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Combined material flow analysis and life cycle assessment as a support tool for solid waste management decision making

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ABSTRACT

Material flow analysis (MFA) and life cycle assessment (LCA) have both widely been applied to support solid waste management (SWM) decision making. However, they are often applied independently rather than conjointly. This paper presents an approach that combines the MFA and LCA methodologies to evaluate large and complex SWM systems from an environmental perspective. The approach was applied to evaluate the environmental performance, focusing on greenhouse gas (GHG) emissions, of a local authority SWM system and to compare it with alternative systems to assess the potential effectiveness of different waste policy measures. The MFA results suggest that national recycling targets are unlikely to be met even if the assessed policies are implemented optimally. It is likely that for the targets to be met, investigated policies would need to be combined with additional policies that target reductions in waste arisings. The LCA results found landfilling of residual waste to be the dominant source of GHG burdens for the existing system, whilst material reprocessing was found to result in GHG benefits. Overall, each of the alternative systems investigated were found to result in lower GHG impacts compared to the existing system, with the diversion of food waste from the residual waste stream found to be potentially the most effective strategy to reduce GHG emissions. The results of this study demonstrate that the complementary methodologies of MFA and LCA can be used in combination to provide policy and decision makers with valuable information about the environmental performance of SWM systems.

1. Introduction

Currently, around 1.3 billion tonnes of municipal solid waste are generated annually worldwide and generation levels are projected to almost double by 2025, driven by rapid population growth, urbanisation, and socio economic development in developing countries (Hoornweg and Bhada-Tata, 2012). A substantial proportion of this waste material can be viewed as a resource. As demand for natural resources continues to rise, there is increasing pressure on the world's natural resource base, which is having severe environmental consequences (Hertwich et al., 2010). At a global scale, climate change is a serious international concern and the extraction, processing, and use of natural resources contributes directly to climate change through the burning of fossil fuels, whilst the disposal of materials in landfills contributes through emissions of greenhouse gases (GHG). Improving solid waste management (SWM) by recovering value in the form of material and energy resources can contribute towards enhanced resource efficiency and GHG mitigation efforts (UNEP, 2010).

The waste management sector is under increasing pressure to improve its environmental performance. In the European Union (EU), Member States are legally obligated to formulate and implement regional policy instruments to meet the environmental SWM objectives and targets outlined in a broad international legal framework. Article 4(1) of the EU Waste Framework Directive establishes the “waste hierarchy”, a five step priority order of waste management comprising, in descending order of priority, prevention, preparation for reuse, recycling, other recovery (e.g. energy from waste), and disposal (EC, 2008). Under the terms of Article 21(1) of the EU Waste Framework Directive, all waste management decisions must be undertaken in line with the waste hierarchy. The Landfill Directive sets a target for member states to reduce the amount of biodegradable municipal solid waste going to landfill to 35% of 1995 levels by 2016 (EC, 1999). A target of achieving 50%

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Nomenclature

AD	anaerobic digestion
CCGT	combined cycle gas turbines
CHP	combined heat and power
EC	European Commission
ELCD	European Life Cycle Database
EU	European Union
GHG	greenhouse gas
GWP	Global Warming Potential
HWRC	household waste recycling centre
ILCD	the International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
IVC	in-vessel composting
LACW	local authority collected waste
LCA	life cycle assessment
LCI	life cycle inventory
LDA	large domestic appliances
MBT	mechanical biological treatment
MFA	material flow analysis
MRF	material recovery facility
NRW	Natural Resources Wales
OWC	open windrow composting
RDF	refuse-derived fuel
SWM	solid waste management
WAG	Welsh Assembly Government
WG	Welsh Government

recycling of key household waste materials (paper, glass, metals, and plastics) is established in the Packaging and Packaging Waste Directive (EC, 1994). Moreover, at a broader level the EU is committed to reducing its GHG emissions by at least 20% and 40% of 1990 levels by 2020 (EC, 2009) and 2030 (EC, 2014), respectively. Managing resources to maximise environmental sustainability and contribute towards the achievement of national targets set by the EU entails important strategic and investment decisions by local waste managers, who are simultaneously tasked with maintaining a reliable and economical waste removal service to residents under increasing budgetary pressures. To reduce the burden on local waste managers and promote environmental sustainable practices, there is a need for strong waste policies that guide and enable effective local decisions and actions.

Analytical tools are required to assist local and national governments in evaluating the environmental performance of potential policy measures and local decisions (Turner et al., 2011). Such tools must be capable of handling the increasing complexity of modern ‘integrated solid waste management’ systems. Modern SWM encompasses a large number of waste treatment technologies, such as incineration, composting, and anaerobic digestion (AD) that are each designed to manage specific waste streams. Many of these technologies provide additional functions, such as secondary materials production and energy production, that necessitate interaction with other sectors, such as manufacturing, agriculture, and energy production (Giugliano et al., 2011). Furthermore, modern SWM systems comprise a global network of facilities, each with distinct technological facets and different levels of operational performance. It is necessary that analytical tools used to support decision making in complex, interdependent systems, such as SWM, adopt a whole system approach that reflects this complexity (Blengini et al., 2012).

In this paper we apply an approach that combines two systems based methodologies, material flow analysis (MFA) and life cycle assessment (LCA), to quantitatively evaluate a complex, municipality-scale SWM system and use scenario analysis to assess the potential effectiveness of different national waste policy measures. The novel contribution of this paper can be summarised as follows:

- Application of a combined MFA and LCA approach to evaluate a complex, multi waste stream SWM system at the meso level.
- Novel use of publically available waste data to comprehensively model waste flows through the system.
- Provision of information to national government regarding the potential effectiveness of waste policy measures.
- Assistance to local government in identifying optimal SWM solutions.

1.1. Case study

Wales is a constituent country of the UK that covers an area of 20,779 km^2 with an estimated population of 3.1 million in 2014 (ONS, 2015). Wales comprises 22 unitary authorities that are individually responsible for arranging waste collection and disposal. The Welsh Government (WG) has introduced a broad and ambitious sustainable development strategy that aims to make sustainable development the core principle of all national and sub national policy and decision making (HMSO, 2015; WAG, 2009). Two key national targets for 2050 have been set: 1) achieving “zero waste” (i.e. eliminating landfilling as far as possible) and 2) reducing national GHG emissions by at least 80% below 1990 levels (HMSO, 2008; WAG, 2010a,b; 2011). Furthermore, the WG have also set a target of a 3% reduction in national GHG emissions per year until 2050, to which waste management is required to contribute (WAG, 2010a).

The city of Cardiff is the capital of Wales and is located in the south of the country. The city covers an area of 140.3 km^2, of which around 76 km^2 is considered urban, and has a population of approximately 341,100 and a dwelling stock of 135,796. The city has a reported recycling rate of 52.2%, marginally below the national average of 52.3% (StatsWales, 2015b).

The Council operates an alternate weekly kerbside collection service for household residual waste and dry recyclables. Waste materials collected for recycling include paper, card, aluminium cans, steel cans, mixed plastics, and mixed glass. Garden and food waste are each collected separately on a weekly basis from households, whilst an optional absorbent hygiene products collection service operates fortnightly. A bespoke bulky waste collection service is also offered. There are four household waste recycling centres (HWRC) and 14 bring sites (recycling banks) located across the city. The Council also runs services for the collection of wastes from: commercial organisations, street cleaning, fly tipping, and municipal parks/grounds.

2. Methodology

A combination of methodologies was applied in this study to quantitatively evaluate the environmental performance of Cardiff's local authority collected waste (LACW) management system and those of alternative systems. LACW comprises all solid waste collected by a local authority. A static MFA approach was applied to

[2] Question 100 is a question in WasteDataFlow that asks local authorities to record the physical flows of collected wastes through all treatment facilities until those wastes reach their end destination.

assess the mass flows and stocks of LACW into, within, and from the defined systems. MFA is descriptive, systematic approach to assess the metabolism of a defined system and is based on the principle of mass conservation (Brunner and Rechberger, 2004). The freeware STAN v2.5, developed at the Vienna University of Technology, was used to conduct the MFA (Cencic and Rechberger, 2008). STAN allows for visualisation of complex MFA systems, performance of data reconciliation, and consideration of data uncertainties. Mass flow quantities are expressed in terms of mean value and standard deviation, defined as normal distributions. Data reconciliation, based on the method of least squares regression (see Fellner et al., 2011), is performed to enforce mass balance constrains on uncertain, conflicting input data. The extent to which data are reconciled is determined by the initial data uncertainty, which is used as the weighting factor. STAN has been used in many previous MFA studies of SWM systems (e.g. Allegrini et al., 2015; Arena and Di Gregorio, 2014; Boldrin et al., 2011; Mastellone et al., 2009).

The environmental impacts of the system described by the MFA were assessed using LCA. LCA has been applied extensively to evaluate SWM systems (see Laurent et al., 2014a). The focus of the LCA undertaken in this study was to evaluate the potential climate change impacts of LACW management and to identify the waste streams and waste treatment processes that contribute most significantly to these impacts. The LCA followed the ISO 14040 and 14044 standards for LCA (ISO, 2006a,b) as far as possible and was performed using the EASETECH software. EASETECH, developed at the Technical University of Denmark, is a LCA model for the assessment of environmental technologies that allows for detailed modelling of complex systems and provides specialist functionality for modelling waste management systems (Clavreul et al., 2014). EASETECH has been used in previous LCA studies of SWM systems (e.g. Butera et al., 2015; Turner et al., 2016; Yang et al., 2014).

2.1. Goal definition

The goal of this study was threefold: 1) to quantify the mass balance of waste of the existing LACW management system; 2) to quantitatively evaluate the environmental performance of the existing LACW management system with regards to GHG emissions; and 3) to compare the environmental performance of the existing LACW management system with that of alternative systems designed to represent the implementation of national waste policy measures. The purpose was to support regional policy and decision making, i.e. ILCD (International Reference Life Cycle Data System) decision context situation B, meso level decision support (EC et al., 2010). The study was carried out in cooperation with the WG and Natural Resources Wales (NRW), the regulator and principal advisor to the WG, who, along with Cardiff County Council, were the key audience. Additionally, aspects of the study will be of value to other national policy makers and local governments, as well as the wider SWM MFA and LCA communities.

The quantitative assessment of the systems under investigation was carried out using LCA. The LCA followed an "attributional" approach (EC et al., 2010), with allocation avoided by means of system expansion. The assessment included the potential environmental impacts from SWM activities as well as impacts on processes in other systems that are affected by SWM activities, principally the recovery of materials and energy production (Giugliano et al., 2011). The system comprised two subsystems: (1) a foreground system that includes processes directly engaged in the management of the reference flow (here, LACW); and (2) a background system that interacts with the foreground system by supplying or receiving resources, including avoided primary material and energy production (Clift et al., 2000).

Background data were taken from international life cycle inventory (LCI) databases (ecoinvent v.2.2 and ELCD v.2), the UK GHG conversion factors repository (Defra et al., 2013), and literature (see Appendix A). Foreground system data were derived from various secondary sources and are described in detail in Section 2.3. Following an attributional approach, average data were generally used to model foreground and background system processes. However, based on the recommendations of the ILCD (EC et al., 2010) and Laurent et al. (2014b) for LCA studies aiming to provide meso level decision support, long term marginal process data were used to reflect the large scale consequences on the energy system from avoided material and energy production. This approach, which is similar to that taken by Rigamonti et al. (2010) and Finnveden et al. (2005), was adopted rather than a "pure" attributional approach as it more accurately reflects the large scale implications of SWM activities on other systems. Specifically, the following assumptions were made concerning material and energy substitution:

- Electricity produced from SWM activities displaced an equivalent amount of electricity generated from combined cycle gas turbines (CCGT), which is considered as being the long term marginal energy source in the UK by the government (DECC, 2014a). Generated electricity is transmitted to the National Grid at an efficiency of 98% (National Grid, 2008).
- Heat produced at facilities fitted with combined heat and power (CHP) units would be used internally due to a lack of established district heating networks in the UK (Hawkey, 2012). Hence, no avoided energy production was associated with heat energy production.
- Secondary products produced from recovered waste materials replace the production of alternative products, including those produced from primary resources. Avoided materials production was modelled based on the average market situation. However, to reflect large scale consequences on the energy system, avoided material production process electricity use was modelled using the long term marginal electricity source, i.e. CCGT. The extent of the displacement was calculated based a substitution ratio, which was calculated as the product of three parameters: (1) recyclability (the amount of a waste material in a recycled product, considering all process material losses); (2) material quality loss (reflects any diminishment in the inherent technical properties of a waste material incurred through reprocessing); and (3) market substitution rate (the actual amount of an alternative product that is replaced at the market). Material losses and material quality losses are defined for each recycled or reused material. Market substitution rates of 1 were used for all product substitution calculations based on the supposition that production of secondary products would not affect the market situation (Briffaerts et al., 2009; Merrild et al., 2012; Rigamonti et al., 2009).

