functional unit of 5 million gallons per day (MGD) of domestic wastewater treated to meet U.S. EPA secondary treatment effluent standards (biological oxygen demand (BOD5) and total suspended solids (TSS) <30 mg L−1) (U.S. EPA, 1988) was chosen. This was selected as the flow rate based on the U.S. EPA’s CHP Partnership 2007 report, which suggested that 5 MGD was the minimum influent flow rate for economically feasible CHP systems (Naik-Dhungel, 2010). A 5 MGD flow rate represents the wastewater generated by approximately 71,000 people based on average U.S. household size of 2.54 persons (U.S. Census, 2015) and typical flow rate per household of 70.6 gal capita−1 d−1 (Tchobanoglous et al., 2003).
2003). The average person in the U.S. generates 4.38 lb d\(^{-1}\) of solid waste, 21.1% of which is food waste (U.S. EPA, 2014). Therefore, a population of 71,000 people generates an estimated 29,700 kg food waste. Assuming a dry food percentage of 30\%, volatile solids related to dry food waste of 95\%, and 1.54 kg chemical oxygen demand (COD) kg\(^{-1}\) food waste (PE Americas, 2011), supplementing food waste to domestic wastewater adds an additional 688 mg L\(^{-1}\) COD. We assumed a medium strength domestic wastewater of 430 mg/L COD (Tchobanoglous et al., 2003) and thus the COD concentration of a combined domestic wastewater and food waste stream was assumed to be 1120 mg L\(^{-1}\).

### 2.2. System design and modeling for life cycle inventory

HRAS and CAS were modeled using GPS-X (Hydromantis, Inc.) whereas AnMBR performance was based on several existing bench- and pilot-scale systems (Martin et al., 2011; Martinez-Sosa et al., 2012; Pretel et al., 2016a,b; Robles et al., 2012; Smith et al., 2013). Baseline model process parameters are listed in Table 1. Although effluent nitrogen speciation is different for CAS due to nitrification, these environmental impacts were not taken into account. It is also important to note that we assumed that U.S. EPA secondary effluent treatment standards could be met in AnMBR despite the increased organic loading rate (OLR) from food waste addition. It was thus assumed that 95\% COD removal occurred in the AnMBR resulting in an estimated effluent 5-day biochemical oxygen demand (BOD\(_5\)) <20 mg/L (Smith et al., 2013). Only one experimental study to our knowledge has been conducted on AnMBR treating both food waste and domestic wastewater (Pretel et al., 2016a,b) and therefore, these assumptions require further experimental validation. AnMBR biogas production was calculated based on COD removal less COD associated with sulfate reduction (Smith et al., 2013) and biomass growth based on yields in (Hu and Stuckey, 2006). A methane yield of 0.369 m\(^3\)/kg COD (adjusted to 15\(^\circ\)C) was assumed (Tchobanoglous et al., 2003). Dissolved methane was calculated based on Henry’s law (Tchobanoglous et al., 2003) and assumed to be at saturation at the baseline. This assumption was varied in the uncertainty analysis (described below). Sludge wastage from the AnMBR and the AD of the CAS and HRAS systems in all scenarios was based on aggregate U.S. sludge disposal practices (60\% land application, 22\% incineration, 17\% landfill (WEF, 2006)). In the HRAS/LF scenario, sludge produced from HRAS was disposed of.

**Table 1**

Baseline model process parameters.

<table>
<thead>
<tr>
<th>System</th>
<th>SRT (d)</th>
<th>HRT (h)</th>
<th>Recycle Ratio</th>
<th>Flux (LMH)</th>
<th>SGD (m(^3)/m(^2) * h)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CAS</td>
<td>10</td>
<td>8</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>HRAS</td>
<td>1.5</td>
<td>2</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>AnMBR</td>
<td>200</td>
<td>8</td>
<td>2Q</td>
<td>10</td>
<td>0.23</td>
</tr>
</tbody>
</table>

Fig. 1. System boundary of HRAS/LF, HRAS/AD, HRAS/CP, and AnMBR. CAS/LF is not shown, but the system boundary and process flow are the same as HRAS/LF.
according to the practices listed above but all food waste was landfilled.

