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Greenhouse gas emission factors for recycling of source-segregated waste materials

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ABSTRACT

A key challenge for the waste management sector is to maximise resource efficiency whilst simultaneously reducing its greenhouse gas (GHG) emissions. For stakeholders to better understand the GHG impacts of their waste management activities and identify emissions reduction opportunities, they need to be able to quantify the GHG impacts of material recycling. Whilst previous studies have been undertaken to develop GHG emission factors (EF) for materials recycling, they are generally insufficient to support decision-making due to a lack of transparency or comprehensiveness in the range of materials considered. In this study, we present for the first time a comprehensive, scientifically robust, fully transparent, and clearly documented series of GHG EFs for the recycling of a wide range of source-segregated materials. EFs were derived from a series of partial life cycle assessments (LCA) performed as far as possible in accordance with the ISO 14040 standard. With the exceptions of soil, plasterboard, and paint, the recycling of source-segregated materials resulted in net GHG savings. The majority of calculated GHG EFs were within the range of data presented in the literature. The quality of secondary data used was assessed, with the results highlighting the dearth of high quality life cycle inventory (LCI) data on material reprocessing and primary production currently available. Overall, the results highlight the important contribution that effective source-segregated materials recycling can have in reducing the GHG impacts of waste management.

1. Introduction

Solid waste management contributes less than 5% to global greenhouse gas (GHG) emissions (Bogner et al., 2007). In response to growing concerns about the threat of climate change, international action aimed at reducing greenhouse gas (GHG) emissions is accelerating and the solid waste management sector is expected to contribute. Previous laggards such as the United States of America (USA) have recently committed to a reduction of GHG emissions of 26–28% below 2005 levels by 2025, whilst China aims to peak carbon emissions by 2030 and obtain 20% of its energy from zero-carbon sources (White House, 2014). The European Union (EU) is committed to reducing its GHG emissions by at least 20% of 1990 levels by 2020 (EC, 2009) and 40% by 2030 (EC, 2014). This commitment has translated into Member States developing their own ambitious GHG emissions reduction targets. For example, the United Kingdom (UK) has committed to reducing their GHG emissions by 80% of 1990 levels by 2050 (HMSO, 2008). A key challenge for the waste management sector is to maximise resource efficiency whilst simultaneously reducing its GHG emissions (Turner et al., 2011). Numerous international studies have shown that the recycling of waste materials can result in net savings of GHG emissions (Björklund and Finnveden, 2005; Franchetti and Kilaru, 2012; Manfredi et al., 2011; WRAP, 2006, 2010a). This is because recycling materials into new ("secondary") products can displace production of "primary" products that can require significant inputs of energy and raw materials. In order for stakeholders to better understand the GHG impacts of their waste management activities and identify GHG emissions reduction opportunities to help achieve national GHG emissions reduction targets, they need to be able to quantify the GHG emissions from material recycling.

Typically, GHG emissions are estimated using emission factors (EFs) that relate the quantity of a pollutant emitted to a unit of activity (e.g., kg fossil CO_2 per tonne of material reprocessed). EFs for different GHGs are usually aggregated and expressed as CO_2 equivalent (CO_2e) per activity unit. In the case of waste material recycling, EFs are often expressed per tonne of waste material collected and sent for recycling (kg CO_2e/t). GHG EFs for waste

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material recycling are typically developed using life cycle assessment (LCA), applied either partially (focusing solely on the climate change potential impact indicator) or fully. LCA is a well established and internationally standardised methodology (ISO, 2006a,b) for quantifying emissions from specified products or systems over their entire life cycle. LCA accounts for both the environmental burdens (e.g., GHG emissions from residual waste disposed of in landfill) and benefits (e.g., the recovery of recycling of waste materials to produce secondary products that replace the production of primary products). However, choices regarding system boundaries definition, model parameterisation, and data selection can significantly affect the calculated results (Finnveden, 1999). Furthermore, GHG EFs are generally developed for specific geographical areas and technologies, and their appropriateness to other situations may be questionable. To ensure that appropriate and representative GHG EFs are applied, a thorough examination of background information relating to methodological choices taken and data sources is essential (Brogaard et al., 2014). However, GHG EFs are rarely accompanied by such detailed documentation.

A number of studies have presented GHG emissions factors for materials recycling that may be used to support decision makers. A Carbon Metric has been developed by WRAP (Waste and Resources Action Programme) for the purpose of assisting the Scottish government in evaluating the GHG impacts of its national solid waste management system and to identify areas for improvement (Pratt, 2014; Pratt et al., 2013). The Carbon Metric is used to measure progress towards national waste reduction targets and evaluate the impacts of waste policies on Scotland's GHG impacts. As part of the ongoing study, WRAP have produced a series of GHG emission factors for recycling, incineration, and landfilling of certain waste materials and waste streams based on secondary data from a range of published and unpublished sources. Whilst details of the data sources used are provided, specific documentation regarding the modelling approach and assumptions made to produce the EFs is absent. Furthermore, comprehensive EFs for materials recycling are lacking.

An adaptation of the Carbon Metric was produced by WRAP for England (WRAP, 2012). The GHG EFs produced in the study were developed to be used by local authorities in conjunction with data from WasteDataFlow[1] to evaluate waste management performance. No accompanying documentation regarding the development of the EFs is provided. Furthermore, EFs for a number of waste materials are not included due to lack of data.

GHG EFs have also been produced by Defra (Department for Environment, Food & Rural Affairs) in the UK for the purposes of organisational GHG emissions reporting (Defra et al., 2013). However, the EFs are only presented as gross results (i.e., only direct GHG impacts are counted) and 'avoided impacts' (i.e., emissions savings through the substitution of primary energy or material production due to the recovery and production of energy from waste or secondary products) are not included (Hill et al., 2013). Two other studies prepared for Defra by Environmental Resources Management (ERM) have reported GHG EFs for waste materials recycling in the UK. Fisher et al. (2006) undertook a macro-level investigation of the flows of carbon and energy to evaluate the GHG impacts associated with alternative management options for key waste materials in the UK, whilst Fisher (2006) evaluated the GHG impacts of solid waste management policies in the context of the UK waste management system. Both studies provide sufficient documentation and are fully transparent in describing their approach and modelling assumptions. However, both studies are limited in terms of the materials for which GHG EFs are presented, with only the common recyclables covered in the assessments.

The United States Environmental Protection Agency (US EPA) has developed a Waste Reduction Model (WARM) to assist solid waste managers and organisations to measure and report their GHG emissions from solid waste management (US EPA, 2015). For ease of use, the model exists as both a web-based calculator and as a Microsoft Excel spreadsheet. As part of the development of the model, GHG EFs were developed for the recycling of 39 different dry materials. Each EF is well documented, with the modelling approach and assumptions taken clearly reported. Despite the broad range of materials considered, the EFs developed for the US EPA's WARM model do not cover the full range of waste materials reported on in WasteDataFlow. Furthermore, the EFs were developed based on the situation in the USA and lack relevance to European systems and technologies.