2.2. Scope definition

The function of the system under investigation is to manage LACW from Cardiff. Hence, the functional unit was defined as the management of the total amount of collected LACW in Cardiff between April 2012 and March 2013. The amount is 168,526 tonnes. LACW is categorised into five primary waste streams (commingled materials, source segregated food waste, source segregated garden waste, source segregated dry recyclables, and residual waste) that are derived from nine waste collection sources (household kerbside collection, bulky waste collection, HWRCs, bring sites, street cleaning, fly tipped materials, non household collections, voluntary sector/community collection, and municipal parks/grounds waste collection). All activities required to manage the LACW from the point of collection to its ultimate disposal, reprocessing, or reuse were considered. Hence, the processes of collection, transportation, intermediate processing (sorting and handling), thermal treatment, biological treatment, material reprocessing and reuse, and landfill were individually quantitatively analysed in terms of their resource consumption and emissions.

2.2.1. System boundaries

The spatial boundary of the system was defined by the administrative boundary of Cardiff County Council, i.e. only waste generated within this defined spatial boundary was considered in the analysis. The temporal boundary was the April 2012 to March 2013 fiscal year. In this study, the "zero burden assumption" was adopted, whereby the environmental impacts from upstream life cycle stages prior to the collection of LACW were not included (Ekvall et al., 2007). Environmental impacts from capital goods were also excluded, an approach consistent with other LCA studies of waste management systems (Cleary, 2009). Due to difficulties in data collection, environmental burdens from operation of HWRCs were also excluded.

Processes included in the assessed system comprise collection and transport of LACW, waste management facilities (including those for the treatment of residuals, such as incinerator bottom ash or fly ash), and utilisation of secondary products (e.g. digestate or compost). Waste management facilities included in the assessed system were identified from Cardiff's WasteDataFlow[1] Question 100[2] returns for the four quarters of the fiscal year 2012–2013 (WasteDataFlow, 2014), which include details of facilities that handled collected LACW. The Cardiff LACW management system comprised 158 individual facilities located across nine different countries. 139 facilities are located in the UK (41 in Wales and 98 in England), five were located in Europe, and 13 were located in Asia. Facilities was categorised into one of 11 different waste treatment technology types (e.g. AD, incineration, material reprocessing, etc.), based on their description in WasteDataFlow, and were allocated individual identification codes that reflected the facility type and location (domestic or foreign). In addition, 15 generic processes were included to manage collected waste where no waste treatment facility was specified in WasteDataFlow. Details of the facilities (unit processes) included in the assessed system are provided in Appendix B. Facilities were modelled in the MFA and LCA as unit processes.

2.2.2. Impact coverage

The quantitative potential environmental impact assessment of this study focused on GHG emissions. Hence, climate change was the only potential impact category included in the life cycle impact assessment. This decision was made for two principal reasons:

- Climate change is recognised as being a significant global environmental problem of foremost importance (IPCC, 2014; ISWA, 2015; WAG, 2010a); and
- Climate change impact has been identified as being a good proxy for the overall environmental impacts of SWM (Defra, 2011).

[1] WasteDataFlow is a publically-available, web-based system established in 2004 to enable local authorities in the UK to report certain municipal solid waste data to the national government.

The GHGs considered in the assessment were carbon dioxide (CO_2) (fossil and biogenic), methane (CH_4), and nitrous oxide (N_2O). Emissions of these GHG represent more than 90% of total GHG emissions from SWM (Bogner et al., 2007).

2.2.3. Scenarios

Four scenarios are compared in this study. The first scenario represents the existing LACW management system, whilst the other scenarios reflect different systems based on national waste policies. Scenarios are compared on the basis of functional equivalence – i.e. it was assumed that waste generation was not influenced by the different scenario assumptions.

S1 *Baseline scenario* Baseline scenario representing the existing LACW management system for Cardiff in April 2012 to March 2013.

S2 *Enhanced food waste capture* Wales' *Municipal sector plan* (WAG, 2011) states that in order for Wales' 70% recycling target for 2025 to be met, it is likely that local authorities will need to separately collect for biological treatment at least 80% of biodegradable waste. To evaluate the potential GHG impacts of enhanced biodegradable waste capture, in this scenario an 80% capture rate of food waste via household kerbside food waste collection is assumed, with food waste diverted from the household residual waste stream. All food waste that is collected is sent to AD, the priority destination for food waste in Wales (WAG, 2010b).

S3 *Enhanced incineration* The *Towards Zero Waste* (WAG, 2010b) waste strategy for Wales sets a target for a maximum level of incineration (with energy recovery) of MSW of 30% by 2025. This scenario was constructed to evaluate the impact of enhanced incineration of LACW. Based on the total quantity of collected LACW for Cardiff in 2012–2013 (168,426 t), the 30% limit set by WG allows for up to 50,530 t of LACW to be incinerated. In this scenario, 50,000 t of residual waste originally collected and sent to landfill is instead sent for incineration.

S4 *Enhanced dry recycling* A target for a minimum level of recycling and biological treatment of municipal solid waste of 70% by 2025 was set by the WG in *Towards Zero Waste* (WAG, 2010b). To evaluate the impact of an enhanced dry recycling strategy, in this scenario optimum household kerbside collection recycling rates (taken from Eunomia et al., 2011), are achieved for key recyclable waste materials (glass, paper, cardboard, steel cans, aluminium cans, and plastics), with materials diverted from the household kerbside collected residual waste stream.

2.3. Life cycle inventory

2.3.1. Reference flow characterisation

A static MFA approach was applied to characterise the mass flows and stocks of LACW into, within, and from the defined system. Due to its complexity and size, the system was divided into four subsystems, representing the management of primary waste streams (residual waste, commingled materials, source segregated dry recyclables, and source segregated food and garden wastes). The import flows into each system were the collected LACW primary waste streams, and the export flows comprised secondary products and emissions. To account for flow multidimensionality, secondary waste types (e.g. paper, glass, food waste) were characterised as fractions of the primary waste stream.

Due to the functional complexity of many processes and the profusion of functionally equivalent processes in the system (i.e. SWM facilities engaged in the same activity), it was necessary to divide many processes into subsystems that contain subprocesses and, in some cases, further subsystems. The four defined systems for the baseline scenario (S1) are, collectively, composed of 548 processes and 1042 flows.

Mass flows of waste into the LACW management system and to and between processes were estimated by aggregating Cardiff's WasteDataFlow Question 100 returns for the four quarters of the fiscal year 2012–2013. Where data were not available from WasteDataFlow (e.g. mass flows of wastes and secondary products from treatment facilities and reprocessing), process transfer coefficients were derived from the LCI process inventory models (detailed below). All data used in this study were single value and were inputted into the MFA as mean values and an associated uncertainty, expressed as a percentage. Details of the data reconciliation approach are provided in Appendix C.

2.3.2. Waste composition

The overall waste composition for the investigated system is presented in Fig. 1. Further details are provided in Appendix D.

2.3.3. Collection and transport

Details of the modelling approach for collection and transport are presented in Appendix E.

2.3.4. Transfer station

Transfer station process electricity and diesel consumptions of 4 kWh/t and 1 l/t, respectively, were assumed, based on measured data from UK based transfer stations reported by Eunomia et al. (2011).

2.3.5. Anaerobic digestion (food waste only)

The AD of food waste process was modelled based on dry, mesophilic (one stage, one phase) technology, as described by Møller et al. (2009). Details of process inputs and parameter values are presented in Table 1. During waste reception, reject rates of 8%, 50%, and 95% were applied for food waste and fine material, other biodegradable waste, and non biodegradable waste, respectively (Bernstad and la Cour Jansen, 2011). Biogas was assumed to be utilised on site in a CHP engine - the most common biogas utilisation technology in the UK (Horne et al., 2013) - with gross electrical and heat energy generation efficiencies of 31% and 49%, respectively (DECC, 2014b). Heat energy was assumed to be used internally. CHP unit process emissions, produced through incomplete combustion, of 434 mg CH_4/MJ_{biogas} and 1.6 mg N_2O/MJ_{biogas} were assumed (Nielsen et al., 2010). Digestate application was modelled based on the average situation in the UK in 2012, with 91% applied to agricultural land, 6% used in horticulture/gardening,

Table 1
LCI data for the AD (food waste only) process. Note that data are presented per tonne throughput.

	Unit	Value	Reference
Pre-treatment			
Inputs			
Diesel	l	0.12	Primary data from industrial source.
Electricity	kWh	1.1	Primary data from industrial source.
Anaerobic digestion			
Inputs			
Diesel	kg	1.3	Fisher (2006).
Electricity	kWh	20.6	Fisher (2006).
Process parameters			
Degradation ratio	$\%_{C\_bio\_and}$	70	Gallert and Winter (1997); Davidsson *et al.* (2007); Yoshida *et al.* (2012).
CH_4 content of biogas	$\%_{biogas}$	63	Smith *et al.* (2001); Møller *et al.* (2009); Banks *et al.* (2011); EASETECH database.

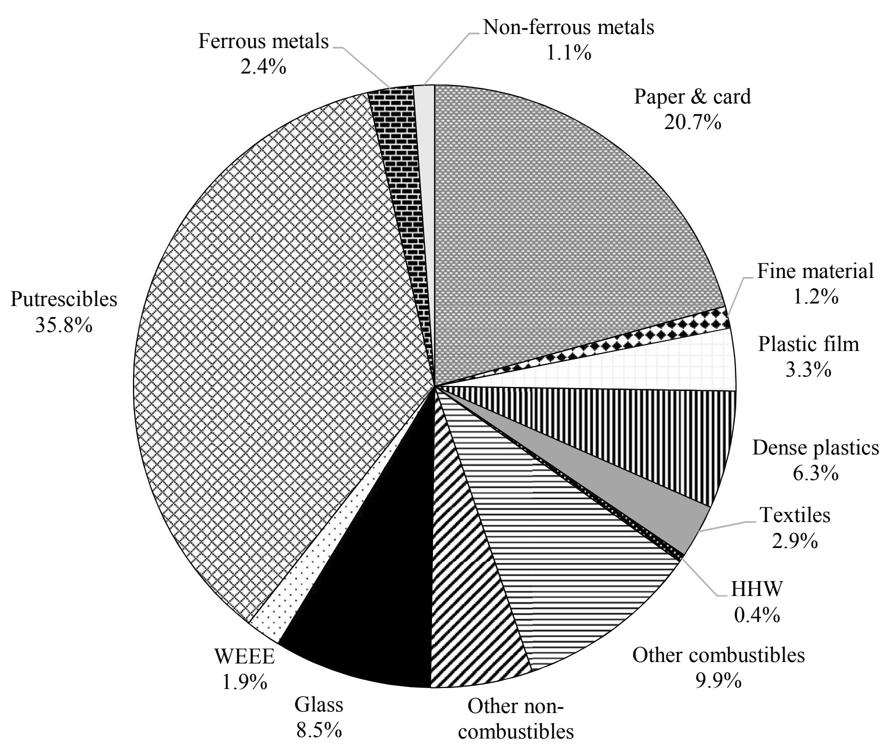

Fig. 1. Composition of LACW collected in Cardiff in 2012–2013 by primary material category (% of total LACW input mass flow). LACW, local authority collected waste; WEEE, waste electrical and electronic equipment; HHW, household hazardous waste.

and 3% used as alternative landfill daily cover (Horne et al., 2013). See Section 2.3.8 for details.