The membrane properties used in the AnMBR model were based on hollow fiber membranes with materials and weights of the membrane components and recommended chemical cleaning procedure obtained from the manufacturer (Hong, 2012). Food waste addition may impact membrane fouling but the relationship between parameters such as suspended solids concentration and membrane fouling is still poorly understood (Lousada-Ferreira et al., 2015). Further, food waste was not found to impact fouling in an AnMBR study evaluating food waste and domestic wastewater treatment (Pretel et al., 2016a,b). Here, we did not implicitly account for food waste addition increasing AnMBR energy demands for fouling control or reducing membrane flux; however, the uncertainty analysis incorporated a high degree of variability in these parameters (i.e., membrane sparging requirements and fluxes) to account for this possibility.

Food waste was assumed to be collected from local sites (e.g., residential homes, restaurants, and grocery stores) and transported to a landfill, composting facility, or centralized wastewater treatment plant. The collection distances were assumed to be 50 km to the landfill and compost facility (Hospido et al., 2005) and 25 km to the centralized wastewater treatment plant. The energy requirement for pretreatment of food waste prior to AD or AnMBR was assumed at the baseline to be 140 kWh/ton (Takata et al., 2013) and was varied in the uncertainty analysis. An ELCD dataset for landfilling of biodegradable waste including landfill gas utilization, was used to model food waste landfills. The dataset assumed that 28% of LFG was used for energy recovery, 22% was flared, and 50% was emitted to the atmosphere.

The sum of all electrical energy demands for treatment including electricity generated via CHP was used to calculate the NEB. Electricity requirements for pumps, blowers, and mixing were estimated (see Supporting Information (SI)). Equations adapted from CAPDETWorks were used to estimate gravity belt thickening and centrifuge dewatering electricity requirements (Guest, 2012). Digesters were heated to 35 °C using heat generated by CHP and any excess heat was not recovered. Electricity generated from CHP was used to offset average U.S. grid electricity and its associated environmental impacts. While non-electrical energy demands like those incurred from sludge transportation and disposal (e.g., diesel) were excluded from NEB calculations, their environmental impacts were accounted for.

2.3. Environmental impact assessment

Tools for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) was used to characterize environmental impacts (Bare et al., 2002). Emissions data for U.S. grid electricity and other unit processes within the system boundary were gathered from U.S. LCI, Ecoinvent, ELCD, and EU & DK (European Commission Joint Research Centre, 2010; Frischknecht et al., 2005; Norris, 2004; Schmidt and consultants, 2010) (SI Table S1). While the impact of gaseous emissions from dissolved methane in AnMBR and AD effluents was considered, all other environmental impacts from the effluents were excluded from the analysis. In addition, biogenic carbon dioxide emissions during treatment were not included (Monteith et al., 2005).

2.4. Uncertainty and sensitivity analysis

Monte Carlo analysis (50,000 simulations) was performed to evaluate the aggregate impact of data uncertainty within the LCA. The majority of parameter uncertainty was associated with AnMBR given its early stage of development. Transportation distances for sludge and food waste and food waste pre-processing energy demand were also evaluated in the uncertainty analysis (Table 2). Uncertainty parameter distribution was assumed to be triangular when data were available to suggest a likely midpoint value. A uniform distribution was used in the absence of a midpoint estimate.

A sensitivity analysis was performed to evaluate performance based on the percentage of food waste diverted from landfills to treatment facilities (AD, composting, and AnMBR). The analysis showed the effect of varying degrees of participation in food waste diversion on NEB and environmental impacts. HRAS/AD, HRAS/CP, and AnMBR were evaluated assuming food waste diversion participation of 0%, 20%, 40%, 60%, 80%, and 100% to their respective treatment options with the remaining food waste being landfilled. CAS/LF and HRAS/LF NEB and environmental impacts were unaf-fected by food waste diversion.

3. Results & discussion

3.1. HRAS/AD and AnMBR maximize energy recovery

All scenarios achieved a negative NEB at the baseline indicating that energy was recovered. CAS/LF had the lowest energy recovery at 1460 kWh/d primarily due to higher energy requirement at the plant as well as less biogas production (approximately 17% less compared to HRAS). Although HRAS/AD and AnMBR had similar NEB, AnMBR recovered 62% more energy via greater biogas production. However, this enhanced energy recovery was offset by high membrane fouling control energy demand which accounted for 61% of the energy demand at the plant and was 5 times greater than the energy demand for HRAS aeration (Fig. 2). Food waste pre-processing energy demand was significant for both HRAS/AD and AnMBR, representing 55% and 29% of total energy demand, respectively. HRAS/AD generated significantly more sludge than AnMBR, and thus had greater energy demand for sludge thickening and dewatering compared to AnMBR (5.8 times greater) with this demand accounting for 38% and <4% of the total for HRAS/AD and AnMBR, respectively.