Finally, a series of conceptual review papers were published in a special edition of Waste Management & Research that provide a transparent assessment of the GHG impacts associated with the recycling of key materials, including paper (Merrild et al., 2009), metals (Damgaard et al., 2009), plastics (Astrup et al., 2009), glass (Larsen et al., 2009), and wood (Merrild and Christensen, 2009). Each paper provides a description of the relevant material reprocessing technologies involved and an overview of the range of GHG contributions associated with each technology. The GHG contributions were quantified in the geographic context of Northern Europe. Whilst the GHG impacts of materials recycling are transparently documented, the papers are limited in scope to covering only the key recyclable materials.

Although a number of studies have been undertaken to produce GHG EFs for materials recycling (source-segregated or commingled), they are generally insufficient to support national policy makers and local decision makers due to a lack of transparency and clarity in documentation or comprehensiveness in terms of the range of waste materials considered. In this paper, we present for the first time a comprehensive series of GHG EFs for the recycling of a wide range of source-segregated materials based on the results of individual material-specific partial LCA studies. We have focused on source-segregated (aka source-separated) materials as they comprise a large proportion of collected dry recyclables from the municipal solid waste stream in the UK (WYG, 2013) and have been found to produce a higher quality recyclate with lower contamination rates compared to commingled materials collection systems (WRAP, 2008). The purpose of developing the GHG EFs is to assist UK and international decision makers at multiple scales (national/regional governments, local authorities, organisations, and entrepreneurs) in measuring environmental performance and identifying optimal solid waste management solutions with regards to GHG emissions.

2. Methodology

In this study, GHG EFs for recycling of source-segregated materials were derived from a series of partial LCAs undertaken for each recyclable material investigated. The focus of the LCA was on the potential climate change impacts of materials recycling. The LCAs were performed as far as possible in accordance with the ISO 14040 standard (ISO, 2006a,b) and following the technical guidance of the International Reference Life Cycle Data System (ILCD) (EC, 2010, 2011). The LCAs were carried out using EASETECH, a LCA model for the assessment of environmental technologies developed at the Technical University of Denmark (Clavreul et al., 2014). EASETECH was selected as the LCA model as it allows for detailed modelling of heterogonous material flows through complex systems and includes specialised functionality for solid waste management system LCA modelling.

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

2.1. Goal definition

The goal of this study was to quantitatively evaluate the GHG emissions from recycling of source-segregated waste materials. The assessment was intended to provide a comprehensive, transparent, and scientifically robust catalogue of GHG EFs to support waste managers involved with decision-making at a local authority level, as well as national policy-makers and organisations seeking to better understand the GHG impacts of their recycling activities. The secondary purpose was to provide a foundation for the development of holistic material recycling LCA studies and to identify areas where high quality data are presently lacking.

The quantitative evaluation followed an "attributional approach", with average data used and allocation issues resolved through system expansion (EC, 2010). To account for multi-functional processes that produce secondary materials and energy, system expansion was performed. This approach incorporates both the direct environmental impacts from waste management processes and the indirect "avoided" impacts associated with the production of secondary products and energy from waste that substitute for primary material and energy production (Giugliano et al., 2011).

Each recycled waste material type was evaluated individually. Each waste material recycling system comprised both (1) a foreground system, which includes waste management processes directly engaged in the management of waste materials, and (2) a background system, which comprises processes that interact with the foreground system, typically by supplying or receiving energy or material, including avoided primary materials production (through recycling) and avoided energy generation (through the generation and provision of energy derived from waste) (Clift et al., 2000).

Background data were taken from a range of sources, including the ecoinvent v2.2 and the European Life Cycle Database (ELCD) life cycle inventory (LCI) databases (Frischknecht et al., 2005; EC, 2008), the UK GHG conversion factor repository (Defra et al., 2013), and the literature. Foreground system data were taken from a range of literature sources and are described in detail in Appendix A. Details of background system data (excluding avoided primary production data—see Section 2.3.1) used are outlined in Appendix B.

It was assumed that secondary products produced from waste materials would replace the production of primary products (i.e., products produced from virgin resources). For each recycled material product, an appropriate substituted primary product was identified and production process data sourced. A substitution ratio (i.e., the amount of primary material production that is avoided as a result of the recycling of an amount of waste material) was then calculated as the product of three parameters: (1) recyclability, which refers to the amount of a waste material that ends up as a recycled product and accounts for all material losses during the recycling process; (2) material quality loss, which relates to changes in the inherent technical properties of a waste material (e.g., the shortening of fibres during paper recycling); and (3) the market substitution ratio, which reflects market elasticity and defines the actual amount of a primary product that is substituted at the market as a consequence of the production of a secondary product. Based on a common approach in the literature (e.g., Briffaerts et al., 2009; Merrild et al., 2012; Rigamonti et al., 2009a), the use of recycled materials in the production of new products was assumed not to affect the market situation. Hence, market substation ratios of 1:1 were set by default for all recycling systems evaluated.

2.2. Scope definition

The function of the systems under investigation is to recycle collected source-segregated waste materials. Hence, the functional unit used in this study was the recycling of 1 t (wet weight) of source-segregated waste material collected and sent for recycling. This study evaluated the life cycle GHG emissions from recycling of waste materials listed in the WasteDataFlow (WasteDataFlow, 2014). In total, 66 material types, categorised into 20 material groups are listed in WasteDataFlow. Of those material types, 52 were included in the evaluation (see Table 1), whilst 14 were omitted. An overview of material types omitted from the investigation is presented in Table 2 along with a justification for their omission.

Table 1
Waste material types included in this study.

Material group	Material type	Material group	Material type
Glass	Green glass	WEEE	LCAs
	Brown glass		SDAs
	Clear glass		CRTs
	Mixed glass		Fluorescent tubes & other light bulbs
Paper & card	Paper		Fridges & freezers
	Card	Batteries	Automotive batteries
	Books		Post-consumer, non-automotive batteries
	Mixed paper & card	Tyres	Car tyres
	Yellow pages		Van tyres
Metal	Steel cans		Large vehicle tyres
	Aluminium cans		Mixed tyres
	Mixed cans	Furniture	Furniture
	Other scrap metal	Rubble	Rubble
	Aluminium foil	Soil	Soil
	Aerosols	Plasterboard	Plasterboard
	Fire extinguishers	Oil	Vegetable oil
	Gas bottles		Mineral oil
	Bicycles	Composite	Composite food & beverage cartons
Plastic	Mixed plastics		Mattresses
	Mixed plastic bottles	Paint	Paint
	PET	Textiles	Textiles & footwear
	HDPE		Textiles only
	PVC		Footwear only
	LDPE		Carpets
	PP	Other	AHPs
Wood	Wood		
	Chipboard & MDF		
	Composite wood materials		

PET, polyethylene terephthalate; HDPE, high-density polyethylene; PVC, polyvinyl chloride; LDPE, low-density polyethylene; PP, polypropylene; MDF, medium-density fibreboard; LDA, large domestic appliance; SDA, small domestic appliance; CRT, cathode-ray tube; AHP, absorbent hygiene product.