2.3.6. Composting

Two composting processes were modelled: in vessel composting (IVC) and open window composting (OWC). During waste reception, reject rates of 8% and 95% were applied for biodegradable waste and non biodegradable waste, respectively (Bernstad and la Cour Jansen, 2011). Details of process inputs and parameter values are presented in Table 2. The IVC process was assumed to be equipped with a biofilter. Biofilter CH_4 oxidation efficiency was modelled based on the mean of reported values in the literature, whilst the biofilter was assumed to have no effect on N_2O emissions due to a lack of consensus in the literature. A biofilter was not included in the OWC process as it is an open technology. Application of compost produced through IVC and OWC was modelled based on the average situation in the UK in 2012 as follows: for IVC derived compost, 68% to agricultural landspreading, 29% in horticulture/gardening, 2% as alternative landfill daily cover, and 1% for incineration; and for OWC derived compost, 73% to agricultural landspreading, 21% in horticulture/gardening, 5% as alternative landfill daily cover, and 1% for incineration (Horne et al., 2013). See Section 2.3.8 for details.

2.3.7. Mechanical biological treatment

Mechanical biological treatment (MBT) involves a combination of mechanical separation and biological treatment (either bio stabilisation/bio drying, IVC, or AD) (Defra, 2013b). The MBT facility in the investigated system comprised integrated IVC technology.

Table 2
LCI data for the IVC and OWC processes. Note that data are presented per tonne throughput.

	Unit	Value	
		IVC	OWC
Pre-treatment			
Inputs			
Diesel	l	0.12[a]	0.12[a]
Electricity	kWh	1.1[a]	1.1[a]
Aerobic composting			
Inputs			
Diesel	kg	0.26[a]	3[b,c,d]
Electricity	kWh	42.4[a]	0.51[b]
Lubricating oil	l	–	0.023[c]
Process parameters			
C degradation rates			
Paper and card & wood	$\%_{C\ bio}$	35[d]	35[e]
Food waste & fine material	$\%_{C\ bio}$	75[d]	75[e]
Garden waste	$\%_{C\ bio}$	50[d]	50[e]
N degradation rates			
Food waste & fine material	$\%_N$	70[e]	70[f]
Garden waste	$\%_N$	10[e]	10[f]
Other biodegradable waste	$\%_N$	25[e]	25[f]
Biofilter CH_4 oxidation efficiency	$\%_{CH_4}$	75[g]	–
Outputs			
Emissions to air			
CH_4	$g/kg_{Degraded\ C}$	2.4[h]	2.1[d]
N_2O	g/kg_N	1.8[h,i]	1.5[h]

[a] Source: Primary data from industrial source.
[b] Source: Fisher (2006).
[c] Source: Boldrin et al. (2009a).
[d] Source: Andersen et al. (2010).
[e] Source: Smith et al. (2001).
[f] Source: EASETECH database.
[g] Source: Based on the mean of reported values from Dalemo et al. (1997); du Plessis et al. (2003); Streese and Stegmann (2005); Brown et al. (2008); Boldrin et al. (2009b, 2011); Qiang et al. (2011); Yazdani et al. (2012).
[h] Source: Boldrin et al. (2009b).
[i] Source: Amlinger et al. (2008).

Based on primary data from a UK based MBT facility, process diesel and electricity consumption was assumed to be 0.32 l/t and 7.1 kWh/t, respectively. During mechanical separation, waste is sorted and classified to separate a fine, compostable fraction and two coarse fractions: one for refuse derived fuel (RDF) preparation and the other for disposal in landfill. Ferrous and non ferrous metals are recovered by overband magnets and eddy current separators, respectively, whilst plastics are recovered by near infra red optical separation. Waste transfer coefficients for the mechanical separation stage were derived from facility specific data from WasteDataFlow. For biodegradable waste, 28% is sent for composting, 64% is used as RDF, and 8% is sent for disposal in landfill. 100% of metals and plastics are recovered for recycling. 5% of non biodegradable waste is sent for composting and 95% is sent for disposal in landfill (Bernstad and la Cour Jansen, 2011). The biological treatment stage was modelled based the IVC process model described in Section 2.3.6. The MBT process produces a solid, residue called compost like output. Compost like output application was modelled based on the average situation in the UK in 2012, with 3% applied to agricultural land, 80% used as alternative landfill daily cover, and 17% disposed of in landfill (Horne et al., 2013). See Section 2.3.8 for details.

2.3.8. Application to land

Application to agricultural land was assumed to require use of a manure spreader, consuming 0.3 l/t and 0.45 l/t of diesel for compost/compost like output and digestate, respectively (Bernstad and la Cour Jansen, 2011; Boldrin et al., 2009b). The degradation and fate of carbon and nitrogen in solid residues after they have been applied to land was modelled using proxy data from Denmark (Bruun et al., 2006; Hansen, 2006b) and Europe (Smith et al., 2001), presented in Table 3. Solid residues applied to agricultural land were assumed to partially substitute for commercial fertilisers, with the amounts substituted calculated based on the solid residue nutrient content and a substitution rate. Solid residues used in gardening/horticulture were assumed to substitute for peat (Smith et al., 2001). Details of fertiliser and peat substitution rates applied are presented in Table 3.

2.3.9. Material recovery facility

Two material recovery facility (MRF) processes were modelled: single stream commingled materials MRF and residual waste MRF. The single stream commingled materials MRF process comprises a series of mechanical separation activities designed to recover the following product streams: mixed container glass, paper, card, high density polyethylene bottles, polyethylene terephthalate bottles, mixed plastics, plastic film, steel cans, and aluminium cans. A residual waste stream is also produced and sent for landfilling. Electricity and diesel consumption of 35 kWh/t and 2 kg/t, respectively, was assumed for the single stream commingled materials MRF process, based on measured data from UK based MRFs reported by Eunomia (2011). Details of product stream material composition are provided in Appendix D. Due to a lack of specific data, electricity consumption for the residual waste MRF process was assumed, based on Pressley et al. (2015), to be approximately 25% more than that of a single stream commingled materials MRF. Hence, an electricity consumption of 44 kWh/t was assumed. Diesel consumption was based on the single stream commingled materials MRF process. The mass of materials recovered at residual waste MRFs was modelled using facility specific data from WasteDataFlow.

2.3.10. Refuse derived fuel production

The RDF production process was modelled based on a theoretical facility with a gas fired drying stage. Electricity consumption of

Table 3
LCI data for the application to land process. Note that data are presented per tonne throughput.

	Unit	Value	
		Compost/CLO	Digestate
Process parameters			
Carbon bound in soil[a]			
Food waste-derived output	$\%_{C\ bio}$	8.2[b]	8.2[b,c]
Garden waste-derived output	$\%_{C\ bio}$	14[d]	–
Mixed organic waste-derived output[d]	$\%_{C\ bio}$	12[b,d]	12[b,c,d]
Fertiliser substitution rates			
Nitrogen fertiliser	$t/t_{N\ applied}$	0.2[f]	0.48[g]
Phosphorous fertiliser	$t/t_{P\ applied}$	0.5[g]	0.5[g]
Potassium fertiliser	$t/t_{K\ applied}$	0.8[g]	0.8[g]
Peat substitution rate	m^3	1.47[h]	1.47[h]
Outputs			
Emissions to air			
N_2O	g/kg_N	1.4[d]	1.4[d]
$CO_{2\ biogenic}$			
Food waste-derived output	$g/kg_{C\ bio}$	91.8[b]	91.8[b,c]
Garden waste-derived output	$g/kg_{C\ bio}$	86[d]	–
Mixed organic waste-derived output[e]	$g/kg_{C\ bio}$	88[b,d]	88[b,c,d]

[a] Relevant for the sensitivity analysis only.
[b] Source: Smith et al. (2001).
[c] Source: Møller et al. (2009).
[d] Source: Bruun et al. (2006).
[e] Based on an assumed composition of 1/3 food waste and 2/3 garden waste.
[f] Source: Hansen (2006a).
[g] Source: Evangelisti et al. (2014).
[h] Based on assumed bulk densities of peat and compost/CLO/digestate of 0.3 t m^{-3} and 0.68 t m^{-3}, respectively (Smith et al., 2001), and an assumed substitution rate of 1 $m^3_{peat}\ m^{-3}_{compost/CLO/digestate}$ (Boldrin et al., 2010).

40 kWh/t of RDF produced was assumed (Burnley et al., 2011). Process RDF yield was based on facility specific data from WasteDataFlow.

2.3.11. Merchant/exporter

The merchant and exporter processes were modelled using the transfer station process as a proxy.

2.3.12. Incineration

The incineration process was modelled based on an average grate furnace incinerator – the most common incinerator type worldwide (Defra, 2013a) – with dry flue gas scrubbing and de NO_x technology, as described by Boesch et al. (2014). Details of energy and material inputs and waste characteristic transfer coefficients for the incineration process are presented in Table 4 and Table 5, respectively. Thermal energy released during the incineration process is recovered via a boiler and a fully condensing turbine and transformed into electrical energy by a generator at an assumed gross generation efficiency of 22% (EC, 2006; Defra, 2013a). A 2% electrical energy loss was included to account for part load operation and heat transfer losses (Consonni and Viganò, 2011).

Non combustible solid residues (i.e. bottom ash and fly ash) are discharged from the base of the furnace and cooled. Fly ash was assumed to be utilised as a backfilling material, with electricity and diesel consumption of 29 kWh/t and 1.5 l/t, respectively, assumed for the process (Fruergaard et al., 2010). Bottom ash is sent for de scrapping to recover metals for recycling. The de scrapping process was modelled using data from Bosech et al. (2014). 97% of incinerator bottom ash is sent for de scrapping, with the remaining 3% sent to landfill. Recovery rates of 30%, 80%, and 30% were assumed for aluminium, steel, and copper, respectively, with non recovered material sent to landfill. 3 kWh/t of electricity is required for de scrapping process.

Table 4
Energy and material inputs for the incineration process. Note that data are presented per tonne throughput.

	Unit	Value	Reference
Inputs			
Water	m^3	1	Astrup et al. (2009); Boesch et al. (2014)
Electricity	kWh	142	EC (2006); Boesch et al. (2014)
Heat, heavy fuel oil	MJ	240	Boesch et al. (2014)
Ammonia	kg	0.5	Astrup et al. (2009); EA (2010)
Lime	kg	10	EA (2010)
Sodium hydroxide	kg	0.19	Astrup et al. (2009); EA (2010)
Activated carbon	kg	0.25	EA (2010)

2.3.13. Landfill

The non hazardous landfill process (also used as a proxy for inert landfill) was modelled based on an average UK medium sized conventional landfill with gas utilisation. Emissions were modelled over a 100 year time period. Waste material decay rates were taken from IPCC (2006). Energy and material inputs are presented in Table 6. Details of the gas management system and its performance over time are presented in Table 7. Landfill gas recovery efficiencies were averaged over the 100 year time period to better reflect an average tonne of waste deposited, rather than the first mass of waste deposited (see Table 8). Collected landfill gas is utilised in an internal combustion engine to generate electricity, with a gross efficiency of 32% assumed (Patterson et al., 2011). A 1% fugitive CH_4 emissions rate was specified for gas utilisation processes (US EPA, 2011).

2.3.14. Material reprocessing

Reprocessing of different dry recyclables was modelled using the process models and LCI data presented in Turner et al. (in press).

2.3.15. Reuse

2.3.15.1. Books and bric a brac. Books and bric a brac were each assumed to be resold in charity shops. An electricity input at the charity shop of 357 kWh/t was assumed (James et al., 2011). No material loss or avoided production was assumed.

2.3.15.2. Paint. No energy or material inputs were included for the reuse of waste paint process. Post reuse, the steel container (200 kg/t) was assumed to be sent for reprocessing. Reused waste paint was assumed to substitute for primary paint production. No material quality loss was assumed. Data for the production of primary paints were adapted for the UK situation from a study undertaken in Abu Dhabi (Nayak and Kumar, 2008). Assuming a paint density of 1 kg/litre, inputs of 33 kWh/t of electricity and 200 kg/t of steel (Saft, 2007) were assumed for the primary paint production process.

