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Factors influencing the life cycle burdens of the recovery of energy from residual municipal waste

Stephen Burnley [a,*], Terry Coleman [b], Adam Peirce [c]

[a] Department of Engineering and Innovation, The Open University, Walton Hall, Milton Keynes MK7 6AA, United Kingdom
[b] Resource and Waste Solutions LLP, Cirencester, Gloucestershire, United Kingdom
[c] Environmental Resources Management, 2nd Floor, Exchequer Court, 33 St Mary Axe, London EC3A 8AA, United Kingdom

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ABSTRACT

A life cycle assessment was carried out to assess a selection of the factors influencing the environmental impacts and benefits of incinerating the fraction of municipal waste remaining after source-separation for reuse, recycling, composting or anaerobic digestion. The factors investigated were the extent of any metal and aggregate recovery from the bottom ash, the thermal efficiency of the process, and the conventional fuel for electricity generation displaced by the power generated. The results demonstrate that incineration has significant advantages over landfill with lower impacts from climate change, resource depletion, acidification, eutrophication human toxicity and aquatic ecotoxicity. To maximise the benefits of energy recovery, metals, particularly aluminium, should be reclaimed from the residual bottom ash and the energy recovery stage of the process should be as efficient as possible. The overall environmental benefits/burdens of energy from waste also strongly depend on the source of the power displaced by the energy from waste, with coal giving the greatest benefits and combined cycle turbines fuelled by natural gas the lowest of those considered. Regardless of the conventional power displaced incineration presents a lower environmental burden than landfill.

1. Introduction

In European and other developed nations, waste management is being transformed from a disposal to a resource recovery activity. This reflects rising concerns on sustainability, restrictions on landfill availability for certain wastes and international and national policies. For example, under the terms of the Waste Framework Directive (2008/98/EC), EU member states are required to achieve a municipal waste recycling rate (including composting) of 50% by 2020 (European Commission, 2008). Recycling rates vary widely across Europe; in 2012 they ranged from 62% in Austria to less than 1% in Romania with four member states (Austria, Belgium, Germany and the Netherlands) having already achieved the 50% target (Eurostat, 2014). However, reaching these recycling rates still leaves a substantial amount of residual waste. For example, if England were to achieve the recycling target of 50% this would leave 11.5 million tonnes of municipal waste remaining for management by other means.

The Waste Framework Directive also calls on member states to adopt the waste hierarchy while noting that, for some specific waste streams, a departure from the hierarchy should be made if the application of life-cycle thinking demonstrates that this departure represents the best overall environmental outcome. Applying the waste hierarchy means that the recovery of energy from waste should normally only be considered for wastes remaining such as residual municipal waste and other mixed low-grade materials. Conventional mass-burn energy from waste (EfW) is the most commonly applied energy recovery technique and seven EU member states currently burn more than a third of their municipal waste (Eurostat, 2014) and have demonstrated that recycling rates of above 50% can be combined with significant use of EfW to minimise the amount going to landfill.

Life cycle assessment (LCA) is an environmental management tool for the evaluation of the overall environmental burdens (impacts and benefits) of providing and using goods and services. The international standards ISO 14040 and 14044 (BSI, 2006a,b) specify the procedure for carrying out and reporting LCA studies. LCAs are now widely used in assessing the environmental impacts and benefits of different waste management options and several software packages have been developed specifically for waste-related LCAs. These tools have been reviewed in detail by Gentil et al. (2010).

* Corresponding author. Tel.: +44 1908 659015; fax: +44 1908 332500.
E-mail address: stephen.burnley@open.ac.uk (S. Burnley).

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This paper reports on an LCA to evaluate the environmental impacts of processing residual municipal waste by EfW. It considers the environmental impacts and benefits of each stage of the EfW operation (energy recovery, metals and aggregate recovery and residue landfill) in terms of impact category (climate change, acidification etc.) and the chemical species responsible for each impact.

We also assess the effect on the results of changes in the efficiency of the power generation stage and changes in the type of fuel used to generate the power displaced by the electricity produced by the EfW.

2. Review of previous studies

Several LCA studies have been undertaken to compare different ways of managing specific waste components. Many of these have focussed on comparing recycling with other options. Generally, recycling is the most environmentally-advantageous way of recovering value from uncontaminated source-segregated materials (Chilton et al., 2010; Michaud et al., 2010; Hanan, 2012; Merrild et al., 2012, for example). However, in a study based in Denmark Merrild et al. (2012) noted that burning plastics and cardboard in a high thermal efficiency combined heat and power (CHP) displacing coal fired heat and power, EfW was better than recycling in terms of climate change impacts for both materials and photochemical ozone formation for plastics.