2.3. System boundaries

The environmental burdens associated with the upstream life cycle stages prior to the arrival of waste materials at the primary treatment facility were not included within the life cycle system, nor were those associated with capital goods. This assumption, known as the "zero burden approach", is considered standard practice in solid waste management LCAs (Ekvall et al., 2007). The systems boundaries vary between the waste material recycling systems investigated and are defined for each waste material individually (see Appendix A). Generally, system boundaries include the sorting, disassembly/dismantlement, treatment, and reprocessing of waste materials and the disposal of rejects.

2.4. Impact coverage

The focus of this study was on potential climate change impact. Hence, climate impact was the only potential impact category

Table 2
Waste material types omitted from this study.

Material group	Material type	Justification
Organic	Green garden waste only	Not collected for dry recycling
	Waste food only	Not collected for dry recycling
	Mixed garden & food waste	Not collected for dry recycling
	Other compostable waste	Not collected for dry recycling
Wood	Wood for composting	Not collected for dry recycling
Commingled	Commingled materials	Considered beyond the scope of this study, which focused on source-segregated materials
IBA	IBA	In the context of this study, IBA was considered as residual waste and not dry recyclate. However, given that IBA is commonly recycled into secondary aggregate for use in road construction and maintenance, the GHG EF presented here for rubble (or an adaptation thereof) could be considered an appropriate proxy
Metal	Metals from IBA	Metal types recovered are highly dependent on incinerated material composition (i.e., Al, Cu, and Fe content), hence the use of a generic GHG EF for IBA metals was considered inappropriate for users
Plastic	PS	Lack of available data
	Other plastics	Not commonly recycled; lack of available data
Composite	Ink & toner cartridges	Lack of specific data or appropriate proxy material
	Video tapes, DVDs, & CDs	Lack of specific data or appropriate proxy material
Other	Other materials	Ambiguity of material type
	Bric-a-brac	Lack of specific data or appropriate proxy material

IBA, incinerator bottom ash; GHG, greenhouse gas; EF, emission factor; PS, polystyrene.

included in this study. Emissions of four GHGs, fossil CO_2, biogenic CO_2, CH_4, and N_2O, were included; combined, emissions of these three gases represent more than 90% of GHG emissions from solid waste management (Bogner et al., 2007).

2.5. Life cycle inventory

2.5.1. Recycling

Due to the large number of waste material recycling systems and processes under investigation and space limitations, detailed information about the data used and assumptions made to model the treatment and reprocessing (including system expansion and market substitution) of each waste material investigated are not presented here, but are presented in detail in Appendix A.

The quality of foreground system process data and avoided primary production data used in this study were qualitatively assessed in conjunction with a pedigree matrix. Data were assessed on a five point scale (1, best quality; 5, worst quality) against five data quality indicators (reliability, completeness, temporal correlation, geographical correlation, and further technological correlation) (see Weidema, 1998; Weidema and Wesnæs, 1996). An overall data quality rating (DQR) for each data set was then calculated based on the ICLD data quality assessment methodology (EC, 2010), as:

$$\mathrm{DQR} = \frac{R + \mathrm{Co} + \mathrm{TeC} + \mathrm{GC} + \mathrm{FTC} + W_i \cdot 4}{i + 4} \qquad (1)$$

where, R, Co, TeC, GC, and FTC are the data set data quality indicator scores for reliability, completeness, geographical correlation, temporal correlation, and further technological correlation, respectively, W_i is the weakest data quality indicator score obtained among i number of data quality indicators. Qualitative descriptions of overall data set quality were then outlined according to the calculated DQR (DQR $\leq$ 1.6, “high quality”; DQR > 1.6 to $\leq$3, “fair quality”; and DQR > 3, “poor quality”). A comprehensive overview of all data sources used to model the foreground system processes and avoided primary production processes and their calculated data quality ratings is presented in Appendix C.

2.5.2. Transport

The transport of waste materials from the primary recycling facility to secondary recycling facilities was included in this study. Distances of 250 km for inter-facility transport of waste materials for recycling and 25 km for the transport of process rejects to a landfill site were assumed and were applied consistently. These distances represent the distance between the start location (primary facility) and the end location (receiving/secondary facility). All transport was assumed to be completed by freight lorry. A lorry diesel consumption of 0.21 kg per vehicle km (Spielmann et al., 2007) and a vehicle payload of 17.6 t (EA, 2010) was assumed, with the EF for diesel fuel consumption taken from Defra, Ricardo-AEA (2013). Fuel consumption from vehicle travel post-unloading was not included.

Table 3
Inventory data for disposal of 1 t of wet waste in a non-hazardous landfill.

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.

2.5.3. Disposal of rejects

Waste material rejected from unit processes was assumed to be disposed of in a landfill site. The process for disposal of rejects to landfill was modelled based on an average UK medium sized (20 m depth; 25 ha area) conventional non-hazardous landfill with landfill gas (LFG) utilisation. Emissions were modelled for a 100 year time horizon. Decay rates for waste material fractions were taken from IPCC (2006). Ancillary material and energy inputs are detailed in Table 3, whilst technical measures of the modelled LFG management system and its performance over time are detailed in Table 4. LFG recovery efficiencies reported in Table 4 were temporally averaged to reflect an average tonne of waste deposited, rather than the first mass of waste deposited (See Table 5). Electricity produced through the utilisation of recovered landfill gas in an internal combustion engine (ICE) was assumed to be exported to the National Grid. The ICE electricity generation efficiency was set to 32% (Patterson et al., 2011), with a Grid transmission efficiency of 98% assumed (National Grid, 2008). Fugitive emissions of methane during ICE and enclosed flaring operations were set to 1% (US EPA, 2011). Due to the high degree of uncertainty, biogenic

Table 4
Technical measures and performance associated with landfill gas recovery, utilisation, and oxidation of the modelled non-hazardous landfill process.

	Period 1	Period 2	Period 3	Period 4	Period 5
Duration (years)	1	4	15	30	50
Methane oxidation (%)	10[a]	10[a]	20[a]	36[b]	36[b]
Gas collected (% generated)	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).