2.3.15.3. Textiles and footwear. The reuse of textiles and footwear process comprises three stages: 1) sorting; 2) transport to end markets; and 3) reuse. Data for the sorting process were taken from the a UK based textiles recovery plant (EA, 2010). Based on Bartlett et al. (2013), it was assumed that 37% of sorted textiles and footwear are reused domestically, whilst 63% are exported for reuse abroad. Based on the market situation in the year 2008, we assumed that 50% of textiles and footwear exported for reuse abroad are sent to Eastern Europe, 17% to South Asia, and 33% to Sub Saharan Africa (McGill, 2009). Textiles and footwear sent for reuse domestically were assumed to be resold in charity shops (see Section 2.3.15.1). No energy or material inputs were included where textiles and footwear are exported. Reused textiles and footwear were assumed to substitute for the production of textiles from

Table 5
Material characteristic transfer coefficients (%) for the incineration process.

	Bottom ash (%)	Fly ash (%)	Emissions to air (%)	Reference
Ash	87	13	–	EASETECH database
Volatile solids	1	–	–	EASETECH database
Energy	1	–	–	–
Fossil C	–	–	100	EASETECH database
Biogenic C	1	–	99	Boesch et al. (2014)
N	1	–	99	Boesch et al. (2014)
Fe	99	1	–	Koehler et al. (2011); Boesch et al. (2014)
Al	82	18	–	Koehler et al. (2011); Boesch et al. (2014)
Cu	95	5	–	Koehler et al. (2011); Boesch et al. (2014)

Table 6
LCI data for the non-hazardous landfill process. Note that data are presented per tonne throughput.

Inputs	Unit	Quantity	Reference
Diesel	kg	1.8	Hall et al. (2005)
Electricity	kWh	8	Manfredi et al. (2009)
Water	kg	0.00038	Hall et al. (2005)
HDPE (liner)	kg	1	Hall et al. (2005)
Gravel	kg	100	Manfredi et al. (2009)
Steel	kg	0.12	Hall et al. (2005)
Synthetic rubber	kg	0.0011	Hall et al. (2005)
Lubricant	kg	0.0089	Hall et al. (2005)

HDPE, high-density polyethylene.

Table 7
Technical measures and performance associated with landfill gas recovery, utilisation, and oxidation for different time periods.

	Period 1	Period 2	Period 3	Period 4	Period 5
Duration (years)	1	4	15	30	50
CH_4 oxidation (%)	10[a]	10[a]	20[a]	36[b]	36[b]
Gas collected (%)	0[a]	50[a]	75[a]	85[a]	0[c]
Gas management	None	Flare	ICE	ICE	None

ICE, internal combustion engine.
[a] Source: US EPA (2011).
[b] Source: Chanton et al. (2009).
[c] Source: Spokas et al. (2006).

virgin fibres (for details, see Appendix A). Due to their comparatively shorter lifetimes and older fashions, market substitution rates of 60%, 75%, and 85% were assumed for textiles and footwear reused in the UK, Eastern Europe, and South Asia and Sub Saharan Africa, respectively (Farrant et al., 2010; McGill, 2009).

2.3.15.4. Large domestic appliances. Waste electrical and electronic equipment large domestic appliances (LDA) were assumed to be reused domestically. However, based on James et al. (2011), it was assumed that 35% of LDA send for reuse are actually recycled, due to insufficient quality. No energy or material inputs were included for the LDA reuse process. Reused LDA were assumed to substitute for the production of primary LDA (for details, see Appendix A). Due to the comparatively shorter lifetimes of reused LDA compared to new LCA, a market substitution rate of 50% was assumed (James et al., 2011).

2.4. Life cycle impact assessment

GHG emissions were characterised by Global Warming Potential (GWP) using a 100 year time period (expressed as t CO_2e), with characterisation factors taken from the Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment Report (Forster et al., 2007). The emissions “savings” from biogenic carbon sequestration in landfills or soils (see Christensen et al., 2009) were excluded. This assumption was investigated as part of the sensitivity analysis.

3. Results and discussion

3.1. Material flow analysis

Comprehensive material flow diagrams for the LACW management systems of each scenario (i.e. the results of the mass balance of waste) are presented in Appendix C. For S1, the baseline scenario, the dominant primary waste stream was residual waste (82,079 t), followed by commingled materials (42,065 t). 63,518 t of waste from the residual waste stream was sent directly to landfill. Of the remaining waste, 18,562 t was sent to one of three residual waste MRFs, with 9,829 t sorted and sent for reprocessing (the remainder was sent to landfill), of which 7,843 t was reprocessed into a secondary product; an overall recycling rate of 42.3% (note that rates varied between facilities). By comparison, all commingled materials were sent to single stream commingled MRFs, with 34,871 t recovered and sent for

Table 8
Temporally-averaged landfill gas collection efficiencies.

Time period duration (years)	Percentage landfill gas collected	Percentage landfill gas not collected
1	0	100
1	35	65
1	50	50
1	65	35
1	70	30
11	75	25
1	77	23
1	79	21
1	81	19
1	83	17
30	85	15
50	0	100

Source: adapted from US EPA (2011).

reprocessing and 27,920 t ultimately reprocessed into a secondary product. This represents an overall single stream commingled MRF recycling rate of 80.0% (note, again, that rates varied between facilities). These results support those of Pressley et al. (2015), who found that recovery rates at single stream commingled MRFs were higher than those at residual waste MRFs. This suggest that policies aiming to increase recycling rates should focus on improving sorting efficiencies at the waste producer level (e.g. households), rather than on recovering materials from the residual waste stream.

Table 9 displays a selection of key indicator results from the MFAs for each scenario. For S1, a reuse/recycling/composting/AD (henceforth, recycling rate) of 49.8% was calculated for the baseline scenario, marginally higher than the 52.8% value reported by Cardiff County Council (StatsWales, 2015b). However, where rejects from reprocessing and biological treatment are included the final (adjusted) recycling rate was estimated as being 45.2%, highlighting the importance of considering downstream reject rates when calculating recycling rates. The recycling rate was highest for S2 (enhanced food waste capture), where high rates of food waste diversion were achieved, whilst the recycling was also higher for S4 (enhanced dry recycling) compared to S1 due to the increased recovery of dry recyclables. For S1, over half of LACW was ultimately landfilled, with 37.7% of LACW sent directly to landfill and 16.4% rejected downstream (Table 9). S3 (enhanced incineration) was found to result in the highest heat and power recovery rate, whilst it was the most successful scenario in terms of diverting material from landfill.

The results of the MFA suggest that it may not be possible for Wales to achieve its recycling rate target of 70% by 2025 given current waste arisings, even if optimum dry recycling and food waste diversion rates are achieved. Rather, it is likely that significant reductions in residual waste arisings will be required, as well as improvements in recycling rates (Timlett and Williams, 2011). Wales have already made progress in this respect, with a steady rise in the national LACW recycling rate from 37.5% in 2008–2009 to 52.3% in 2012–2013, accompanied by a 9.9% reduction in LACW arisings over the same period (StatsWales, 2015a). However, reductions in total waste arisings have stagnated in recent years. In order for Wales to continue progressing towards its recycling targets, further reductions in waste arisings are likely to be required.

3.2. Life cycle interpretation

3.2.1. Evaluation of the existing LACW management system

Fig. 2 shows the total GHG emissions per scenario and the contribution by primary waste stream. S1, the baseline scenario, was found to result in a net GHG burden (8009 t CO_2e). Note that positive values represent an environmental load (i.e. GHG burden), whilst negative values represent an environmental saving (i.e. GHG benefit). GHG benefits were observed for the management of commingled materials (−9410 t CO_2e), source segregated dry recyclables (−3212 t CO_2e), and source segregated food waste (−2596 t CO_2e). However these benefits were heavily outweighed by the GHG burdens from residual waste management (21,778 t CO_2e).

Contribution to total primary waste stream GHG impacts by process type for each scenario is shown in Fig. 3. For S1, the dominant contribution in terms of GHG burdens was landfilling of residual waste (25,866 t CO_2e). Material reprocessing & reuse was the only process type that consistently resulted in GHG benefits. For S1, the most significant source of GHG benefits was from reprocessing of commingled materials (−19,219 t CO_2e). The contribution of biological treatment to total GHG impacts was found to be negative for the source segregated food waste stream (i.e. GHG benefit) but positive for the source segregated garden waste stream (i.e. GHG burden). This is due to the different types of biological treatment used to manage each waste stream (see Appendix C). Food waste was treated predominately by AD, a process that utilises produced biogas to generate electricity that is exported to the National Grid and substitutes for electricity generated from CCGT, resulting in GHG benefits. By comparison, garden waste was largely treated by composting (either IVC or OWC), through which energy cannot be recovered. This finding correlates with those of Yoshida et al. (2012) and Bernstad and la Cour Jansen (2011) who each found that AD of organic waste resulted in a greater GHG benefit compared to composting.

Overall, transport was identified as the third largest contributor to the total GHG impacts for S1 (Fig. 3; 7296 t CO_2e; 10% contribution). However, its relative contribution to the GHG impacts of each primary waste stream management system was highly variable. Whilst the contribution of transport to the GHG impacts from the residual waste (1492 t CO_2e; 4% contribution) and source segregated dry recyclables (160 t CO_2e; 3% contribution) management systems was minor, transport contributed substantially to the GHG impacts from source segregated garden waste management (565 t CO_2e; 24% contribution). This is likely due to the considerable distances that garden waste was transported for treatment. For example, 16,780 t of collected garden waste was transported between 78 and 181 km to an IVC facility, resulting in substantial GHG burdens. The contribution of transport to the total GHG impacts of commingled materials management was also found to be large (4543 t CO_2e; 16% contribution). This is likely due to the large quantity of recyclate that was exported overseas for reprocessing. Of the 42,055 t of commingled materials collected, 11,334 t

Table 9
Comparison of MFA indicator results for the four scenarios. Note that S1, S2, S3, and S4 refer to Scenarios 1–4, respectively.

Indicator	Description	Scenario			
		S1	S2	S3	S4
Landfill rate (%)[a]	Total tonnage of LACW sent directly to landfill plus total tonnage rejected from other facilities, divided by total tonnage of LACW arisings	54.0	48.0	29.0	52.3
Reuse/recycling/composting/AD rate (%)[a]	Sum of tonnage of LACW sent for reuse, reprocessing, composting, or AD, divided by the total tonnage of LACW arisings	52.8	59.5	52.8	55.1
Heat and power recovery rate (%)[a]	Sum of tonnage of LACW sent for incineration or AD, divided by total tonnage of LACW arisings	8.9	15.4	38.5	8.9
Landfill diversion rate (%)	Sum of tonnage of LACW not sent directly to landfill (i.e. LACW sent to reuse/recycling/composting/AD/thermal treatment), divided by total tonnage of LACW	62.3	68.9	92.0	64.8
Adjusted reuse/recycling/ composting/AD rate (%)	Sum of tonnage of LACW sent for reuse, reprocessing, composting, or AD minus sum of tonnage rejected, divided by the total tonnage of LACW arisings	45.2	50.8	44.4	47.0

MFA, material flow analysis; LACW, local authority collected waste; AD, anaerobic digestion.
[a] Waste management performance indicators for Welsh local authorities.