Other authors have considered the entire municipal waste stream and these tended to focus on comparisons between landfill and thermal processing (Gunamantha and Sarto, 2012; Assamoi and Lawryshyn, 2012, for example) or on comparing different thermal processing technologies (for example, Bates, 2009; Watson et al., 2009; Burnley et al., 2012; Rigamonti et al., 2012). The results of these studies were all highly dependent on the thermal efficiency of the energy recovery process and the conventional fuel displaced by the recovery process. Mathiesen et al. (2009) discussed some of the issues in identifying the "marginal technology" (the energy production technology or technologies displaced by the EfW) and noted that making the selection was a complex process which should be subject to sensitivity analysis when performing LCA studies.

Kaplan et al.'s (2009) LCA of EfW and landfill in the USA selected 1 MW h of electrical power production rather than mass of waste managed as the functional unit. They concluded that EfW emitted less $CO_{2(eq)}$, SO_2 and NOx than coal-fired power or from landfill with power generation, but more of each pollutant than conventional gas-fired power. EfW performed better than landfill without energy recovery (where the landfill gas is vented or flared) in terms of $CO_{2(eq)}$ and SO_2, but worse in terms of NOx. A sensitivity analysis found a 12.5% reduction in CO_2 emissions when the ferrous metal was recycled from the bottom ash and the results were highly sensitive to the thermal efficiency of the EfW process. However, the choice of 1 MW h of electrical power as the functional unit did not allow a direct comparison of EfW and landfill as waste management methods. Other impact categories such as toxicity and resource depletion were not considered.

Some authors have considered different EfW technologies such as processing the waste to produce a refuse derived fuel (RDF) followed by conventional combustion, gasification or pyrolysis. RDF production and combustion tended to be less beneficial than burning unprocessed waste unless the RDF could be burned in a much higher efficiency process (Bates, 2009; Rigamonti et al., 2012). Gasification and pyrolysis studies were limited by the lack of reliable data on full-scale facilities, but theoretical studies by Watson et al. (2009) and Burnley et al. (2011) failed to demonstrate any benefits of gasification over conventional EfW in terms of greenhouse gas emissions. However, gasification may result in reduced emissions in other impact categories. Arena and Di Gregorio (2013) compared operational data from a Korean gasifier (which processed untreated residual municipal waste) with typical European 'mass burn' EfWs. Energy efficiency and climate change impacts were not assessed, but it was noted that the gasifier produced a less leachable residue with a greater potential for reuse as an aggregate and so achieved a greater reduction in landfill needs.

The recovery of metals from EfW bottom ash is becoming common practice, not least because metal recovery is financially beneficial and improves the quality of any aggregate recovered from the ash. Grosso et al. (2011) modelled the quantities of steel and aluminium that could be reclaimed from EfW in Italy, but did not consider the environmental implications of this. In a later LCA study Rigamonti et al. (2012) calculated that metals recycling was responsible for around half of the reduction in human toxicity life cycle impacts of EfW and also contributed in a minor way towards climate change emissions reduction.

Clearly, there have been many studies looking at the overall environmental burdens of waste recycling and recovery processes. However few, if any, consider the precise chemical species and material resources that are responsible for the burdens. Knowledge of the overall burdens gives an indication of the areas where improvements should be made to a particular technology, but a detailed breakdown of the burdens is essential in order to prioritise improvements (for example should SO_2 or NO_x be targeted as a priority if acidification impacts are to be reduced?).

A key factor influencing the results of waste LCA studies is the assumption made about the "marginal energy" – the fuel for the conventional power and heat displaced by the waste management system – as pointed out by Mathiesen et al. (2009). This marginal energy is best defined as the electricity and heat that are taken offline when the waste-derived energy is available. Lund et al. (2010) noted that the debate in the literature goes back to 1998. The selection of marginal energy source can often be simply a matter of policy. For example, the UK government position is that power produced by combined cycle gas turbines (CCGT) is the marginal fuel because that represents the current trend in new plant commissioning (DECC, 2008). In contrast, Denmark uses coal as the marginal fuel because one aspect of national policy on climate change is to phase out coal-fired power generation. In reality, the marginal source will vary. For example Lund et al. (2010) commented that where coal is the lowest cost fuel it will be used as base load and therefore only be the marginal fuel during periods of low demand. However, if gas is used to meet peak demand periods, this will be the marginal fuel when power demand is at its peak. The complexity of the situation has been demonstrated in the UK where the use of coal increased by 24% in 2012 (DECC, 2013) due to the low price of coal. Weber et al. (2010) took the USA as an example and noted that it can be almost impossible to determine the electricity mix for a given location at any one time with factors including total power demand, the complexity of the distribution grid and contractual issues confusing the picture. They noted that, in extreme cases, the CO_2 emissions associated with a product or service could differ by a factor of 100 depending on the assumptions made. They concluded that the international community should strive to ensure a consistent approach was taken perhaps through the production of national and regional emission factors for conventionally-produced power.