Table 5
Temporally averaged collection efficiencies for landfill gas produced from an average tonne of landfilled waste.

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).

carbon storage (i.e., non-degraded biogenic carbon remaining within the landfill after the 100 year time horizon) was not included in the evaluation. This assumption was included as part of a sensitivity analysis.

2.5.4. Life cycle impact assessment

Results were characterised according to their global warming potential (GWP) using a 100 year time horizon (GWP100; expressed as kg CO_2-equivalents per functional unit). The GWP characterisation factors published by the Intergovernmental Panel on Climate Change (IPCC) in its Fourth Assessment Report were used (Forster et al., 2007). Whist emissions of biogenic CO_2 were included in the study (GWP100 = 0), the emissions "savings" resulting from the permanent storage of biogenic carbon in landfills or soils (i.e., the mass of non-degraded biogenic carbon after the 100 year timeframe considered) were excluded from the study but were quantified and compared with the results as part of the sensitivity analysis (see Section 3.3.2).

3. Results and discussion

3.1. GHG emission factors for recycling

The GHG EFs for source-segregated materials recycling developed in this study are presented in Table 6. The results are broken down into a gross and a net value, where the gross value is the total GHG emissions before accounting for avoided primary material or energy production and the net value is the total GHG emissions including avoided primary production. Note that a negative value represents a GHG emissions saving. Of the recycled materials investigated, 50 were found to result in net GHG savings. The extent of these savings varied considerably between different materials, with the highest savings found for recycling of aluminium and aluminium foil (both −8143 kg CO_2e/t), mixed cans, other scrap metal, aerosols, and bicycles (all −3577 kg CO_2e/t), and textile materials (all −3376 kg CO_2e/t). The lowest GHG emission savings were found for recycling of rubble (−2 kg CO_2e/t) and carpets (−9 kg CO_2e/t). Recycling of three materials, plasterboard (3.6 kg CO_2e/t), soil (27 kg CO_2e/t), and paint (86 kg CO_2e/t), was found to result in net GHG emissions. In the case of plasterboard and paint, this is likely the result of the large amount of residual waste produced during processing and subsequently disposed of to landfill (plasterboard) or treated by energy-intensive processes (paint), which produce negative GHG impacts. The positive result found for soil recycling was likely due to the low nutrient quality of the waste material, which would limit its mineral fertiliser substitution potential when applied to agricultural land, and the exclusion of carbon sequestration/binding from the calculations; a decision considered as part of the sensitivity analysis (see Section 3.3.2).

The lowest gross GHG emissions were associated with recycling of rubble (16 kg CO_2e/t), low density polyethylene (LDPE) (29 kg CO_2e/t), and soil (41 kg CO_2e/t). The results for rubble and soil reflect the low energy and material input requirements for secondary aggregate production (Huang, 2007) and agricultural land spreading (Boldrin et al., 2009). Recycling of paper (1576 kg CO_2e/t), post-consumer, non-automotive batteries (1129 kg CO_2e/t), and aluminium cans and foil (1113 kg CO_2e/t) were found to result in the highest gross GHG emissions. Despite the high gross GHG emissions for recycling of these materials, their recycling was found to result in considerable net GHG savings, particularly in the case of aluminium cans and foil where the significant advantage of recycling over primary production is highlighted.

3.2. Contribution analysis

The contributions of different processing stages to the total net GHG EFs developed for the recycled materials evaluated are presented in Fig. 1. GHG emissions from avoided primary production were found to be the major contributor for the majority of recycled material, accounting for between 25% and 97% (average = 70%) of net GHG emissions from material recycling. This highlights the importance of appropriate avoided primary production data selection in LCA studies of materials recycling (see e.g., Brogaard et al., 2014; EC, 2011; Merrild et al., 2008; Söderman, 2003). The contributions to total net GHG emissions from avoided primary production relative to reprocessing were found to be particularly important in recycling of LDPE, PET, textiles, and aluminium cans and foil, where significant differences between gross and net GHG emissions were identified (see Table 6).

Emissions from transport were found to be of limited significance to the calculated net GHG EFs, contributing, on average, just 1% of net emissions. This correlates with the findings of Merrild et al. (2012) and Salhofer et al. (2007). Similarly, disposal of process rejects was found to contribute, on average, just 2% to total net GHG emissions. Its contribution was, however, significant for recycling of many composite materials, including carpets (28%), AHPs (23%), and plasterboard (23%). This is likely a result of the high biodegradable contents of these materials and the high material loss rates associated with composite material component separation (Ward, 2004; WRAP, n.d.-a,b).

3.3. Comparison with literature GHG EFs for recycling

Also presented in Table 6 is an overview of relevant material recycling GHG EFs from the literature, including the calculated range, average, and standard deviation of values. Note that literature data included in the overview are from studies reporting

Table 6
Comparison of calculated emission factors (gross and net) with emission factors (net) reported in the literature. For literature data, average and standard deviation (st. dev.) is provided where more than two reference studies were identified. Note that whilst only a summary of literature data is presented here, detailed emission factor data for each waste material type by literature source are presented in Appendix D..