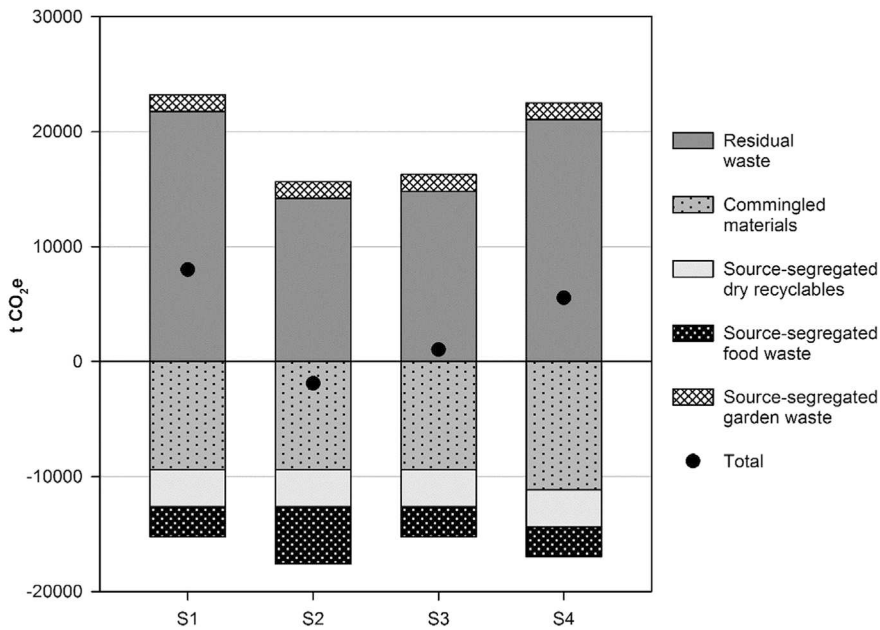

Fig. 2. LCA results (GHG emissions, t CO_2e/FU) of different scenarios and the contribution by primary waste stream. Note that S1, S2, S3, and S4 refer to Scenarios 1–4, respectively. LCA, life cycle assessment; GHG, greenhouse gas; FU, functional unit.

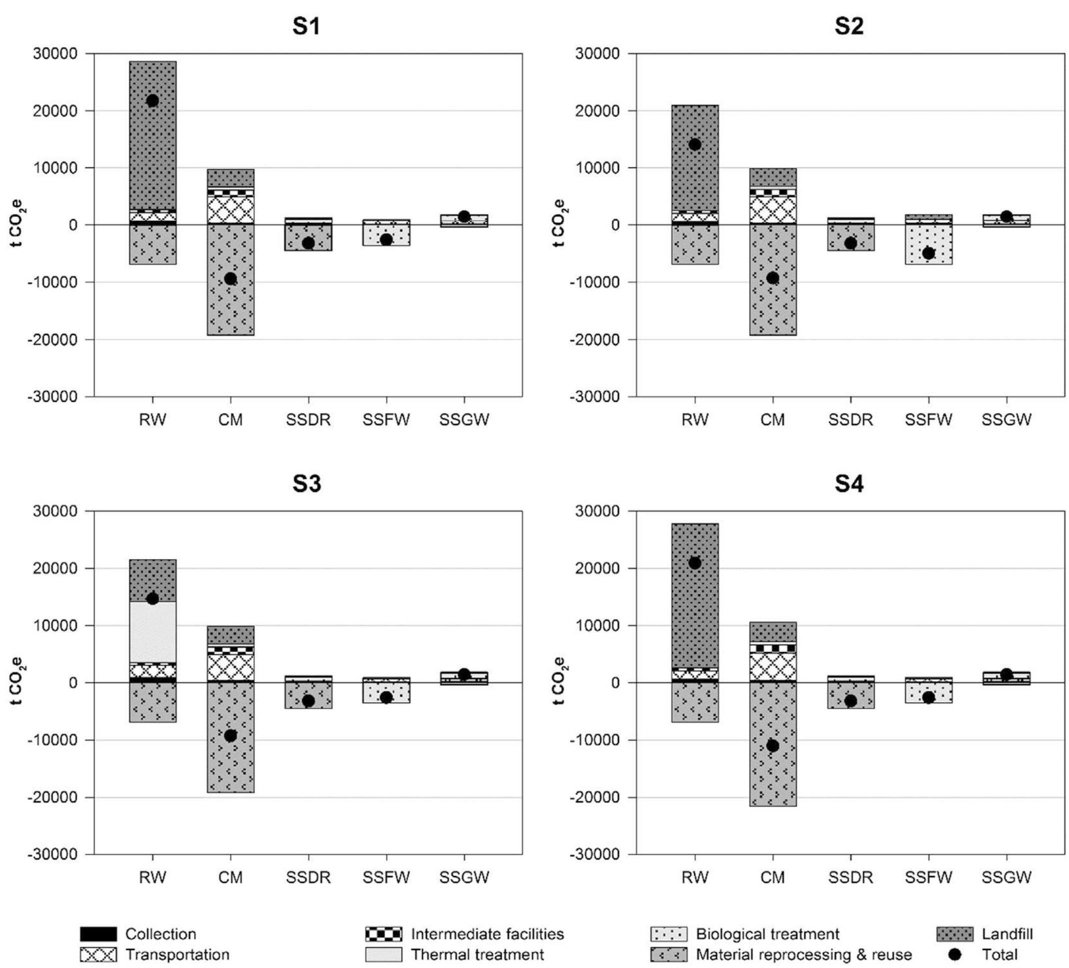

Fig. 3. LCA results (GHG emissions, t CO_2e/FU) of different scenarios per primary waste stream and the contribution process type. Note that S1, S2, S3, and S4 refer to Scenarios 1–4, respectively. RW, residual waste; CM, commingled materials; SSDR, source-segregated dry recyclables; SSFW, source-segregated food waste; SSGW, source-segregated garden waste; LCA, life Cycle Assessment; GHG, greenhouse gas; FU, functional unit.

(predominately paper) was exported to reprocessing facilities in Asia (see Appendix C). These results suggests that the calculated GHG impacts from materials recycling in this situation are more greatly affected by transport distance compared to results from previous studies (Larsen et al., 2009; Merrild et al., 2012; Salhofer et al., 2007). The formulation and implementation of policy measures that promote domestic reprocessing of waste materials may help to reduce the GHG impacts of transport and, consequently, enhance the overall GHG benefits from materials recycling.

Table 10 presents the contributions of wastes from different collection sources to the total GHG impacts for each scenario. For S1, the largest contribution to the total GHG impacts was from household kerbside collected waste, particularly residual waste, which was the most significant source of GHG burdens, and commingled materials, the largest source of GHG benefits. Overall GHG impacts from bulky waste collection, fly tipped materials, the voluntary sector, and municipal parks/grounds waste collection were found to be minor, with each contributing less than 1% to the total GHG impacts. An interesting result from a policy perspective concerns the contribution of street cleaning waste management to the total GHG impacts. Street cleaning waste management was found to contribute 8.8% (751 t CO_2e) of total GHG impacts for S1. Waste from street cleaning comprised both residual waste, the management of which incurred a GHG burden of 2269 t CO_2e, and commingled materials, the management of which resulted in a GHG saving of −1518 t CO_2e. Diversion of street cleaning waste from the residual waste stream by promoting recycling is an area not commonly targeted by policy makers, but could be an effective source of potential GHG benefits.

3.2.2. Comparison of scenarios

Compared to S1, each of the three alternative scenarios were found to result in lower overall GHG burdens (Fig. 2). S2 was found to be the best performer with an overall GHG impact of −1930 t CO_2e, the only scenario for which a net GHG benefit was found. The contribution of household kerbside collection residual waste to total GHG impacts was substantially reduced in S2 compared to S1 (Table 10), with a large portion of the GHG burdens of residual waste management replaced by GHG benefits from the management of source separated food waste. This highlights the considerable influence of the organic content of residual waste on total GHG burdens (see Section 3.2.3.2).

After S2, S3 was the next best performer (1046 t CO_2e; Fig. 2). S3 showed the greatest reduction in GHG burdens from landfill compared to S1 (Fig. 3). For S3, the substantial GHG burdens from landfill were replaced by comparatively lower GHG burdens from incineration (Fig. 3), suggesting that incineration represents a better option with regards to GHG impacts compared to landfill.

Table 10
LCA results (GHG emissions, t CO_2e) and contribution analysis (%t CO_2e/FU) of different scenarios per collection source and primary waste stream. Note that S1, S2, S3, and S4 refer to Scenarios 1–4, respectively.

	Scenario							
	S1		S2		S3		S4	
	t CO_2e	% t CO_2e/FU	t CO_2e	% t CO_2e/FU	t CO_2e	% t CO_2e/FU	t CO_2e	% t CO_2e/FU
Household kerbside collection								
Residual waste	17,457	40.3	9874	25.9	11,645	32.7	16,738	37.8
Commingled materials	−7545	17.4	−7545	19.8	−7545	21.2	−9287	21.0
Source-segregated garden waste	1573	3.6	1573	4.1	1573	4.4	1573	3.6
Source-segregated food waste	−2596	6.0	−4954	13.0	−2596	7.3	−2596	5.9
Source-segregated dry recyclables	15	0.0	15	0.0	15	0.0	15	0.0
Total	8904	67.4	−1,0367	63.0	3092	65.7	6443	68.2
Bulky waste collection								
Source-segregated garden waste	2	0.0	2	0.0	2	0.0	2	0.0
Source-segregated dry recyclables	−72	0.2	−72	0.2	−72	0.2	−72	0.2
Total	−70	0.2	−70	0.2	−70	0.2	−70	0.2
HWRCs								
Residual waste	−2084	4.8	−2084	5.5	−1709	4.8	−2084	4.7
Commingled materials	−319	0.7	−319	0.8	−319	0.9	−319	0.7
Source-segregated garden waste	192	0.4	192	0.5	192	0.5	192	0.4
Source-segregated dry recyclables	−2560	5.9	−2560	6.7	−2560	7.2	−2560	5.8
Total	−4771	11.9	−4771	13.5	−4396	13.4	−4771	11.6
Bring sites								
Commingled materials	−28	0.1	−28	0.1	−28	0.1	−28	0.1
Source-segregated dry recyclables	−595	1.4	−595	1.6	−595	1.7	−595	1.3
Total	−623	1.4	−623	1.6	−623	1.8	−623	1.4
Street cleaning								
Residual waste	2270	5.2	2270	6.0	2017	5.7	2270	5.1
Commingled materials	−1518	3.5	−1518	4.0	−1518	4.3	−1518	3.4
Total	752	8.8	752	10.0	499	9.9	752	8.6
Fly-tipped materials								
Residual waste	208	0.5	208	0.5	237	0.7	208	0.5
Total	208	0.5	208	0.5	237	0.7	208	0.5
Non-household collection								
Residual waste	3926	9.1	3926	10.3	2624	7.4	3926	8.9
Total	3926	9.1	3926	10.3	2624	7.4	3926	8.9
Voluntary sector/community collection								
Source-segregated dry recyclables	0	0.0	0	0.0	0	0.0	0	0.0
Total	0	0.0	0	0.0	0	0.0	0	0.0
Municipal parks/grounds waste collection								
Source-segregated garden waste	−317	0.7	−317	0.8	−317	0.9	−317	0.7
Total	−317	0.7	−317	0.8	−317	0.9	−317	0.7
Total	8008	100.0	−1930	100.0	1046	100.0	5548	100.0

GHG, greenhouse gas; FU, functional unit; LCA, Life Cycle Assessment; HWRC, household waste recycling centre; LACW, local authority collected waste.

This compares favourably with the findings of previous studies (e.g. Arena et al., 2003; Fernández-Nava et al., 2014; Massarutto et al., 2011). The GHG benefits of S3 would likely be further enhanced if district heating networks were more widely established in the UK. This would create markets for thermal energy recovered through incineration, leading to further GHG benefits through avoided thermal energy production (Giugliano et al., 2011; Turconi et al., 2011).

Of the three alternative scenarios, S4 was found to result in the improvement on the results of the baseline scenario, with a total GHG impact of 5548 t CO_2e (Fig. 2). This is likely due to the relatively high dry recyclables sorting efficiencies that are already achieved in Cardiff, particularly for mixed glass, paper, and dense plastics. Considering the results for S2, this finding also suggests that waste policies that target residual waste prevention may be more effective in achieving GHG emissions reductions compared to those that target greater recovery of dry recyclables.