Cleary (2009) reviewed 20 published waste management LCA studies and noted that eight of them did not mention or were unclear on the source of the displaced energy, six took the marginal fuel to be coal because it was the least efficient source and the remaining six used the national average mix for the country in question. Cleary observed that none of the studies carried out any sensitivity analysis. Mathiesen et al. (2009) reviewed a number of historical LCAs and concluded that, in hindsight, the assumed marginal fuel turned out to be incorrect. They concluded that LCA studies where energy was significant should be subject to sensitivity analysis using a range of marginal energy technologies based on different future scenarios regarding fossil and renewable resource usage and CHP use.

In summary, the literature suggests that recycling materials is generally a better option than EfW for the individual fractions of the waste stream that are amenable to recycling and combustion, but for cardboard and plastics the situation is not always clear cut. EfW is always preferable to landfill for residual waste, but there is no consensus on the relative performance of conventional EfW against other techniques such as gasification or pyrolysis. The importance of the thermal efficiency of the EfW process (Bates, 2009; Merrild et al., 2012), metals recovery (Rigamonti et al., 2012) and the power/heat fuel displaced (Bates, 2009; Merrild et al., 2012; Burnley et al., 2012) have all been noted in previous research, but not investigated in a systematic way. In this research, the "improvement analysis" stage of the LCA process is carried out by considering the sensitivity of the LCA results to these factors and identifying the species responsible for the various impacts. The impact on the results of the assumed marginal energy source is also considered.

3. Method

As far as possible the requirements of the ISO 14040/14044 standard for LCA were followed (BSI, 2006a,b). The analysis was carried out using WRATE (Waste and Resources Assessment Tool for the Environment), an LCA tool developed by the Environment Agency for England and Wales (Burnley et al., 2012). A "consequential approach" was taken in this study. Consequential LCAs look at the impact of a particular decision (for example, the implementation of an EfW project). This means that the analysis considered the marginal source of energy and material manufacture displaced by the EfW implementation rather than the average sources used in "attributional LCAs".

The functional unit of the study was the treatment of 1000 tonnes of residual municipal waste from households. The physical and chemical composition of this waste used the WRATE default values for municipal waste in England in 2007 (Burnley, 2007) and the category composition, moisture content and net calorific value (NCV), or lower heating value, are shown in Table 1. The system studied covered the EfW process and the reprocessing or landfill disposal of the solid residues. The environmental burdens associated with collecting the waste (container provision and collection vehicle burdens) were not included because these would be identical in all the scenarios considered. Also, any environmental burdens arising during the manufacture or use of the materials that formed the waste were excluded (known as the "zero burdens approach"), a common practice in waste-related LCAs. The impacts related to residue transport were also excluded because the differences in these values between the scenarios are relatively small (typical transport distances of 100 km for reclaimed metals and pollution abatement plant residues, 40 km for reclaimed aggregate and 25 km for bottom ash sent to landfill). Environmental benefits due to energy generation, metals and aggregate recycling were included.

Table 1
WRATE default values for UK residual waste composition.

Category	
Paper (%)	17.4
Cardboard (%)	6.6
Dense plastics (%)	6.2
Film plastics (%)	3.8
Textiles (%)	2.8
Nappies and hygiene products (%)	2.3
Wood (%)	3.6
Miscellaneous combustible material (%)	6.1
Miscellaneous non-combustible material (%)	2.7
Garden waste (%)	12.2
Food waste (%)	19.3
Glass (%)	7.9
Ferrous metals (%)	3.1
Non-ferrous metals (%)	1.3
Fine material (%)	2.0
Waste Electronic and Electrical Equipment (WEEE) (%)	2.2
Hazardous household waste (%)	0.5
Overall net calorific value (MJ kg^{-1})	8.88
Overall moisture content (%)	30.3

The life cycle inventories were calculated using WRATE's databases which were compiled from several sources. The capital and operating burdens of the EfW, materials recycling and landfill processes were obtained by the Environment Agency from published sources and from discussions with plant operators, and were subjected to peer-review by a group of experts (industrialists, consultants and academics) from organisations under contract to the Environment Agency before their inclusion in WRATE. Inventories for resources used (such as lime, ammonia and carbon consumption by the EfW) were taken from the ecoinvent LCA database version 2.1 (Frischknecht et al., 2005).