Waste material type	Calculated emission factor		Literature emission factors		
	Gross kg CO_2e/t	Net kg CO_2e/t	No. of reference studies	Range kg CO_2e/t	Average ± st. dev. kg CO_2e/t
Green glass	395	−314	6	−762 to −201	−417 ± 176
Brown glass	395	−314	6	−762 to −201	−417 ± 176
Clear glass	395	−314	6	−762 to −201	−417 ± 176
Mixed glass	395	−314	6	−762 to −201	−417 ± 176
Paper	1576	−459	7	−3891 to 390	−1195 ± 1303
Card	559	−120	5	−3439 to −280	−1010 ± 1095
Books	562	−117	3	−3428 to −811	−1709 ± 1489
Mixed paper & card	559	−120	4	−888 to −280	−601 ± 242
Yellow pages	562	−117	2	−2910 to −888	−1899 ± 1430
Steel cans	529	−862	7	−2360 to −496	−1337 ± 674
Aluminium cans	1113	−8143	7	−19340 to −5040	−11334 ± 3512
Mixed cans	883	−3577	3	−4828 to−2573	−3789 ± 1138
Other scrap metal	883	−3577	3	−4828 to −2573	−3789 ± 1138
Aluminium foil	1113	−8143	1	–	−9267
Aerosols	883	−3577	–	–	–
Fire extinguishers	651	−673	–	–	–
Gas bottles	651	−673	–	–	–
Bicycles	883	−3577	–	–	–
Mixed plastics	339	−1024	6	−2324 to 1470	−788 ± 1007
Mixed plastic bottles	336	−1084	5	−2324 to 1470	−922 ± 1321
PET	155	−2192	6	−2324 to −566	−1570 ± 600
HDPE	379	−1149	5	−2324 to −253	−1055 ± 792
PVC	379	−1549	3	−2324 to −566	−1259 ± 936
LDPE	29	−972	4	−1586 to −850	−744 ± 981
PP	379	−1184	3	−2324 to −566	−1279 ± 925
Wood	502	−444	5	−2712 to 1	−619 ± 882
Chipboard & MDF	502	−444	5	−2723 to 1	−620 ± 886
Composite wood materials	502	−444	3	−1266 to 1	−357 ± 431
LDAs	428	−866	2	−1266 to −181	−626 ± 431
SDAs	463	−1349	1	–	−1482
CRTs	272	−228	1	–	−2767
Fluorescent tubes & other light bulbs	518	−779	–	–	–
Fridges & freezers	469	−853	3	−1042 to −181	−626 ± 431
Automotive batteries	938	−435	2	−563 to −487	−525 ± 54
Post-consumer, non-automotive batteries	1129	−205	2	−563 to −487	−525 ± 54
Car tyres	206	−636	2	-1910 to -430	−1170 ± 1047
Van tyres	197	−671	2	−1910 to −430	−1170 ± 1047
Large vehicle tyres	197	−671	2	−1910 to −430	−1170 ± 1047
Mixed tyres	206	−636	2	−1910 to −430	−1170 ± 1047
Furniture	502	−444	1	–	−921
Rubble	16	−2	4	−9 to 2	−2 ± 5
Soil	41	27	2	−2 to 2	0 ± 2
Plasterboard	59	4	2	−139 to 33	−53 ± 122
Vegetable oil	647	−2759	1	–	−725
Mineral oil	647	−2759	2	−725 to −725	−725 ± 0
Composite food & beverage cartons	629	−452	1	–	−1730
Mattresses	478	−1241	–	–	–
Paint	364	86	1	–	−2840
Textiles & footwear	401	−3376	5	−7869 to−930	−3606 ± 2709
Textiles only	401	−3376	5	−7869 to −930	−3606 ± 2709
Footwear only	401	−3376	2	−5891 to−4385	−5138 ± 1065
Carpets	181	−10	1	–	−2601
AHPs	53	0	–	–	–

PET, polyethylene terephthalate; HDPE, high-density polyethylene; PVC, polyvinyl chloride; LDPE, low-density polyethylene; PP, polypropylene; MDF, medium-density fibreboard; LDA, large domestic appliance; SDA, small domestic appliance; CRT, cathode-ray tube; AHP, absorbent hygiene product.

GHG EFs—see Appendix D for further details. Data from LCA studies and LCI databases were not included (see Brogaard et al., 2014 for a review of these data sets). GHG EFs were identified from the literature for 46 of the 53 recycled materials investigated. Of those, a sufficient number of studies (≥2 per recycled material) were available to form EF ranges for 39 materials. The majority of calculated material recycling GHG EFs fell within the range of literature values, whilst the calculated factor for automotive batteries was marginally higher than the maximum literature value. The calculated EFs for card, books, mixed paper & card, and yellow pages (−120, −117, −120, and −117 kg CO_2e/t, respectively) were notably higher (i.e., lower net GHG savings) than the maximum literature values (−280, −811, −280, and −888 kg CO_2e/t, respectively). Factors for these materials were calculated using the same recycling system, which was based on secondary corrugated board base paper production and substitution of primary corrugated board base papers (Hischier, 2007). Based on a review of LCI data sets, Brogaard et al. (2014) shows that GHG emissions from primary corrugated cardboard production are, on average, notably lower than those for primary non-corrugated cardboard production. Hence, our assumption that only corrugated cardboard base papers would be substituted rather than non-corrugated may