3.2.3. Scenario analysis

3.2.3.1. Carbon sequestration. A sensitivity analysis was performed to investigate the impact of including carbon sequestration on the LCA results through scenario analysis. For each scenario, the amount of carbon sequestered was calculated as the amount of biogenic carbon that, after 100 years, remains in a landfill or, in the case landspreading of organic solid residues from biological treatment, is bound to soil. Fig. 4 shows the total GHG impact for each scenario and the contribution by process type. The inclusion of carbon sequestration had a substantial impact on the calculated results. Overall, net GHG benefits were observed for each of the four scenarios. Landfilling was found to result in overall GHG benefits, effectively rendering landfills a “carbon sink”. This contradicts the findings of the UK's national GHG inventory return for GHG emissions from SWM (Salisbury et al., 2015), despite the fact that carbon sequestration is also considered in the national inventory analysis (Hogg et al., 2011).

The ranking of scenarios in terms of their environmental performance was affected by the choice of including carbon sequestration. Whilst S2 remained the best option and S4 was still found to result in additional GHG benefits compared to S1, the overall GHG burdens for S3 were found to be considerably higher than S1 (Fig. 4). This suggests that when carbon sequestration is included in the calculations, landfilling is generally a better option compared to incineration in terms of GHG impact, which contradicts the results of previous studies (e.g. Arena et al., 2003; Fernández-Nava et al., 2014; Massarutto et al., 2011). Given the significant effects of including carbon sequestration in the LCA on the calculated results, these authors suggest that further research into carbon degradation rates for waste materials in UK landfills is needed.

3.2.3.2. Residual waste composition. A sensitivity analysis using scenario analysis was performed to test the influence of the modelled organic content of the residual waste stream on the overall LCA results. In this study, residual waste composition data were taken from a national waste composition study (WRAP, 2010) that predates the introduction of separate household food waste collection in Cardiff. The use of these data may, therefore, result in an overestimation of GHG burdens as the total organic content of the input waste may be overestimated. Two scenarios were analysed, with the proportion of organic material in residual waste increased and decreased by 10% compared to the baseline values for each scenario. Compared to the total GHG impact of S1 (8008 t CO_2e), the 10% increase in organic content resulted in a +15.5% variation (result: 9254 t CO_2e), whilst the 10% decrease resulted in a −9.4% variation (result: 6826 t CO_2e). The sensitivity analysis shows that the overall LCA results are highly sensitive to the organic content of residual waste.

4. Conclusions and recommendations

In this paper we have presented an approach that combines the systematic methodologies of MFA and LCA to quantitatively evaluate the environmental performance of large and complex SWM systems. The approach was applied to evaluate the GHG emissions performance of a local authority SWM system and compare its performance with those of alternative systems to assess the potential effectiveness of different waste policy measures. The results of the MFA show that S2 (enhanced food waste capture) resulted in the highest recycling rate of the investigated scenarios, whilst S3 (enhanced incineration) resulted in the highest rate of waste diversion from landfill. Overall, our results suggest that, in this case, the national government is unlikely to achieve its recycling targets, even if each of the assessed policy measures are implemented optimally. It is likely that in order for the recycling targets to be achieved, these policy measures need to be combined with further policies focused on waste prevention.

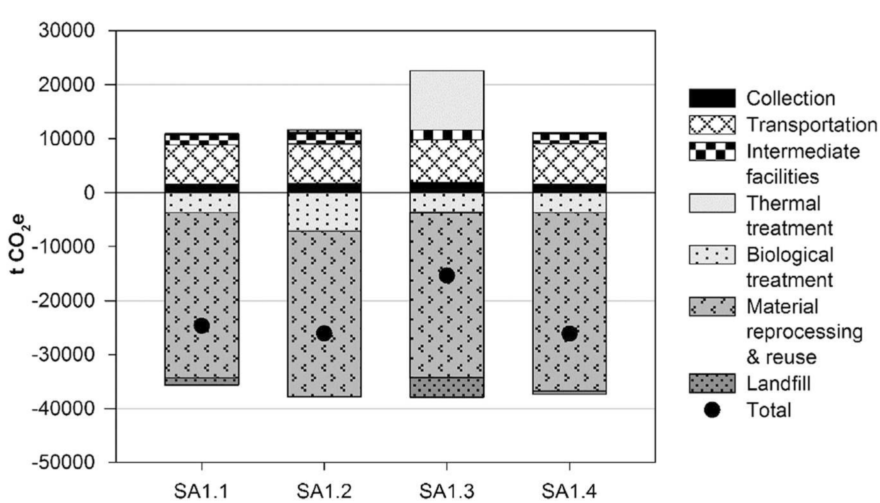

Fig. 4. Sensitivity analysis (SA1) results (GHG emissions, t CO_2e/FU) of different scenarios and the contribution by process type – the inclusion of carbon sequestration. Note that SA1.1, SA1.2, SA1.3, and SA1.4 refer to Scenarios 1–4, respectively. GHG, greenhouse gas; FU, functional unit.

The GHG impacts of the existing SWM system were evaluated through LCA, focusing on potential climate change impacts. The results showed that the dominant source of GHG burdens was from residual waste management, principally through landfilling, which was the most significant source of GHG burdens. Material reprocessing and AD were found to be the greatest sources of GHG benefits. GHG burdens from transport were found to be generally minor, except with regards to the management of source segregated garden waste and commingled materials.

Compared to the baseline scenario, each of the three alternative scenarios were found to perform better in terms of GHG impact. Overall, the best performing scenario was S2 (enhanced food waste capture), which was the only scenario that resulted in an overall GHG saving. However, the results of the LCA were found to be highly sensitive to the choice of excluding GHG benefits from carbon sequestration. When carbon sequestration was included, all four scenarios were found to result in net GHG benefits. The order of the four scenarios in terms of overall performance was also found to be affected by the choice, with the GHG burdens for S3 (enhanced incineration) found to be greater than those of the baseline scenario. The results were also found to be sensitive to the organic content of the residual waste stream.

Overall, this paper has demonstrated that the complementary methodologies of MFA and LCA can be used in combination to provide valuable information about the environmental performance of a SWM system. The approach can be applied to assist national governments in appraising existing and possible waste policies and to support local waste managers in identifying environmentally optimal SWM strategies.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.04.077.

References

Allegrini, E., Butera, S., Kosson, D.S., Van Zomeren, A., Van der Sloot, H.A., Astrup, T.F., 2015. Life cycle assessment and residue leaching: the importance of parameter, scenario and leaching data selection. Waste Manag. 38, 474–485. http://dx.doi.org/10.1016/j.wasman.2014.12.018.

Amlinger, F., Peyr, S., Cuhls, C., 2008. Greenhouse gas emissions from composting and mechanical biological treatment. Waste Manag. Res. 26, 47–60. http://dx.doi.org/10.1177/0734242X07088432.

Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2010. Mass balances and life-cycle inventory for a garden waste windrow composting plant (Aarhus, Denmark). Waste Manag. Res. 28, 1010–1020. http://dx.doi.org/10.1177/0734242X10360216.

Arena, U., Di Gregorio, F., 2014. A waste management planning based on substance flow analysis. Resour. Conserv. Recycl. 85, 54–66. http://dx.doi.org/10.1016/j.resconrec.2013.05.008.

Arena, U., Mastellone, M.L., Perugini, F., 2003. The environmental performance of alternative solid waste management options: a life cycle assessment study. Chem. Eng. J. 96, 207–222. http://dx.doi.org/10.1016/j.cej.2003.08.019.

Astrup, T., Moller, J., Fruergaard, T., 2009. Incineration and co-combustion of waste: accounting of greenhouse gases and global warming contributions. Waste Manag. Res. 27, 789–799. http://dx.doi.org/10.1177/0734242X09343774.

Banks, C.J., Cheshire, M., Heaven, S., Arnold, R., 2011. Anaerobic digestion of source-segregated domestic food waste: performance assessment by mass and energy balance. Bioresour. Technol. 102, 612–620. http://dx.doi.org/10.1016/j.biortech.2010.08.005.

Bartlett, C., McGill, I., Willis, P., 2013. Textiles Flow and Market Development Opportunities in the UK, Waste & Resources Action Programme, Banbury, UK. Available from: http://www.wrap.org.uk/content/uk-textile-product-flow-and-market-development-opportunities.

Bernstad, A., la Cour Jansen, J., 2011. A life cycle approach to the management of household food waste – a Swedish full-scale case study. Waste Manag. 31, 1879–1896. http://dx.doi.org/10.1016/j.wasman.2011.02.026.

Blengini, G.A., Fantoni, M., Busto, M., Genon, G., Zanetti, M.C., 2012. Participatory approach, acceptability and transparency of waste management LCAs: case studies of Torino and Cuneo. Waste Manag. 32, 1712–1721. http://dx.doi.org/10.1016/j.wasman.2012.04.010.

Boesch, M.E., Vadenbo, C., Saner, D., Huter, C., Hellweg, S., 2014. An LCA model for waste incineration enhanced with new technologies for metal recovery and application to the case of Switzerland. Waste Manag. 34, 378–389. http://dx.doi.org/10.1016/j.wasman.2013.10.019.

Bogner, J., Abdelrafie Ahmed, M., Diaz, C., Faaij, A., Gao, Q., Hashimoto, S., Mareckova, K., Pipatti, R., Zhang, T., 2007. Waste management. In: Metz, B., Davidson, O.R., Boesch, P.R., Dave, R., Meyer, L.A. (Eds.), Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the International Panel on Climate Change. Cambridge University Press, Cambridge, UK and New York, NY, USA.

Boldrin, A., Andersen, J.K., Christensen, T.H., 2009a. Environmental Assessment of Garden Waste Management in Århus Kommune. Technical University of Denmark, Copenhagen, Denmark. Available from: https://www.aarhus.dk/~/media/Subsites/AVA/Om-AVA/Bibliotek/Rapporter-og-undersoegelser/Aarhus-haveaffald-engelsk-rapport.pdf.

Boldrin, A., Andersen, J.K., Moller, J., Christensen, T.H., Favoino, E., 2009b. Composting and compost utilization: accounting of greenhouse gases and global warming contributions. Waste Manag. Res. 27, 800–812. http://dx.doi.org/10.1177/0734242X09345275.

Boldrin, A., Hartling, K.R., Laugen, M., Christensen, T.H., 2010. Environmental inventory modelling of the use of compost and peat in growth media preparation. Resources. Conserv. Recycl. 54, 1250–1260. http://dx.doi.org/10.1016/j.resconrec.2010.04.003.

Boldrin, A., Andersen, J.K., Christensen, T.H., 2011. Environmental assessment of garden waste management in the Municipality of Aarhus, Denmark. Waste Manag. 31, 1560–1569. http://dx.doi.org/10.1016/j.wasman.2011.01.010.

Briffaerts, K., Spirinckx, C., Van der Linden, A., Vrancken, K., 2009. Waste battery treatment options: comparing their environmental performance. Waste Manag. 29, 2321–2331. http://dx.doi.org/10.1016/j.wasman.2009.03.019.

Brown, S., Kruger, C., Subler, S., 2008. Greenhouse gas balance for composting operations. J. Environ. Qual. 37, 1396–1410. http://dx.doi.org/10.2134/jeq2007.0453.

Brunner, P.H., Rechberger, H., 2004. Practical Handbook of Material Flow Analysis. Lewis Publishers, Boca Raton, FL, USA.

Bruun, S., Hansen, T.L., Christensen, T.H., Magid, J., Jensen, L.S., 2006. Application of processed organic municipal solid waste on agricultural land – a scenario analysis. Environ. Model. Assess. 11, 251–265. http://dx.doi.org/10.1007/s10666-005-9028-0.

Burnley, S.J., Phillips, R., Coleman, T., Rampling, T., 2011. Energy implications of the thermal recovery of biodegradable municipal waste materials in the United Kingdom. Waste Manag. 31, 1949–1959. http://dx.doi.org/10.1016/j.wasman.2011.04.015.

Butera, S., Christensen, T.H., Astrup, T.F., 2015. Life cycle assessment of construction and demolition waste management. Waste Manag. 44, 196–205. http://dx.doi.org/10.1016/j.wasman.2015.07.011.

Cencic, O., Rechberger, H., 2008. Material flow analysis with software STAN. J. Environ. Eng. Manag. 18, 3–7.