A consequential LCA implies that all facilities are taken to represent the state of the art. In areas with a highly developed district heating network, the state of the art would be represented by high efficiency EfW plants operating in combined heat and power mode where the overall thermal efficiencies can exceed 100% on a lower heating value basis through the use of flue gas condensation and heat pumps (Damgaard et al. (2012). However, this is not the case in the UK where CHP EfW systems are uncommon and there are uncertainties over the heating load factor, as discussed by Giugliano et al. (2008). Therefore this study used WRATE data obtained from the Basingstoke EfW in Hampshire, a power-only system operating at a net energy efficiency of 21%. The plant is of a moving grate design with a capacity of 95,000 tonnes per year. The pollution control system comprises selective non-catalytic reduction (SNCR) NOx control, dry scrubbing with lime and carbon injection followed by bag filtration. Typical annual reagent consumption figures are lime 912 tonnes, urea 140 tonnes and activated carbon 17 tonnes. To provide a comparison with the higher efficiency plant becoming more common in Europe, a sensitivity analysis was carried out by considering a plant with an efficiency of 29% based on the highest values for power-only plants reported in the literature (Murer et al., 2011).

As discussed above, there is much debate and uncertainly regarding the "marginal" source of power displaced by the EfW process. In this work, we have assumed that a future government priority will be to minimise the carbon intensity of power generation so EfW would displace coal. As part of the sensitivity analysis and also to consider the current UK policy on marginal power, the effect of displacing CCGT as the marginal source was considered.

The environmental burdens were categorised and then characterised using the ecoinvent database (Frischknecht et al., 2005) to calculate the environmental impacts. The categories used are a sub-set of the CML 2001 (Guinée, 2002) categories considered by the UK's Department for Environment Food and Rural Affairs (Defra) to be most relevant for LCAs related to municipal waste management.

- Global warming potential over 100 years expressed as kg CO_2 equivalent.

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- Acidification potential expressed as kg SO_2 equivalent.
- Generic eutrophication potential expressed as kg PO_4 equivalent.
  - Freshwater aquatic ecotoxicity over an infinite time period expressed as kg 1,4-dichlorobenzene equivalent.
  - Human health over an infinite time period expressed as kg 1,4-dichlorobenzene equivalent.
- Depletion of abiotic resources expressed as kg antimony equivalent.

The results were then normalised by comparing the impact with that produced in a year by the average EU 15 (pre-enlargement) citizen, referred to as Euro-persons equivalent (EP) for the year 1995 (Huijbregts et al., 2001).

3.1. Scenarios considered

To determine the significance of each aspect of the energy recovery process on the overall environmental impact, a baseline scenario representing common UK EfW practice was compared with three variations. The four scenarios were therefore:

1. **Baseline;** a power-only EfW plant operating at a net thermal efficiency of 21%. The values for the atmospheric emissions were taken from the plant during normal operation. In common with most facilities, the emission levels are lower than the emission limit values to ensure that these limits are achieved. Ferrous metal was recovered from the bottom ash at the EfW plant. The remaining bottom ash was further processed to recover ferrous and non-ferrous metals and a low-grade aggregate for recycling. The overall efficiency of these recovery processes was set at 80% and 70% for ferrous and non-ferrous metal recovery respectively (based on values supplied by the WRATE peer-reviewer). It was assumed that there would be no degradation in metal quality, so substitution of virgin metals would take place on a one-to-one basis. These recovery rates compare with values reported by Grosso et al. (2011) of up to 83% for ferrous metal and 70% for aluminium. The pollution abatement residues (combined fly ash and dry scrubber residue) and the material rejected by the bottom ash recovery process were assumed to be landfilled without treatment in hazardous and non-hazardous sites respectively that meet the requirements of the EU Landfill Directive in terms of liner design. Based on the WRATE mass balance, the masses of these residues sent to landfill are 45 kg per tonne of waste entering the incinerator for pollution abatement residue and 11.4 kg per tonne of waste for the unrecovered bottom ash.
2. **No aluminium recovery;** as baseline, only ferrous metal is reclaimed from the ash.
3. **No metals recovery;** as baseline, but with no metal reclamation at the EfW or during ash processing.
4. **No reclamation;** as baseline but with no metal or aggregate recovery from the residues.

These scenarios were compared with the alternative of sending the untreated waste to a landfill site that meets the requirements of the Landfill Directive where the landfill gas produced from the biodegradable carbon in waste is combusted to generate electricity. The WRATE model assumes that the biodegradable carbon degrades under a first-order reaction using different rate constants for the rapidly, medium and slowly degrading materials. Over the 100 year timescale of the model, a proportion of the carbon does not degrade so is "locked up" in the landfill. However, unlike some LCA models (such as EASETECH) WRATE does not allow the user to allocate a negative CO_2 emission to this sequestered carbon as advocated by some researchers (see for example, Christensen et al. (2009)).

