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Life cycle costing of municipal food waste management systems: The effect of environmental externalities and transfer costs using local government case studies

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ABSTRACT

Municipal food waste is of growing interest worldwide as governments strive to reduce wastes' environmental impact. For local governments responsible for waste collection, treatment and disposal; finding an affordable management and treatment solution with the smallest impact on the environment is paramount. This paper introduces a life cycle costing tool, incorporating both an Environmental LCC and Societal LCC to provide decision support to local government. The tool was tested on two case study waste catchments; comparing seven unique food waste management systems. Results indicated that food waste anaerobic digestion or co-digestion, shown to have smaller environmental impacts, can be implemented with marginal increase (1.7% to 11.6%) in overall cost. Moreover, with efficiencies food waste anaerobic digestion systems can demonstrate the same cost as business-as-usual. Composting systems were also revealed to have consistently lower costs per household across case studies than business-as-usual, although environmental impacts were generally higher than digestion systems. Further analysis determined the required rate of policy incentives (i.e. landfill levy, electricity tariffs and carbon credits) to promote various alternative food waste management systems.

1. Introduction

There is a need to reduce and more efficiently manage the worlds wasted food. It has been estimated that one third of all food produced for human consumption is lost or wasted. This equates to 1.3 billion tonnes/year of food waste globally; if this food were a country it would be ranked as the 3rd largest greenhouse gas emitter (FAO (Food and Agricultural Organisation of the United Nations), 2011; Lipinski et al., 2017). Wasted food generated in the home or business contributes as much as 61% of food loss or waste in many nations (Lipinski et al., 2017). Whilst reduction of food waste is critical, recovering resources from municipal food waste (FW) is also considered a valuable component to reducing the environmental and economic impact of global food production and consumption. Especially the unavoidable peelings, skins, coffee grinds etc. of FW.

One of the most common forms of FW management across the globe is sending it to landfill. Largely due to it being the cheapest and easiest solution, particularly in regions where there is adequate space (Randell et al., 2014). Diverting FW from landfill by using alternative treatment methods like anaerobic digestion (AD) or composting however, has frequently shown to have a better outcome for the environment (Laurent et al., 2014; Morris et al., 2013).

To promote the use of alternative FW treatment methods many governments have begun to impose financial incentives and other policies to help them compete against landfill. Policy measures were demonstrated to increase the use of alternative treatment methods in Germany and the United Kingdom (Edwards et al., 2015). Some policies aim to discourage landfilling of waste through imposing a levy or outright ban as in many EU nations. Other policies ensure the generation of FW and the recovery of energy or resources from waste through mandatory collection of source separated FW. As landfilling FW contributes significantly to greenhouse gas emissions, the taxing of carbon emissions or paying for the abatement of carbon emissions is also used.

Whilst the above policy measures focus on making alternative FW treatments competitive; alternative FW treatments typically require a different method for pre-processes like separation, collection, and pretreatment. Moreover, they require a different end-use or disposal method for by-products of the treatment process. Each pre-process or

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post-process can incur additional costs or benefit. Therefore, waste managers need to gauge the complete cost of a FW management system and not just assess a treatment method or technology in isolation. To do so a system wide analysis tool like life cycle costing (LCC) is suitable (De Menna et al., 2018).

Despite LCC being a longstanding framework originating in the 1930′s, it is only over the past five years that a small but growing number of FW focused LCC studies have been published (De Menna et al., 2018). This is due to the development of the Environmental-LCC (E-LCC) framework by a working group from the Society of Environmental Toxicology and Chemistry (SETAC). The framework is able to incorporate multiple stakeholders and better accompanies a traditional LCA goal and scope (De Menna et al., 2018). Despite a recent uptick in publications on FW management there is still limited examples of FW LCC confinging knowledge on how different geographical regions, FW management systems and methodological settings impact results. As an example only one Australian study has looked at a financial analysis of selected waste management systems using a life cycle approach, it used a broad scope (state level) and did not include externality costs (non-marketed goods like environmental pollution) in their analysis (Schacher et al., 2007). In waste management systems many parameters in the sorting, collection and pre-treatment steps are locality-specific and therefore impose a limitation on general broad financial analysis. Whilst this study the aforementioned study is valuable, it does not provide the resolution required for waste management systems that operate on a local scale and with variables that can be vastly different from one jurisdiction to the next (i.e. waste characteristics, sorting rates, collection distances etc.). One recently published LCC study (research) based in the USA provided a case study analysis that included local variables, compared various waste management systems, and included externality costs (Martinez-Sanchez et al., 2017). However, the applicability of this study to Australian conditions is limited as a USA specific LCA toolkit developed by Levis et al. (2014) was used. Moreover, the USA study only included air emission externality costs and did not include anaerobic co-digestion of FW and sewage sludge (SS) as an alternative FW management system. This leaves a gap in the literature regarding the LCC of anaerobic co-digestion of FW with SS (AcoD) both as part of a waste system and as a standalone treatment technology; a gap highlighted by Burn et al. (2014).