Chanton, J., Powelson, D.K., Green, R.B., 2009. Methane oxidation in landfill cover soils - is a 10% default value reasonable? J. Environ. Qual. 38, 654–663. http://dx.doi.org/10.2134/jeq2008.0221.

Christensen, T.H., Gentil, E., Boldrin, A., Larsen, A.W., Weidema, B.P., Hauschild, M., 2009. C balance, carbon dioxide emissions and global warming potentials in LCA-modelling of waste management systems. Waste Manag. Res. 27, 707–715. http://dx.doi.org/10.1177/0734242X08096304.

Clavreul, J., Baumeister, H., Christensen, T.H., Damgaard, A., 2014. An environmental assessment system for environmental technologies. Environ. Model. Softw. 60, 18–30. http://dx.doi.org/10.1016/j.envsoft.2014.06.007.

Cleary, J., 2009. Life cycle assessments of municipal solid waste management systems: a comparative analysis of selected peer-reviewed literature. Environ. Int. 35, 1256–1266. http://dx.doi.org/10.1016/j.envint.2009.07.009.

Clift, R., Doig, A., Finnveden, G., 2000. The application of Life Cycle Assessment to integrated solid waste management. Part 1-methodology. Process Saf. Environ. Prot. 78, 279–287. http://dx.doi.org/10.1205/095758200530790.

Consonni, S., Viganò, F., 2011. Material and energy recovery in integrated waste management systems: the potential for energy recovery. Waste Manag. 31, 2074–2084. http://dx.doi.org/10.1016/j.wasman.2011.05.013.

Dalemo, M., Sonesson, U., Björklund, A., Mingarini, K., Frostell, B., Jönsson, H., Nybrant, T., Sundqvist, J.O., Thyselius, L., 1997. ORWARE – a simulation model for organic waste handling systems. Part 1: model description. Resour. Conserv. Recycl. 21, 17–37. http://dx.doi.org/10.1016/S0921-3449(97)00020-7.

Davidsson, Å., Gruvberger, C., Christensen, T.H., Hansen, T.L., Jansen, J.l.C., 2007. Methane yield in source-sorted organic fraction of municipal solid

waste. Waste Manag. 27, 406–414. http://dx.doi.org/10.1016/j.wasman.2006.02.013.
Department of Energy & Climate Change (DECC), 2014a. Valuation of Energy Use and Greenhouse Gas (GHG) Emissions – Background Documentation. DECC, London, UK. Available from: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/360044/2014_Background_Documentation_to_DECC_HMT_Supplementary_Appraisal_Guidance.pdf.
Department of Energy & Climate Change (DECC), 2014b. What Is Combined Heat and Power? Available from: http://chp.decc.gov.uk/cms/what-is-chp/.
Department for Environment Food & Rural Affairs (Defra), 2013a. Incineration of Municipal Solid Waste. Defra, London, UK. Available from: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/221036/pb13889-incineration-municipal-waste.pdf.
Department for Environment Food & Rural Affairs (Defra), 2013b. Mechanical Biological Treatment of Municipal Solid Waste. Defra, London, UK. Available from: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/221039/pb13890-treatment-solid-waste.pdf.
Department for Environment Food & Rural Affairs (Defra), Ricardo-AEA, Carbon Smart, 2013. Greenhouse Gas Conversion Factor Repository. Available from: http://www.ukconversionfactorscarbonsmart.co.uk/.
Department for Environment Food & Rural Affairs (Defra), 2011. Government Review of Waste Policy in England 2011. Defra, London, UK. Available from: https://www.gov.uk/government/publications/government-review-of-waste-policy-in-england-2011.
du Plessis, C., Strauss, J.M., Sebapalo, E.M.T., Riedel, K.-H.J., 2003. Empirical model for methane oxidation using a composted pine bark biofilter. Fuel 82, 1359–1365. http://dx.doi.org/10.1016/S0016-2361(03)00040-1.
Ekvall, T., Assefa, G., Bjorklund, A., Eriksson, O., Finnveden, G., 2007. What life-cycle assessment does and does not do in assessments of waste management. Waste Manag. 27, 989–996. http://dx.doi.org/10.1016/j.wasman.2007.02.015.
Environment Agency (EA), 2010. Waste and Resources Assessment Tool for the Environment (WRATE) LCA Model, Rotherham, UK.
Eunomia Research & Consulting, Resource Futures, HCW Consultants, 2011. Kerbside Collection Options: Wales. Waste & Resources Action Programme, Cardiff, UK. Available from: http://www.eunomia.co.uk/reports-tools/kerbside-collections-options-wales/.
European Commission (EC), 1994. European parliament and council directive 94/62/EC of 20 December 1994 on packaging and packaging waste. Off. J. Eur. Communities L 365, 10–23. Available from: http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=URISERV:l21207.
European Commission (EC), 1999. Council directive 1999/31/EC of 26 April 1999 on the landfill of waste. Off. J. Eur. Communities L 182, 1–19. Available from: http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex:31999L0031.
European Commission (EC), 2006. Integrated Pollution Prevention and Control. Reference document on the best available techniques for waste incineration. EC, Luxembourg. Available from: http://eippcb.jrc.ec.europa.eu/reference/BREF/lcp_bref_0706.pdf.
European Commission (EC), 2008. Directive 2008/98/EC of the European parliament and of the Council of 19 November 2008 on waste and repealing certain directives. Off. J. Eur. Union L 312, 3–30. Available from: http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0098.
European Commission (EC), 2009. Decision No 406/2009/EC of the European Parliament and of the Council of 23 April 2009 on the effort of Member States to reduce their greenhouse gas emissions to meet the Community's greenhouse gas emission reduction commitments up to 2020. Off. J. Eur. Union L 140, 136–148. Available from: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:140:0136:0148:EN:PDF.
European Commission (EC), Joint Research Centre (JRC), Institute for Environment and Sustainability (IES), 2010. International Reference Life Cycle Data System (ILCD) Handbook – General Guide for Life Cycle Assessment – Detailed Guidance. Publications Office of the European Union, Luxembourg. http://dx.doi.org/10.2788/38479.
European Commission (EC), 2014. 2030 framework for Climate and Energy Policies. Available from: http://ec.europa.eu/clima/policies/2030/index_en.htm.
Evangelisti, S., Lettieri, P., Borello, D., Clift, R., 2014. Life cycle assessment of energy from waste via anaerobic digestion: a UK case study. Waste Manag. 34, 226–237. http://dx.doi.org/10.1016/j.wasman.2013.09.013.
Farrant, L., Olsen, S.I., Wangel, A., 2010. Environmental benefits from reusing clothes. Int. J. Life Cycle Assess. 15, 726–736. http://dx.doi.org/10.1007/s11367-010-0197-y.
Fellner, J., Aschenbrenner, P., Cencic, O., Rechberger, H., 2011. Determination of the biogenic and fossil organic matter content of refuse-derived fuels based on elementary analyses. Fuel 90, 3164–3171. http://dx.doi.org/10.1016/j.fuel.2011.06.043.
Fernández-Nava, Y., del Río, J., Rodríguez-Iglesias, J., Castrillón, L., Marañón, E., 2014. Life Cycle Assessment (LCA) of different municipal solid waste management options: a case study of Asturias (Spain). J. Clean. Prod. 81, 178–189. http://dx.doi.org/10.1016/j.jclepro.2014.06.008.
Finnveden, G., Johnansson, J., Lind, P., Moberg, Å., 2005. Life cycle assessment of energy from solid waste – part 1: general methodology and results. J. Clean. Prod. 13, 213–229. http://dx.doi.org/10.1016/j.jclepro.2004.02.023.
Fisher, J.I., 2006. Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions. London, UK. Available from: http://randd.defra.gov.uk/Document.aspx?Document=WR0609_5737_FRP.pdf.
Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. Changes in atmospheric constituents and in radiative forcing. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007. The Physical Science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK and New York, NY, USA.
Fruergaard, T., Hyks, J., Astrup, T., 2010. Life-cycle assessment of selected management options for air pollution control residues from waste incineration. Sci. Total Environ. 408, 4672–4680. http://dx.doi.org/10.1016/j.scitotenv.2010.05.029.
Gallert, C., Winter, J., 1997. Mesophilic and thermophilic anaerobic digestion of source-sorted organic wastes: effect of ammonia on glucose degradation and methane production. Appl. Microb. Biotechnol. 48, 405–410. http://dx.doi.org/10.1007/s002530051071.
Giugliano, M., Cernuschi, S., Grosso, M., Rigamonti, L., 2011. Material and energy recovery in integrated waste management systems. An evaluation based on life cycle assessment. Waste Manag. 31, 2092–2101. http://dx.doi.org/10.1016/j.wasman.2011.02.029.
Hall, D., Plimmer, B., Taylor, D., 2005. Life Cycle Assessment Landfill Emissions. Golder Associates (UK), Stanton-on-the-Wolds, UK.
Hansen, T.L., 2006a. Environmental modelling of use of treated organic waste on agricultural land: a comparison of existing models for life cycle assessment of waste systems. Waste Manag. Res. 24, 141–152. http://dx.doi.org/10.1177/0734242X06062485.
Hansen, T.L., 2006b. Life cycle modelling of environmental impacts of application of processed organic municipal solid waste on agricultural land (EASEWASTE). Waste Manag. Res. 24, 153–166. http://dx.doi.org/10.1177/0734242X06063053.
Hawkey, D.J.C., 2012. District heating in the UK: a technological innovation systems analysis. Environ. Innov. Soc. Transit. 5, 19–32. http://dx.doi.org/10.1016/j.eist.2012.10.005.
Hertwich, E.G., van der Voet, E., Suh, S., Tukker, A., Huijbreghts, M., Kazmierczyk, P., Lenzen, M., McNeely, J., Moriguchi, Y., 2010. Assessing the Environmental Impacts of Consumption and Production: Priority Products and Materials. A Report of the Working Group on the Environmental Impacts of Products and Materials to the International Panel for Sustainable Resource Management, United Nations Environment Programme, Paris, France. Available from: http://www.greeningtheblue.org/sites/default/files/Assessing%20the%20environmental%20impacts%20of%20consumption%20and%20production.pdf.
Her Majesty's Statutory Office (HMSO), 2008. Climate Change Act 2008. HMSO, London, UK. Available from: http://www.legislation.gov.uk/ukpga/2008/27/contents.
Her Majesty's Statutory Office (HMSO), 2015. Well-being of Future Generations (Wales) Act 2015. HMSO, London, UK. Available from: http://www.legislation.gov.uk/anaw/2015/2/contents/enacted.
Hogg, D., Ballinger, A., Oonk, H., 2011. Inventory Improvement Project – UK Landfill Methane Emissions Model. Final report to Defra and DECC, Bristol, UK. Available from: http://sciencesearch.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=17448.
Hoornweg, D., Bhada-Tata, P., 2012. What a Waste. A global review of solid waste management, Washington, DC, USA. Available from: http://hdl.handle.net/10986/17388.
Horne, J., Scholes, P., Areikin, E., Brown, B., 2013. A Survey of the UK Organics Recycling Industry in 2012. Banbury, UK. Available from: http://www.wrap.org.uk/sites/files/wrap/ASORI%202012.pdf.
International Solid Waste Association (ISWA), 2015. ISWA Declaration on Climate Change and the Waste and Resource Management Sector. ISWA, Vienna, Austria. Available from: https://www.iswa.org/fileadmin/galleries/Events/GA_Package_2015/CC_Declaration_2015.pdf.
Intergovernmental Panel on Climate Change (IPCC), 2006. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. IPCC, Hayama, Japan. Available from: http://www.ipcc-nggip.iges.or.jp/public/2006gl/.
Intergovernmental Panel on Climate Change (IPCC), 2014. Climate change 2014: synthesis report. In: Pachauri, R.K., Meyer, L. (Eds.), Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland. Available from: http://www.ipcc.ch/report/ar5/syr/.
International Organization for Standardization (ISO), 2006a. ISO 14044:2006 Environmental Management – Life Cycle Assessment – Requirements and Guidelines. ISO, Geneva, Switzerland. Available from: http://www.iso.org/iso/home/store/catalogue_tc/catalogue_detail.htm?csnumber=38498.
International Organization for Standardization (ISO), 2006b. ISO 14040:2006 Environmental Management – Life Cycle Assessment – Principles and Framework. ISO, Geneva, Switzerland. Available from: http://www.iso.org/iso/catalogue_detail?csnumber=37456.
James, K., Maddox, P., Fisher, K., 2011. Benefits of Reuse. Case Study: Electrical Items, Waste & Resources Action Programme, Banbury, UK. Available from: http://www.wrap.org.uk/sites/files/wrap/Electricals%20reuse_final.pdf.
Koehler, A., Peyer, F., Salzmann, C., Saner, D., 2011. Probabilistic and technology-specific modeling of emissions from municipal solid-waste incineration. Environ. Sci. Technol. 45, 3487–3495. http://dx.doi.org/10.1021/es1021763.