Sensitivity analyses considered the impact of changing the marginal fuel displaced from coal to CCGT and increasing the energy efficiency of the EfW from 21% to 29%.

To assess the repeatability of the results, the baseline scenario LCA was repeated using the EASETECH LCA tool. The unit operations were taken from the EASETECH database and the impact methodology and normalisation factors recommended by the International Reference Life Cycle Data System (ILCD) were used (European Commission, 2010).

4. Results and discussion

4.1. Overall

The overall results for the baseline EfW and landfill scenarios are shown in Table 2. Note that a negative value represents a reduction in that impact category so constitutes an environmental benefit. Table 3 presents a detailed breakdown of the chemical species and materials responsible for the environmental burdens in each category for the baseline scenario. These data are of particular importance for identifying the emissions that contribute to the environmental impacts of EfW and for determining the most cost-effective way of improving the environmental performance of EfW. However, many published LCA studies fail to report these values.

Table 2 shows that conventional EfW is environmentally more advantageous than landfill as a means of managing residual municipal waste in each impact category. These results confirm previous studies (Hanan, 2012; Gunamantha and Sarto, 2012; Assamoi and Lawryshyn, 2012, for example).

The baseline EfW system achieves an environmental benefit for all categories apart from eutrophication. The eutrophication impacts are principally due to atmospheric emissions of oxides of nitrogen (NOx) (Table 3). These results refer to a plant fitted with SNCR equipment to reduce NOx emissions to values in the range 120–180 mg m^{-3} (compared to the EU Industrial Emissions Directive (2010/75/EU) limit of 200 mg m^{-3} (European Commission, 2010). The effectiveness of the SNCR system is partly dependent on the quantity of ammonia injected into the furnace, but as this increases, the ammonia discharged to the environment (or slip) also increases. This effect was investigated by Møller et al. (2011) for an EfW with an SNCR system and wet scrubber. The authors estimated that unreacted ammonia leaving the furnace was partitioned between the fly ash and scrubber effluent in the proportions 37:63. They went on to conclude that SNCR is only

Table 2
Environmental burdens by scenario (Europersons equivalent per1000 tonnes residual waste burned).

Scenario		Total	Power generation	Materials recovery	Landfill
Baseline	Climate change	−33	−23	−9.9	0.01
	Resource depletion	−140	−118	−22	0.06
	Acidification	−13	−5.4	−8.0	0.01
	Eutrophication	0.3	1.8	−1.5	0.05
	Human toxicity	−27	−1.5	−26	0.3
	Aquatic ecotoxicity	−35	−3.6	−33	1.4
Landfill	Climate change	10	xx	0	10
	Resource depletion	−40	xx	0	−40
	Acidification	−2.6	xx	0	−2.6
	Eutrophication	8.0	xx	0	8.0
	Human toxicity	−0.4	xx	0	−0.4
	Aquatic ecotoxicity	0.4	xx	0	0.4

xx – Included in landfill category.

Table 3
Baseline impacts by substance (Europersons equivalent per 1000 tonnes residual waste burned).

	Medium	Total	Power generation	Metal/aggregate recovery	Residue landfill
Total climate change		*−33.1*	*−23.1*	*−9.9*	*0.01*
Fossil CO_2	A	−27.4	−19.4	−8.0	0.008
Methane	A	−5.1	−4.6	−0.5	0.0005
Tetrafluoromethane	A	−1.3	0.001	−1.3	0.000003
Carbon monoxide	A	−0.1	−0.001	−0.1	0.0001
Others		0.8	0.9	0	0.0014
Total resource depletion		*−140*	*−118*	*−22.4*	*0.06*
Coal	R	−128	−114	−14	0.003
Crude oil	R	−6.1	−0.8	−5.4	0.05
Natural gas	R	−1.1	−0.03	−1.1	0.0004
Others		−4.8	−3.2	−1.9	0.05
Total acidification		−13.4	−5.73	−8.0	0.01
Sulphur dioxide	A	−12.6	−6.14	−6.45	0.008
Ammonia	A	−0.237	−0.09	−0.15	0.0001
Nitrogen oxides	A	2.0	3.57	−1.57	0.005
Others		0.037	−0.09	0.06	−0.003
Total eutrophication		*0.31*	*1.8*	*−1.5*	*0.05*
Phosphate	W	−0.414	−0.02	−0.399	0.003
COD	W	−0.248	−0.04	−0.214	0.001
Ammonia	A	−0.111	−0.07	−0.04	−0.16
Nitrous oxide	A	0.171	0.19	−0.018	0
Nitrogen oxides	A	1.11	1.99	−0.875	0.003
Others		−0.018	−0.05	−0.024	0.203
Total human toxicity		*−26.9*	*−1.5*	*−25.7*	*0.3*
PAH	A	−23.1	−0.2	−23	0.001
Hydrogen fluoride	A	−1.9	−1.0	−0.9	0.00004
Vanadium ion	W	−1.0	−0.1	−0.9	0.00007
Selenium	W	−0.2	−0.2	−0.2	0.00002
Nickel	A	−0.2	−0.1	−0.2	0.0002
Arsenic	A	−0.2	−0.1	−0.05	0.0001
Barium	W	0.2	−0.01	−0.004	0.2
Others		−0.5	0.21	−0.5	0.1
Total aquatic ecotoxicity		*−35.2*	*−3.6*	*−33*	*1.4*
Vanadium ion	W	−31.4	−1.53	−29.9	0.002
Tributyltin compounds	W	−2.08	−0.982	−1.1	0.001
Beryllium	W	−0.893	−0.374	−0.465	0.0001
Barite	W	−0.212	−0.03	−0.182	0.002
Copper ion	W	−0.2	−0.05	−0.05	0.0002
Selenium	W	−0.159	−0.03	−0.132	0.0006
PAH hydrocarbons	A	−0.104	−0.0006	−0.103	0
Molybdenum	W	0.108	−0.017	−0.009	0.135
Nickel	W	0.257	−0.001	0	0.258
Barium	W	0.725	−0.04	−0.02	0.785
Others		−0.342	0.3546	−1.969	0.176