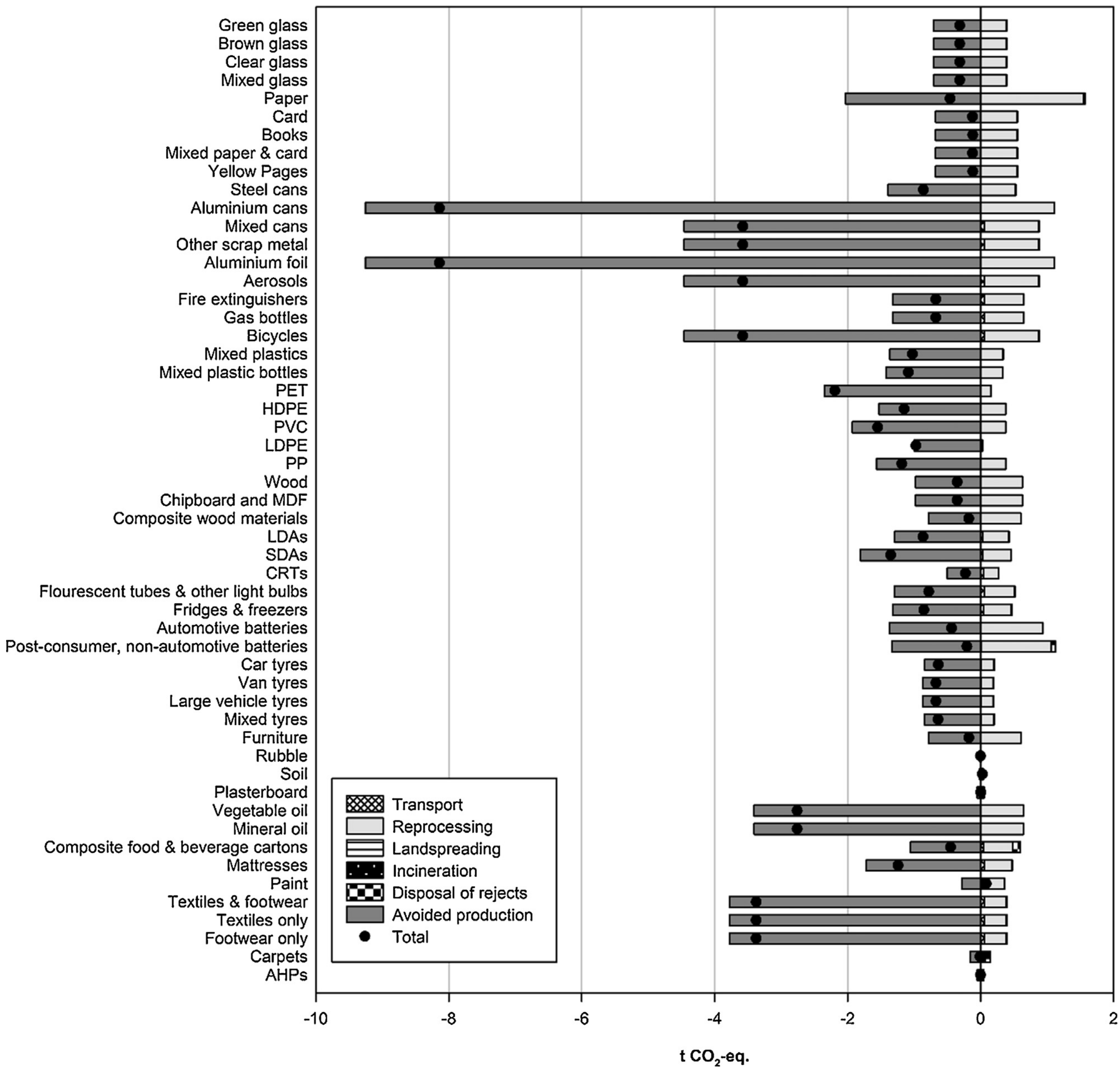

Fig. 1. Contribution analysis of the net greenhouse gas (GHG) emissions from recycling of source-segregated materials. PET, polyethylene terephthalate; HDPE, high-density polyethylene; PVC, polyvinyl chloride; LDPE, low-density polyethylene; PP, polypropylene; MDF, medium-density fibreboard; LDA, large domestic appliance; SDA, small domestic appliance; CRT, cathode-ray tube; AHP, absorbent hygiene product.

explain the higher calculated net GHG EFs compared to literature values.

In general, the material recycling GHG EFs calculated in this study were lower (i.e., they produced lower net GHG savings) than the average literature values. A notable exception concerns the EFs for plastics recycling, where the calculated factors for seven of the eight plastic types investigated were markedly higher than the average of the literature data. This may be a consequence of the default 1:1 market substation ratio applied in this study, which does not consider material-specific market situations. In the case of plastics, there are generally a number of differences between the primary and secondary polymers in terms of their market value, mechanical properties, colour, and other side characteristics, such as smell. To reflect these differences, Rigamonti et al. (2009b) recommends that a market substitution ratio of 1:0.81 be applied in the case of plastics recycling. The influence of the market substitution ratio parameter on the calculated EFs was further considered as part of a sensitivity analysis (see Section 3.3.2).

3.4. Sensitivity analysis

3.4.1. Perturbation analysis

We applied a market substitution ratio of 1:1 in all primary material production substitution calculations, based on the assumption that the market situation would be unaffected by the supply of secondary products. To test the sensitivity of the results to this assumption, market substitution ratio parameter values were varied by −10% and sensitivity ratios (SR) were calculated for each material recycling system. A SR is the ratio between the percentage change in the model result and the percentage change in the parameter value (Clavreul et al., 2012). Where more than two primary products are substituted in a material recycling system, market substitution rate SRs were calculated for each market substitution

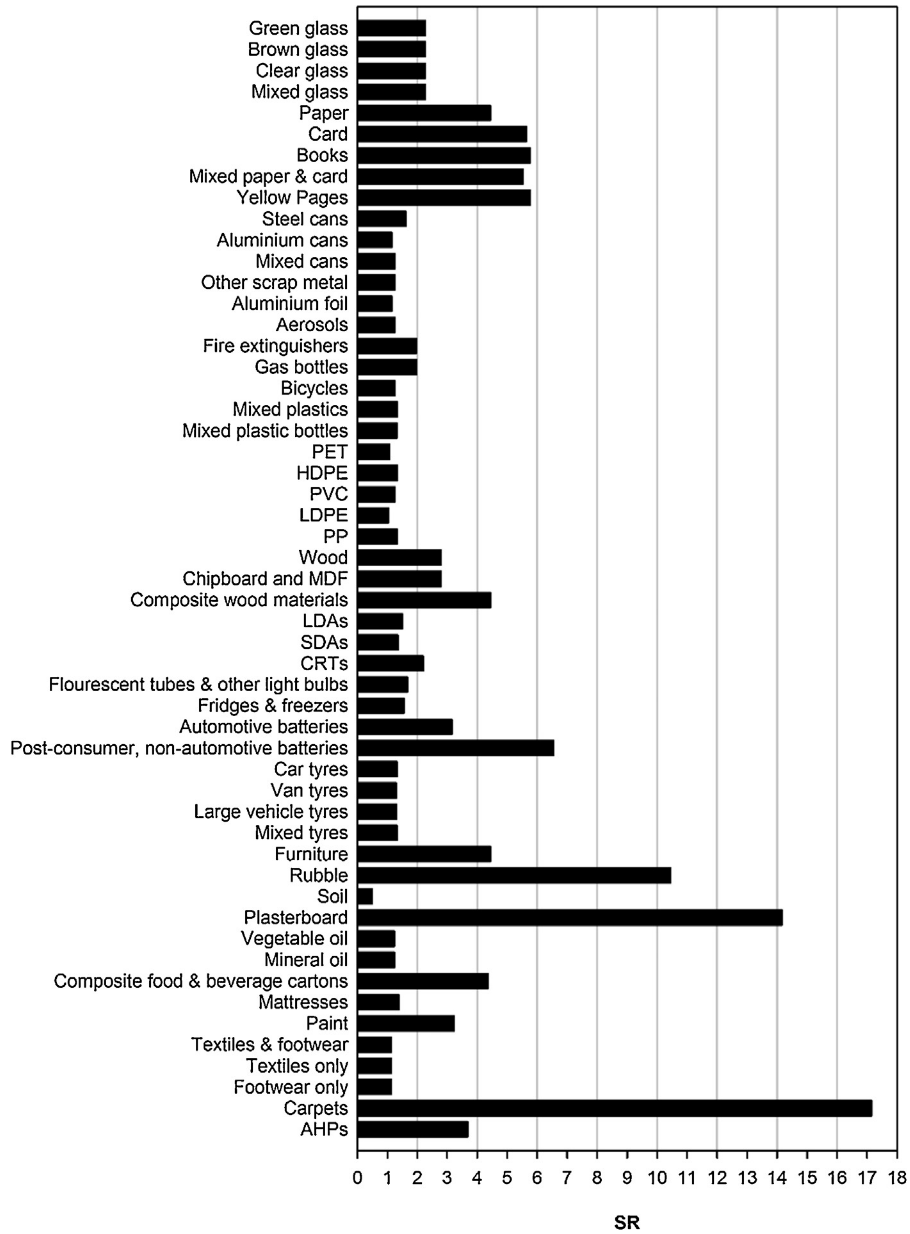

Fig. 2. Market substitution ratio parameter sensitivity ratios (SR) of each source segregated waste material type recycling system. Note that SRs are presented for each waste material type in more detail in Appendix E. AHP, absorbent hygiene product; WEEE, waste electrical and electronic equipment; CRT, cathode-ray tube; MDF, medium-density fibreboard; PP, polypropylene; LDPE, low-density polyethylene; PVC, polyvinyl chloride; HDPE, high-density polyethylene; PET, polyethylene terephthalate.

individually and then summed to produce an overall market substitution ratio SR for that material recycling system. Fig. 2 presents the SR results (as absolute value) for the material recycling systems evaluated (SR results are presented in greater detail in Appendix E). This analysis highlights the large influence of the market substitution ratio parameter value on material recycling net GHG emissions. Market substitution ratio SR values greater than 1 were calculated for all but one of the material recycling systems investigated, meaning that a variation in parameter value causes a larger relative variation in the result (Clavreul et al., 2012). Soil was the only material found to have a market substitution ratio parameter SR value less than 1 (SR = 0.5). This is likely due to the low contribution of avoided primary production to total net GHG emissions (25%) in the soil recycling system (see Fig. 1). Recycled material systems that showed particular sensitivity to the market substitution ratio parameter include carpets (SR = 17.2), plasterboard (SR = 14.2), and rubble (SR = 10.5).