Larsen, A.W., Vrgoc, M., Christensen, T.H., Lieberknecht, P., 2009. Diesel consumption in waste collection and transport and its environmental significance. Waste Manag. Res. 27, 652–659. http://dx.doi.org/10.1177/0734242X08097636.
Laurent, A., Bakas, I., Clavreul, J., Bernstad, A., Niero, M., Gentil, E., Hauschild, M.Z., Christensen, T.H., 2014a. Review of LCA studies of solid waste management systems–part I: lessons learned and perspectives. Waste Manag. 34, 573–588. http://dx.doi.org/10.1016/j.wasman.2013.10.045.
Laurent, A., Clavreul, J., Bernstad, A., Bakas, I., Niero, M., Gentil, E., Christensen, T.H., Hauschild, M.Z., 2014b. Review of LCA studies of solid waste management systems – part II: methodological guidance for a better practice. Waste Manag. 34, 589–606. http://dx.doi.org/10.1016/j.wasman.2013.12.004.
Manfredi, S., Tonini, D., Christensen, T.H., Scharff, H., 2009. Landfilling of waste: accounting of greenhouse gases and global warming contributions. Waste Manag. Res. 27, 825–836. http://dx.doi.org/10.1177/0734242X09348529.
Massarutto, A., de Carli, A., Graffi, M., 2011. Material and energy recovery in integrated waste management systems: a life-cycle costing approach. Waste Manag. 31, 2102–2111. http://dx.doi.org/10.1016/j.wasman.2011.05.017.
Mastellone, M.L., Brunner, P.H., Arena, U., 2009. Scenarios of waste management for a waste emergency area. J. Industrial Ecol. 13, 735–757. http://dx.doi.org/10.1111/j.1530-9290.2009.00155.x.
McGill, M., 2009. Carbon Footprint Analysis of Textile Reuse and Recycling, Centre for Environmental Policy (MSc thesis). Imperial College London, London, UK.
Merrild, H., Larsen, A.W., Christensen, T.H., 2012. Assessing recycling versus incineration of key materials in municipal waste: the importance of efficient energy recovery and transport distances. Waste Manag. 32, 1009–1018. http://dx.doi.org/10.1016/j.wasman.2011.12.025.
Møller, J., Boldrin, A., Christensen, T.H., 2009. Anaerobic digestion and digestate use: accounting of greenhouse gases and global warming contribution. Waste Manag. Res. 27, 813–824. http://dx.doi.org/10.1177/0734242X09344876.
National Grid, 2008. Investigation into Transmission Losses on UK Electricity Transmission System. National Grid, London, UK. Available from: http://www.nationalgrid.com/NR/rdonlyres/4D65944B-DE42-4FF4-88DF-BC6A81EFA09B/26920/ElectricityTransmissionLossesReport1.pdf.
Nayak, G., Kumar, V., 2008. Jotun Paints - Product Life Cycle Assessment. dcarbon8, London, UK. Available from: http://www.thefuturebuild.com/assets/images/uploads/products/1521/dc8_jotunpaints_productcarbonfootprint_final_formasdar1.pdf?/product-life-cycle-assessment-981475/file-files/DC8_JotunPaints_ProductCarbonFootprint_FINAL_for%2520Masdar.pdf.
Nielsen, M., Nielsen, O.-K., Thomsen, M., 2010. Emissions from decentralised CHP plants 2007-energinet.dk. Environmental project no. 07/1882. Project report 5-emission factors and emission inventory for decentralised CHP production. National Environmental Research Institute, Aarhus University, Aarhus, Denmark. Available from: http://www2.dmu.dk/Pub/FR786.pdf.
Office for National Statistics (ONS), 2015. Overview of UK Population, Estimates for England, Wales, Scotland and Northern Ireland Mid-2014-released 25 June 2015. Available from: http://www.ons.gov.uk/ons/rel/pop-estimate/population-estimates-for-uk–england-and-wales–scotland-and-northern-ireland/mid-2014/sty–-overview-of-the-uk-population.html.
Patterson, T., Esteves, S., Dinsdale, R., Guwy, A., 2011. Life cycle assessment of biogas infrastructure options on a regional scale. Bioresour. Technol. 102, 7313–7323. http://dx.doi.org/10.1016/j.biortech.2011.04.063.
Pressley, P.N., Levis, J.W., Damgaard, A., Barlaz, M.A., DeCarolis, J.F., 2015. Analysis of material recovery facilities for use in life-cycle assessment. Waste Manag. 35, 307–317. http://dx.doi.org/10.1016/j.wasman.2014.09.012.
Qiang, H., Qiang, Z., Nazim, C., Danny, M., 2011. Biofilter: a promising tool for mitigating methane emission from manure storage. J. Arid Land 3, 61–70. http://dx.doi.org/10.3724/SP.J.1227.2011.00061.
Rigamonti, L., Grosso, M., Giugliano, M., 2009. Life cycle assessment for optimising the level of separated collection in integrated MSW management systems. Waste Manag. 29, 934–944. http://dx.doi.org/10.1016/j.wasman.2008.06.005.
Rigamonti, L., Grosso, M., Giugliano, M., 2010. Life cycle assessment of sub-units composing a MSW management system. J. Clean. Prod. 18, 1652–1662. http://dx.doi.org/10.1016/j.jclepro.2010.06.029.
Saft, R., 2007. Life cycle assessment of a pyrolysis/gasification plant for hazardous paint waste. Int. J. Life Cycle Assess. 12, 230–238. http://dx.doi.org/10.1065/lca2007.05.332.
Salhofer, S., Schneider, F., Obersteiner, G., 2007. The ecological relevance of transport in waste disposal systems in Western Europe. Waste Manag. 27, S47–S57. http://dx.doi.org/10.1016/j.wasman.2007.02.025.
Salisbury, E., Thistlethwaite, G., Young, K., Cardenas, L., Thomas, A., 2015. Greenhouse Gas Inventories for England, Scotland, Wales and Northern Ireland: 1990–2013, London, UK. Available from: http://naei.defra.gov.uk/reports/reports?report_id=810.
Smith, A., Brown, K., Ogilvie, S., Rushton, K., Bates, J., 2001. Waste Management Options and Climate Change. Final Report to the European Commission. DG Environment, Luxembourg. Available from: http://ec.europa.eu/environment/waste/studies/pdf/climate_change.pdf.
Spokas, K., Bogner, J., Chanton, J., Morcet, M., Aran, C., Graff, C., Golvan, Y., Hebe, I., 2006. Methane mass balance at three landfill sites: what is the efficiency of capture by gas collection systems? Waste Manag. 26, 516–525. http://dx.doi.org/10.1016/j.wasman.2005.07.021.
StatsWales, 2015a. Local Authority Municipal Waste Management, 2008-09 to 2013-14. Available from: https://statswales.wales.gov.uk/Download/File?fileId=379.
StatsWales, 2015b. Reuse/recycling/composting rates by local authority and year. Welsh Government. Available from: http://wales.gov.uk/statistics-and-research/local-authority-municipal-waste-management/?lang=en.
Streese, J., Stegmann, R., 2005. In: Potentials and Limitations of biofilters for Methane Oxidation, Proceedings Sardinia 2005, Tenth International Waste Management and Landfill Symposium, S. Margherita di Pula, Cagliari, Italy. Available from: http://www.image.unipd.it/cossu/Environmental%20Project%20Work/Prevous%20years/Articoli%20scientifici/Stegmann%20-%20Biofilters%20for%20methane%20oxidation.pdf.
Timlett, R., Williams, I.D., 2011. The ISB model (infrastructure, service, behaviour): a tool for waste practitioners. Waste Manag. 31, 1381–1392. http://dx.doi.org/10.1016/j.wasman.2010.12.010.
Turconi, R., Butera, S., Boldrin, A., Grosso, M., Rigamonti, L., Astrup, T., 2011. Life cycle assessment of waste incineration in Denmark and Italy using two LCA models. Waste Manag. Res. 29, 78–90. http://dx.doi.org/10.1177/0734242X11417489.
Turner, D.A., Kemp, S., Williams, I., 2011. Carbon footprinting in the UK waste management sector. Carbon Manag. 2, 677–690. http://dx.doi.org/10.4155/CMT.11.67.
Turner, D.A., Williams, D.I., Kemp, S., 2016. Greenhouse gas emission factors for recycling of source-segregated waste materials. Resour. Conserv. Recycl. 105, 186–197. http://dx.doi.org/10.1016/j.resconrec.2015.10.026.
United Nations Environment Programme (UNEP), 2010. Waste and Climate Change: Global Trends and Strategy Framework. UNEP, Osaka, Japan. Available from: http://www.unep.or.jp/ietc/Publications/spc/Waste&ClimateChange/Waste&ClimateChange.pdf.
US EPA, 2011. Background Information Document for Life-cycle Inventory Landfill Process Model. Durham, NC, USA. Available from: http://www4.ncsu.edu/~jwlevis/Landfill-2011.pdf.
WasteDataFlow, 2014. Welcome to the WasteDataFlow Website. Available from: http://www.wastedataflow.org/.
Welsh Assembly Government (WAG), 2009. One Wales: One Planet. The Sustainable Development Scheme of the Welsh Assembly Government. WAG, Cardiff, UK. Available from: http://www.wales.nhs.uk/sitesplus/documents/829/One%20Wales-%20One%20Planet%20(2009).pdf.
Welsh Assembly Government (WAG), 2010a. Climate Change Strategy for Wales. WAG, Cardiff, UK. Available from: http://gov.wales/topics/environmentcountryside/climatechange/publications/strategy/?lang=en.
Welsh Assembly Government (WAG), 2010b. Towards Zero Waste: the Overarching Waste Strategy Document for Wales. WAG, Cardiff, UK. Available from: http://gov.wales/topics/environmentcountryside/epq/waste_recycling/zerowaste/?lang=en.
Welsh Assembly Government (WAG), 2011. Municipal Sector Plan. Part 1. Welsh Assembly Government, WAG Cardiff, UK. Available from: http://gov.wales/topics/environmentcountryside/epq/waste_recycling/publication/municipalsectorplan/?lang=en.
Waste & Resources Action Programme (WRAP), 2010. The Composition of Municipal Solid Waste in Wales. WRAP, Cardiff, UK. Available from: http://www.wrapcymru.org.uk/content/composition-municipal-solid-waste-wales-0.
Yang, N., Damgaard, A., Lu, F., Shao, L.M., Brogaard, L.K., He, P.J., 2014. Environmental impact assessment on the construction and operation of municipal solid waste sanitary landfills in developing countries: China case study. Waste Manag. 34, 929–937. http://dx.doi.org/10.1016/j.wasman.2015.07.011.
Yazdani, R., Barlaz, M.A., Augenstein, D., Kayhanian, M., Tchobanoglous, G., 2012. Performance evaluation of an anaerobic/aerobic landfill-based digester using yard waste for energy and compost production. Waste Manag. 32, 912–919. http://dx.doi.org/10.1016/j.wasman.2012.01.003.
Yoshida, H., Gable, J.J., Park, J.K., 2012. Evaluation of organic waste diversion alternatives for greenhouse gas reduction. Resour. Conserv. Recycl. 60, 1–9. http://dx.doi.org/10.1016/j.resconrec.2011.11.011.