A = air, W = water, and R = resource.

beneficial in terms of eutrophication and acidification burdens if a maximum of 10–20% of the ammonia captured by the scrubber is released to the environment and less than 40% of the ammonia slip to the fly ash is released to the environment. Further work is required to establish the optimum ammonia dosage for systems incorporating dry or semi-dry scrubbing systems where the bulk of the unreacted ammonia would be discharged to atmosphere.

Had the EfW been equipped with selective catalytic reduction (SCR) control, the atmospheric NOx emission concentration would be reduced to a value in the range 40–70 mg m^{-3} (European Commission, 2006). The effect of this on the eutrophication burdens is shown in Table 4 where the EfW-related burdens are reduced from 2.0 Europersons equivalent per 1000 tonnes to −0.9 Europersons equivalent per 1000 tonnes due to a reduction in NOx emissions. Whether this potential improvement in emissions justifies the additional cost would need to be determined on a case-by-case basis and would depend on the relative costs of the two systems. The European Commission (2006) reported total costs (capital and operating) of €3.02 per tonne and €1.59 per tonne for SCR and SNCR respectively based on a 150,000 tonnes per year plant in Austria. Local conditions, such as existing ambient NOx levels, also play a part in the decision about which technology to use. For example, in urban areas where ambient NOx levels are close to, or in excess of, local air quality standards, more stringent EfW emission levels may be appropriate.

Table 4
Eutrophication impacts for baseline plant fitted with SCR NOx control (Europersons equivalent per 1000 tonnes residual waste burned).

	Medium	Total	Power generation	Metal/aggregate recovery	Residue landfill
Eutrophication		*−2.42*	*−0.904*	*−1.57*	*0.05*
Phosphate	W	−0.414	−0.019	−0.399	0.003
COD	W	−0.248	−0.035	−0.214	0.001
Ammonia	A	−0.13	−0.09	−0.04	−0.16
Nitrous oxide	A	0.171	0.19	−0.018	0
Nitrogen oxides	A	−1.77	−0.902	−0.875	0.003
Others		−0.029	−0.048	−0.024	0.203

The climate change and resource depletion benefits of EfW are both due to the reduction in fossil fuel combustion related to the power exported and metals recycled. The CO_2 emissions from combustion of plant or animal based materials in the waste (paper, food and garden waste and natural textiles) are from short cycle carbon and are regarded as being renewable, and hence do not contribute towards climate change impacts. Acidification reduction is due to the reduction in SO_2 emissions associated with displacing coal use in both power generation and in the extraction of metals from ores.

Despite the many concerns expressed over the years about the emissions of dioxins and furans from EfW it is interesting to note that these compounds do not make any significant contributions to the burdens in any of the categories as shown in Table 3.

Landfill represents the only alternative to thermal processing for the disposal of non-recyclable waste and it is important to note that the landfill system is also environmentally beneficial with respect to resource depletion, acidification and human toxicity. These benefits are all due to the power displaced by landfill gas recovery. The escape of some of the landfill gas and ammonia and metals in leachate discharges are responsible for the adverse impacts in terms of climate change, eutrophication and aquatic eco-toxicity respectively.