3.4.2. Scenario analysis

The GHG emissions impact of excluding carbon sequestration from the material recycling systems evaluated was investigated through scenario analysis. Carbon contained in organic materials that do not degrade during the 100 year time period considered is assumed stored within the landfill indefinitely, effectively removing it from the global carbon cycle. As this process would not occur naturally, landfills constitute as an anthropogenic carbon "sink". Based on a simple carbon mass balance model, Christensen et al. (2009) asserts that where the IPCC GWP characterisation factors are used, which count biogenic carbon emissions to air (as CO_2) as neutral, biogenic carbon sequestered in landfill should be ascribed a GWP of −1. As GHG emissions from carbon sequestration were not included in the calculated EFs for recycling, a scenario (S2) that included carbon sequestration was modelled to test the sensitivity of baseline results (S1) to this choice. Results where variation between S1 and S2 was greater than 1% are presented in Table 7.

Table 7
Comparison of net greenhouse gas emission factors for selected recycled waste materials where carbon sequestration is excluded (S1) or included (S2). Variation is presented in absolute terms. Only results where variation (%) is greater than 1% are presented.

Waste material type	S1	S2	Variation	
	kg CO_2e/t	kg CO_2e/t	kg CO_2e/t	%
Paper	−459	−491	32	7
Card	−120	−128	8	6
Books; yellow pages	−117	−124	7	6
Mixed paper & card	−123	−132	9	7
Composite wood materials; furniture	−177	−244	68	38
Car tyres	−636	−760	124	19
Van tyres; large vehicle tyres	−671	−904	232	35
Mixed tyres	−640	−777	137	21
Rubble	−2	−43	41	2595
Soil	27	−55	82	301
Plasterboard	4	−12	15	435
Composite food & beverage cartons	−242	−260	18	8
Mattresses	−1241	−1353	112	9
Carpets	−9	−139	130	1469
AHPs	14	−27	41	300

AHP, absorbent hygiene product.

The choice of including carbon sequestration had little influence on the calculated net GHG EFs for the majority of materials (36 of 53, variation < 1%). However, the choice had a significant impact on the calculated EFs for a number of materials, particularly rubble (variation = 2595%, 41 kg CO_2e/t), carpets (variation = 1469%, 130 kg CO_2e/t), plasterboard (variation = 435%, 15 kg CO_2e/t), soil (variation = 301%, 82 kg CO_2e/t), and AHPs (variation = 300%, 41 kg CO_2e/t). The benefits of including carbon sequestration were most significant for soil – where a portion of biogenic carbon (here assumed to be 14% based on Bruun et al., 2006) remains bound to the *in situ* soil after the 100 year time period (Boldrin et al., 2009) – and for composite and aggregate materials that contain a high proportion of biodegradable material. As noted previously, composite materials recycling systems are characterised by high material losses and, consequently, a large amount of this biodegradable material is landfilled. Of the composite materials identified above, most contain a high proportion of paper/fibre or, in the case of carpets, natural textiles, which have high lignin contents and do not degrade readily in a landfill (Barlaz, 1998). Hence, their disposal in landfill results in net GHG savings when carbon sequestration is included. Overall, the choice of including carbon sequestration resulted in net GHG savings being calculated for all recycled materials investigated.

3.5. Limitations

The calculated EFs presented in this paper were based on numerous subjective modelling choices and methodological assumptions. Furthermore, this study was fundamentally limited by the availability of data (predominantly from secondary sources) and the quality of these data. These assumptions and limitations are each sources of uncertainty and may have influenced the results (Cellura et al., 2011). In the following sections, the influence of key methodological aspects on the results of the study are identified and discussed.

A significant source of uncertainty in this study relates to the availability and use of secondary data, upon which the study was entirely reliant. A number of materials were not included in the evaluation due to a lack of specific data (e.g., “ink & toner cartridges” and “video tapes, DVDs, and CDs”), whist the EFs for a number of materials were estimated by using the modelled recycling systems of related materials as proxies due to a lack of more specific data. Such EFs should be treated with a high degree of uncertainty.

Where secondary data sources were identified, choices related to secondary data selection were ultimately made subjectively. Furthermore, the quality of selected secondary data may also affect the LCA results. Foreground system processes and avoided primary production processes were assessed in relation to established data quality requirements. None of the foreground system process data sets used in this study were rated “high quality”, whilst the majority (51%; 48 of 94) were rated “poor quality” (the remainder were rated “fair quality”; 46 of 94). Similarly, only 7% (2 of 29) of included avoided primary production process data sets were “high quality”, 59% (17 of 29) were “fair quality”, and the remainder were identified as “poor quality” (34%; 10 of 29). This highlights the dearth of high quality LCI process data sets for materials recycling currently available and represents a significant source of uncertainty.

For a number of the materials, the complex and varied recycling routes encountered in the real world are not adequately reflected in the calculated EFs. For example, glass was assumed to be recycled in a 100% closed loop system where it is remelted and used in secondary container glass production. In reality, waste glass recycling applications are multifarious and include (non-exclusively) the production of secondary aggregates, bricks, insulation glass fibre, and filtration media (see Enviros Consulting Ltd, 2003). Similarly, waste paper was assumed to be used exclusively in the production of secondary newsprint. Real-world utilisation pathways of recovered paper are numerous and depend on the paper grade and market situation (Frees et al., 2005, WRAP, 2010b). Large variations in energy use and, consequently, life-cycle GHG emissions exist between different paper grade production processes (Laurijssen et al., 2010). Furthermore, Wang et al. (2012) showed that, in the context of paper recycling, the choice of reprocessing technology and substituted virgin production process significantly affect the LCA results. Hence, the simplified recycling systems used in the calculation of EFs may limit their representativeness of the real-world situation.