Although all CAS and HRAS scenarios involved energy recovery from AD of wastewater sludge, CAS/LF, HRAS/LF, and HRAS/CP recovered the least overall energy. CAS/LF and HRAS/LF also recovered energy in the form of LFG, but the majority of LFG was used for direct-use heat projects (U.S. EPA, 2015). Redirecting food waste from landfills to composting facilities potentially decreases these GW impacts by not producing methane if aerobic conditions are maintained. However, composting does not recover electrical energy from food waste and therefore is less compelling on an energy basis.

The uncertainty analysis for AnMBR returned a broad 95% confidence interval for the NEB as a result of the wide-range of fouling control parameters reported to date (Fig. 2). The majority of the energy demand was a result of the high specific gas demand (SGD), highlighting a critical research area for AnMBR that could significantly improve NEB. The uncertainty in energy demands associated with food waste pre-processing did not significantly contribute to AnMBR NEB uncertainty. Overall, AnMBR NEB uncertainty suggests a high likelihood for AnMBR to remarkably out-compete all other scenarios pending further technological development.
3.2. Fugitive emissions are a major concern for CAS/LF, HRAS/LF, HRAS/CP, and AnMBR

Fugitive emissions of GW contributing gases such as methane are a major concern in waste management processes producing biogas. AnMBR is particularly challenged by the likelihood of effluent dissolved methane emissions significantly contributing to GW impact. For the present study, 0% dissolved methane recovery was assumed at the baseline resulting in dissolved methane emissions constituting 81% of AnMBR GW impacts (Fig. 3). Reducing GW impact for AnMBR hinges on effluent dissolved methane recovery which could limit GW emissions by directly preventing methane release and indirectly offsetting emissions associated with primary energy production. Dissolved methane recovery technologies that are both energetically and economically viable have been explored in the past several years with limited success (Bandara et al., 2012).
However, a recent study suggested feasibility of an AnMBR coupled with a hollow fiber membrane contactor (Cookney et al., 2016) and it is anticipated that further technological developments in this area will yield improved efficiencies. Recovery of 25% dissolved methane would make AnMBR GW impacts similar to HRAS/LF and HRAS/CP whereas 60% dissolved methane recovery is necessary for comparable impacts to HRAS/AD. This level of recovery seems likely given recent research and the likelihood of prioritization of this key research area. It is important to note that food waste addition to AnMBR does not negatively impact GW from dissolved methane emissions; AnMBR effluent methane concentrations remain similar despite the increase in OLR from food waste. Therefore, all methane generated from food waste can theoretically be easily recovered in the produced biogas resulting in advantageous scaling of the NEB and GW impacts (i.e., food waste increases energy recovery and thus offsets GW impacts from primary energy production while also not contributing to GW impacts via direct emissions).

CAS/LF, HRAS/LF, and HRAS/CP had comparable GW impacts although lower in magnitude than AnMBR (Fig. 3). GW impacts from landfilling were primarily associated with fugitive LFG emissions, while impacts from composting came from evolved gases such as methane and nitrous oxide which form under undesirable anaerobic conditions and in the presence of excess nitrogen, respectively (Zeman et al., 2002). Strategies for methane emission reduction for landfilling and composting necessitates improving LFG capture and practicing proper compost heap management, respectively (Beck-Friis et al., 2000). We assumed that 28% of LFG was used for energy recovery purposes, however, we estimated that 100% LFG utilization would improve the NEB such that HRAS/AD is comparable to HRAS/AD and AnMBR. It is important to note that MSW in landfills can continue to produce LFG for 20–30 years and maintenance of capture systems as they age is crucial to avoid operational issues (e.g., blockage due to condensate) that could result in fugitive emissions. HRAS/AD was the only scenario to offset carbon (negative GW impact; Fig. 3) by nearly complete prevention of fugitive methane emissions.