The composition of the municipal waste is based on WRATE data for English residual waste from the year 2007 when the national recycling rate was around 30% (Burnley et al., 2012). The physical and chemical composition and the heating value of the residual waste will vary with the scope of any kerbside recycling scheme in place and recycling rates have increased significantly since that date. However, a recycling scheme designed to maximise both dry recycling and composting/anaerobic digestion would have to collect high NCV and low moisture material such as paper and plastics, low NCV and high moisture material such as food and garden waste and zero NCV material such as glass and metals. Therefore the implementation of such a scheme would have very little effect on the overall NCV of the residual waste, as illustrated in Table 5. This analysis assumes that a recycling policy such as applies in the EU would result in a drive to maximise the collection of both wet and dry recyclables. It is considered that this is the most realistic scenario, but intensive collection of only wet or dry recyclables would result in a change in NCV, but not to the extent that the impact on the overall results would be significant.

The results for the EASETECH assessment of the baseline scenario are shown in Table 6. Whilst a direct numerical comparison cannot be made between the two sets of results, comparing Tables 2 and 6 shows a number of common trends:

- The relative importance of the power generation and materials recovery processes for climate change and resource depletion.
- The importance of materials recovery in achieving an overall benefit in terms of acidification, human toxicity and ecotoxicity.
- The overall contribution to eutrophication, only partially offset by materials recovery.

This agreement gives confidence in the reliability of the two LCA tools, the process data used and the assumptions made. Similar conclusions were reached by Turconi et al. (2011) who used SimaPro and EASEWASTE (the forerunner of EASETECH) to compare EfW in Italy and Denmark. They demonstrated that the key differences between the two locations were due to the difference in fuel substituted by the EfW and the atmospheric emissions from the EfW plants.

4.2. Metals and aggregate recovery

The effects of different metal and aggregate recovery strategies on the baseline results are shown in Fig. 1. This demonstrates the importance of aluminium recovery in achieving the maximum benefits to human toxicity and aquatic eco-toxicity impacts. Inspection of Table 3 shows that these benefits are due to the reduction in the emissions of polycyclic aromatic hydrocarbons (PAHs) to air (related to anode production and electrolysis of the alumina) and vanadium emissions to water (related to electricity generation for the electrolysis process) associated with virgin aluminium production. If aluminium is not recovered, EfW still results in a reduction in impacts in these categories, but the benefits are marginal. Aluminium recovery also improves the performance in all the other impact categories. Ferrous metal recovery is also environmentally beneficial, but these benefits are small in comparison with those arising from aluminium recovery. These results confirm the findings of Rigamonti et al. (2012) with respect to climate change and human toxicity whilst providing additional insight into other impact categories. The EASETECH results (Table 6) confirm the importance of metal recovery in reducing toxicity impacts. In all categories the relative contributions of metal recovery and energy recovery were similar.

Table 5
Impact of intensive recycling separation on calorific value of residual waste.

Component	Component CV ($MJ\ kg^{-1}$)	Initial waste composition (%)	Proportion reclaimed for recycling (%)	Residual waste composition (%)
Paper	10.751	17.4	90	5.7
Cardboard	10.751	6.6	90	2.2
Dense plastics	24.901	6.2	90	2.0
Film plastics	21.32	3.8	60	5.0
Textiles	14.338	2.8	50	4.6
Nappies and hygiene products	5.453	2.3	0	7.5
Wood	13.867	3.6	70	3.5
Miscellaneous combustible material	13.867	6.1	0	19.9
Miscellaneous non-combustible material	2.525	2.7	0	8.8
Garden waste	4.577	12.2	85	6.0
Food waste	3.393	17.4	80	11.4
Glass	0	7.9	95	1.3
Ferrous metals	0	3.1	95	0.5
Non-ferrous metals	0	1.3	95	0.2
Fine material	3.464	2	0	6.5
WEEE	0	2.2	0	7.2
Hazardous household waste	0	0.5	0	1.6
Overall NCV ($MJ\ kg^{-1}$)		8.249		8.341

Table 6
Baseline scenario impacts from EASTECH (Europersons equivalent per 1000 tonnes residual waste burned).

	Total	Power generation	Materials recovery	Landfill
Climate change (IPPC 2007)	−38	−25	−14	0.2
Fossil resource depletion (CLM 2012)	−99	−89	−12	2
Acidification (ReCiPe)	−11	−1	−10	0.1
Eutrophication (CLM 2001)	1	1	−0.1	0.02
Human toxicity carcinogenic (USEtox)	−501	−1	−506	6
Human toxicity non-carcinogenic (USEtox)	−2395	−6	−2389	0.1
Aquatic ecotoxicity (USEtox)	−4	0.06	−6	2