A further limitation concerns the validity of results in the context of the global, interconnected waste market. The quantity of recyclate collected in the UK has increased markedly over the past 10 years and, whilst the domestic reprocessing sector continues to expand, there is insufficient capacity to absorb these materials. Furthermore, whilst domestic markets for a number of recyclable materials remain under-developed, many recovered materials are in high demand globally and represent valuable commodities when traded on foreign markets (APSRG, 2013). Consequently, export rates of recyclable materials have increased, with the UK currently exporting around 15 million tonnes of recyclate per year (Defra, 2013). The relationship between globalisation and increased complexity and geographical dispersal of secondary product manufacturing and application makes it difficult for the LCA practitioner to identify appropriate reprocessing and substituted production data sets and material recycling model parameter values – i.e., material loss, material quality loss, and market substitution ratio – without an intimate knowledge of recyclate export destinations and foreign waste markets. Due to the inherent complexity of globalised waste trade and a lack of knowledge about foreign reprocessing and secondary product markets, the EFs calculated in this study were based on the assumption that all waste materials and components are reprocessed in the UK and secondary products substitute for domestic primary production.

The choice to exclude exportation of recyclable materials from the EF calculations may affect the results of the study as the geographical situation in which materials are recycled can be an important factor in LCA studies due largely to variations in regional energy mix. The energy used in secondary and, more importantly, primary materials production, particularly electricity, is commonly a key contributor to net GHG emissions from materials recycling (Brogaard et al., 2014; Friedrich and Trois, 2013). The GHG emissions intensity of energy systems varies considerably between countries due to differences in grid energy source mixes. A substantial portion of recyclable materials recovered in the UK are exported to destinations in Asia and Eastern Europe that are connected to highly carbon intensive energy systems. Consequently, the GHG savings due to the indirect effects of materials recycling on avoided primary production may be higher where materials are recycled in countries with GHG intensive energy dependencies (Sevigné-Itoiz et al., 2014). For example, Friedrich and Trois (2013) found that recycling of glass, metals, plastics, and paper in South Africa resulted in much higher GHG savings when compared to other countries due to the high GHG emissions intensity of the South African grid electricity system. McMillan and Keoleian (2009) showed that GHG emissions from aluminium production were significantly higher in Asian and Oceanian countries compared to European countries.

In this section, we have described the major limitations that contribute towards uncertainty in the calculated results. Whilst the number of limitations identified are numerous, it should be noted that these limitations are not unique to this study, but are ubiquitous in solid waste management LCA studies involving material recycling systems (Lazarevic et al., 2010; Reap et al., 2008a,b). We have attempted to minimise, quantify, or explain these limitations as far as possible. However, additional work is required to better address these limitations and, thereby, enhance the robustness of the derived GHG EFs and enable more informed and less uncertain decision-making.

Where sufficient literature data were identified to enable comparison, it was found that the majority of the GHG EFs calculated in this study are within the range of data presented in the literature. Notable exceptions include card, books, mixed paper & card, and yellow pages, for which calculated EFs were notably higher (i.e., lower net GHG savings) than the maximum values reported in the literature. This was likely due to our conservative assumption that primary corrugated cardboard base papers would be substituted rather than non-corrugated cardboard base papers, the production of which has been shown to generally result in lower GHG emissions (Brogaard et al., 2014).

In general, GHG emissions contributions from transport and disposal of rejects were found to be negligible, with the calculated EFs dominated by negative (i.e., GHG burdens) and positive (i.e., GHG benefits) GHG contributions from material reprocessing and avoided primary production, respectively. The results of the sensitivity analysis showed that the calculated EFs were found to be highly sensitive to the assumed market substitution ratio, with the results for recycling of plasterboard, carpets, and rubble found to be particularly sensitive to the parameter value. Given the dominance in terms of GHG contribution of avoided primary production and the high sensitivity of results to the market substitution ratio, it is essential that practitioners undertaking LCA studies that include materials recycling ensure that their models are parameterised appropriately and representatively. The sensitivity analysis also showed that the choice of including carbon sequestration in the calculations was largely inconsequential.

All foreground system processes and avoided primary production processes used were assessed in relation to their data quality. Of the data sources used to model materials reprocessing, none were assessed as being of high quality, whilst 51% were considered poor quality. Generally, the quality of data used to model avoided primary production was found to be marginally better, with 7% of data sources assessed as being of high quality and 59% considered fair quality. Overall, these findings highlight the dearth of high quality LCI data on materials reprocessing and primary production available to LCA practitioners in the UK. To reduce uncertainty and enable waste managers to make better informed, environmentally sound decisions, there is an urgent need for more representative, more appropriate, and higher quality LCI data related to materials recycling.

The EFs presented in this paper were developed to support decision makers at multiple scales (national/regional governments, local authorities, companies, and entrepreneurs). These EFs are intended to be used to enable the appraisal of environmental performance with regards to GHG impacts, and to evaluate potential waste policies and decisions in order to identify optimal solid waste management solutions. The secondary purpose was to provide a foundation for the development of material recycling EFs for different geographic areas and the undertaking of holistic LCA studies incorporating solid waste materials recycling.

The focus of this paper was to evaluate the environmental performance of source-segregated material recycling systems. In order to contribute towards the growing debate amongst policy-makers and waste managers in the UK and abroad concerning the relative merits of different dry recycling collection systems (particularly with regards to household kerbside collection) (see e.g., Cimpan et al., 2015; Eunomia Research & Consulting et al., 2011; Fitzgerald et al., 2012; Punkkinen et al., 2012), there is a need for further research that compares the environmental performance of source-segregated waste collection systems with that of alternative systems, namely single- or dual-stream commingled collection. Such research would also serve as a means of developing additional GHG EFs for recycling of materials collected from single- or duel-stream commingled collection systems, which could be of considerable benefit to decision makers.

4. Conclusions

In this study, we performed a series of partial LCAs, following an attributional approach, to quantitatively evaluate the GHG emissions from recycling of source-segregated waste materials. The results of these assessments were used to derive the first set of comprehensive GHG EFs that can be used to contextualise the contribution of source-segregated recycling to carbon management at a national and/or local level. The results showed that, with the exceptions of soil, plasterboard, and paint, the recycling of source-segregated waste materials resulted in net GHG emissions savings. Calculated GHG EFs ranged from 86 kg CO_2e (paint) to −8143 kg CO_2e (aluminium cans and aluminium foil) per tonne of material collected for recycling. The avoided GHG emissions from the recovery of high frequency materials such as LDPE, PET, textiles, steel cans, and aluminium cans were found to be notable, highlighting the importance of effective source-segregated recycling of key waste materials in reducing the GHG impacts of waste management.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at http://dx.doi.org/10.1016/j.resconrec.2015.10.026.

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