3.3. The compared waste management scenarios feature unavoidable environmental trade-offs

CHP of produced biogas was essential for energy recovery from food waste in HRAS/AD and AnMBR but negatively impacted several environmental impact categories (e.g., carcinogenics, non-carcinogenics, and ozone depletion). HRAS/LF and HRAS/CP had lower impacts in these categories due to lower biogas production and less reliance on CHP (Fig. 4). Using carcinogenics as an example, CHP increased impacts for AnMBR by 45%, whereas the offset electricity only reduced these impacts by 51%. Therefore, the negative effect of CHP on carcinogenics far outweighed any offset from reduced demand on primary energy production. Since HRAS/CP featured the lowest CHP utilization, it also had the lowest ozone depletion impacts. In addition, HRAS/CP avoided ozone depletion impacts associated with food waste landfills. LFG emissions result in significant release of ozone depleting compounds, such as chlorofluorocarbons (CFCs), which negatively impacted HRAS/LF and CAS/LF (SI Fig. S1). Produced electricity from CHP did provide a net environmental benefit in acidification, respiratory effects, and smog impacts. This highlights the tradeoff between achieving a more beneficial NEB via CHP of biogas and negative environmental impacts in the aforementioned categories. Advances in biogas CHP efficiency or alternative approaches such as steam-methane reforming to produce hydrogen may be advantageous to recover energy from biogas with lower environmental impacts. It is important to note that comparing impact tradeoffs, such as those described above, requires impact category weighting.

Assigning relative importance to each impact category is outside the scope of this study and is location-specific, requiring weighting of local versus regional or global impacts. For example, smog minimization may be a priority in urban areas while eutrophication may be a greater concern in nutrient-sensitive watersheds. In addition, we assumed a U.S. energy grid mix of 52% coal, 20% nuclear, 16% natural gas, 7% hydropower and 5% coming from wind, photovoltaic, geothermal and other small energy sources. When CHP offsets the energy produced using a “dirtier” grid mix (e.g., electricity produced exclusively by a coal power plant) there is a greater reduction in environmental impacts associated with primary energy production. On the other hand, grid mixes weighted heavily towards nuclear, hydropower, and wind/photovoltaic sources will have diminished environmental gains from CHP. Therefore, location-specific analyses of local energy sources can significantly influence which technologies to implement.

Environmental impacts related to sludge transportation and disposal significantly benefitted AnMBR because the technology produces substantially lower quantities of sludge compared to aerobic processes. Transportation-based impacts were a function of the mass of food waste and sludge collected and distance transported, quantified as the payload distance which provides better context to the scale of transportation emissions. HRAS/AD produced 3.6 times more sludge than CAS/LF, HRAS/LF, and HRAS/CP and 5.7 times more sludge than AnMBR, which made transportation a large contributor to HRAS/AD emissions. Total payload distances, considering food waste management and sludge handling were 1960, 1960, 2440, 1960 and 1050 tkm for CAS/LF, HRAS/LF, HRAS/AD, HRAS/CP and AnMBR, respectively. Transportation distances for food waste were assumed to be 25 km to a centralized treatment facility (HRAS/AD and AnMBR scenarios) and 50 km to landfill or compost facilities (HRAS/LF and HRAS/CP scenarios) further reducing the transportation-based emissions for AnMBR. AnMBR transportation impacts contributed significantly only to smog (18%), ecotoxicity (14%), and non-carcinogenics (8%). Although AnMBR had reduced transportation-based impacts, the majority of these savings were offset by impacts related to membrane materials and required maintenance (chemical cleaning) that were a factor unique to AnMBR (SI Fig. S1). For example, 23% of AnMBR ecotoxicity impacts were associated with membranes, higher than the equivalent for transportation (14%). HRAS/AD benefited from the reduced distance for food waste transport but this was largely outweighed by high sludge production. Impacts for HRAS/AD thus varied significantly with transportation distance: varying sludge transport distance by ±10% for HRAS/AD affected ecotoxicity and smog by ±10% and ±27%, respectively. Overall, transportation impacts for HRAS/AD contributed significantly to smog (38%), ecotoxicity (38%) and non-carcinogenics (18%).

Composting in HRAS/CP had the greatest impact on respiratory effects (97%), eutrophication (98%), and acidification impacts (64%) (SI Fig. S1). Composting associated respiratory effects were associated with ammonia gas emissions which react with atmospheric NOx and SOx to form particulate matter (Harrison and Yin, 2000). Atmospheric ammonia also causes eutrophication in nitrogen-limited watersheds and soil acidification as it deposits into the surrounding ecosystem (Fangmeier et al., 1994). However, composting does provide an offset to ecotoxicity impacts due to the mitigation of nitrogen and phosphorus fertilizer production from land applying produced compost.