This study therefore determines and compares the LCC of seven waste management systems, including different AcoD based systems. The LCC is applied to two local government waste catchments using the newly developed LCA tool detailed in Edwards et al. (2017a) as the costing framework. The LCC includes externality costs of air and water pollution, as well as relevant taxes, capital and operating expenditure. Furthermore, the study uses sensitivity and scenario analysis to explore the impact of different policy measures and physical parameters including increases in landfill levies, a price on greenhouse gas air emissions, increasing sorting efficiencies, and an increase in methane generation rates. In doing so the research demonstrates the key policy levers and their impact on the LCC of various FW management systems. The paper seeks to provide local government and decision makers a unique insight into the financial and environmental impact of various FW management systems. Furthermore, it seeks to demonstrate the potential financial effect of current and prospective government policy measures on FW management systems; policy measures a government may utilise to promote a more environmentally friendly FW management system.

2. Materials and method

2.1. System boundary

The system boundary was considered to begin at the point where waste was collected and ends at the point at which waste was either disposed of or re-used as a product. This means the system begins upon the collection process, incorporates all transportation, pre-treatment, treatment and end-use processes. The 'zero burden' approach was adopted in the study, whereby the impact of all life cycle stages prior to collection of waste were considered identical, meaning these prior processes would not affect the directional outcomes of the study. Electricity and biosolids/compost by-products generated in select FW management systems are considered within the system boundary. These products are modelled using attributional modelling, i.e. the products were considered to replace similar products on the market in full and apportioned to the energy output (electricity offsets the average electricity supply as per the local grid mix (Edwards et al., 2017a,b), or the mass of N, P and K in the case of compost and biosolids (offsetting the equivalent quantity of N, K2O and P2O5 fertiliser).

Allocation was avoided by expanding the system boundary to include sewage sludge treatment as well as other waste streams like garden waste and inert waste. Given FW is currently collected as co-mingled with residual waste (i.e. non-recyclable inert material that is sent to landfill like textiles, soft plastics, and sanitary items) the tool developed incorporates the management of all residual waste, to avoid unfair LCA and LCC comparisons between systems. Moreover, as the anaerobic co-digestion of FW and SS is used in two systems being compared, so to the simultaneous composting of FW and garden waste (GW), the tool also incorporates the treatment of SS and GW. For SS, the system boundary covers all sludge generated at the wastewater treatment plant (WWTP) once it has been dewatered and immediately before being fed into the anaerobic digester.

The total air emissions from landfill and biosolids/compost land application is modelled over 100 years and all emissions are deemed to occur instantaneously in the model so as not to prejudice systems. This approach is common for waste management LCA that seek to mitigate the unavoidable constraint of comparative LCA where different time frames are observed across competing systems.

2.1.1. Functional unit

The primary functional unit is to collect, treat and manage one years' worth of municipal collected residual waste, FW and GW by each local government case study and one years' worth of SS generated at the local WWTP. The two case studies reference flows (in wet weight) ascribed are;

- Melton city council (CASE 1) provides waste services to 36,919 households. 10,461 Mg of residual waste, 8559 Mg of FW, 8125 Mg of GW, and 22,574 Mg of SS
- Sutherland shire council (CASE 2) provides waste services to 82,470 households. 33,280 Mg of residual waste, 17,920 Mg of FW, 13,000 Mg of GW, and 91,300 Mg of SS

The primary purpose of each system modelled is to safely collect and treat or recycle kerbside waste for every serviced household. Whilst cost items are spread across councils, collection companies and households; ultimately the burden falls upon the household as rate payer. Hence, to view the total cost of a system in the context of "cost per serviced household" is useful for decision makers. This involves excluding the cost to treat SS as it is derived from both industrial and domestic wastewater and is charged to domestic users through water utility agents. However, for the systems AcoD and INSINK, which imposes costs of FW management to the water utility, only additional costs brought about by FW management have been factored in.

Therefore, a secondary functional unit, used only in Section 3.1.1.1 is the municipal waste collection and treatment service provided to individual households. The refence flows for this include:

- CASE 1 – services 36,919 households for mixed residual and FW, and 21,050 households for GW. (COMP system increases to 36,919 households for GW and FW collection, as well as for residual bins serviced)