The uncertainty analysis for CAS and the three HRAS-based scenarios all exhibited small confidence intervals relative to the AnMBR scenario due to the maturity of CAS, HRAS, AD, landilling, and composting technologies. This contrasted with AnMBR, as it is an emerging technology with a high likelihood of efficiency gains as it reaches full-scale implementation. AnMBR GW impact was
Fig. 4. Radar chart showing the environmental impacts compared by TRACI impact category for CAS/LF, HRAS/LF, HRAS/AD, HRAS/CP, and AnMBR. (A) Emissions factors of each scenario. (B, C, D, E, F) Confidence interval from uncertainty analysis for CAS/LF, HRAS/LF, HRAS/AD, HRAS/CP, and AnMBR, respectively. The radar chart units are an emissions factor and range from $10^{-6}$ to 10 (SI Table S4). The emissions factor is the environmental impacts of HRAS/LF, HRAS/AD, HRAS/CP, and AnMBR normalized to the environmental impacts of CAS/LF. The emissions factor for CAS/LF is 1. Points within the 0 value hexagon indicate a “savings” in the respective category. The inner and outer dotted-lined represent the 2.5 and 97.5% percentiles, respectively.
most sensitive to SGD, methane oversaturation, and dissolved methane recovery (SI Table S3). Ozone depletion was especially sensitive to dissolved methane recovery. Although AnMBR environmental impacts could be improved significantly, the technology already shows promise in terms of the NEB, acidification, eutrophication, respiratory effects, and smog impacts. Future development of the technology will likely improve AnMBR performance beyond that of HRAS/LF and HRAS/AD both environmentally and energetically. However, unavoidable trade-offs will remain, such as with CHP contributing significantly to various impact categories. Composting can thus be an attractive strategy depending on location-specific weighting of impact categories.

3.4. AnMBR requires a minimum of 40% food waste diversion to achieve net energy recovery

Participation in food waste diversion programs is paramount to achieving potential environmental benefits. AnMBR showed the greatest sensitivity to food waste diversion, affecting both energy and GW potential significantly (Fig. 5). The NEB for AnMBR became positive at 40% food waste diversion and was greater than all other scenarios at 80% and 100% diversion. GW impact decreased with increasing food waste diversion participation as a result of increased energy recovery and reduced LFG impacts. Direct methane emissions remain a concern for AnMBR regardless of food waste diversion.

Fig. 5. (A) NEB (electricity only) and GW for CAS/LF, HRAS/LF, HRAS/AD, HRAS/CP, and AnMBR at varying food waste diversion (%). CAS/LF and HRAS/LF are represented by horizontal lines. HRAS/AD, HRAS/CP, and AnMBR are represented by dashed lines for NEB and solid lines for GW. Box and whisker plots for (B) NEB and (C) GW of all scenarios at 100% food waste diversion.
waste diversion because dissolved methane concentration in the effluent is unaffected.

HRAS/AD displayed similar NEB and GW trends to AnMBR, although less sensitive to food waste diversion participation. At 0% food waste diversion, all HRAS scenarios were predicted to recover net energy. HRAS/AD NEB becomes more favorable at increasing food waste diversion whereas the inverse occurs for HRAS/CP. Composting reduces energy recovery possible from LFG capture making food waste diversion less energetically attractive for this scenario. At 100% food waste diversion, HRAS/CP has one of the least favorable NEBs, comparable to CAS/LSF. However, GW impact remains relatively constant with varying food waste diversion, since landfill and composting had similar trends in this impact category.

4. Conclusions

Co-management of domestic wastewater and food waste provides synergies that could improve environmental impacts of conventional segregated management. Currently, HRAS/AD provides significant energy recovery relative to CAS/LSF, HRAS/LSF, and HRAS/CP but features environmental tradeoffs related to biogas CHP. AnMBR offers slightly higher energy recovery at the current state of development and has the greatest potential to minimize environmental impacts assuming technological developments are achieved prior to full-scale implementation (e.g., reduce fouling control energy demand and recover effluent dissolved methane). Experimental verification of AnMBR performance during co-management of domestic wastewater and food waste is an important next step in AnMBR research.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biortech.2016.10.031.

References

European Commission Joint Research Centre, 2010. ELCD Life Cycle Inventory Database.