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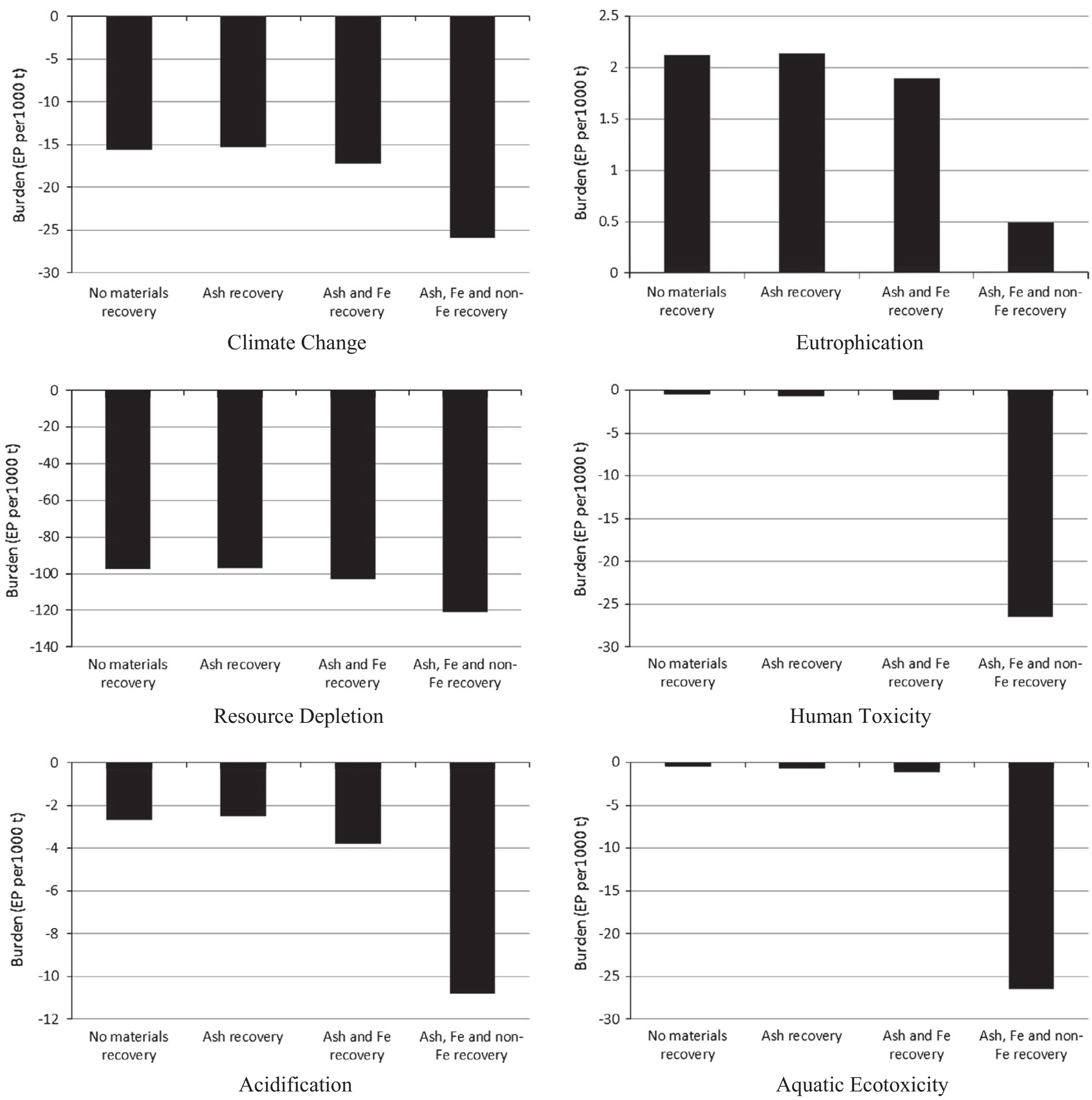

Fig. 1. Effect of aggregate and metal recovery on baseline scenario burdens (Europersons equivalent per 1000 tonnes of residual waste burned).

WRATE and EASETECH both assume that the displaced aluminium is manufactured using power generated by the European average mix of fuels. These benefits would be reduced if it is assumed that the displaced aluminium was produced using renewable sources such as hydropower. Furthermore, like all LCA studies, this analysis takes no account of the geographical location of the environmental impacts and benefits. This presents no issues for global impacts such as climate change, but would need to be taken into account for impacts such as toxicity, eutrophication and acidification, if the results were to be used in a more locally-based environmental impact assessment study.

Aggregate recycling has financial benefits to EfW operators in the UK in that it reduces the needs for landfill and the associated costs of disposal and landfill tax. However, a comparison of the impacts for the "no materials recovery" and "ash recovery" options in Fig. 1 demonstrates that the impacts of replacing aggregates by bottom ash are minimal. In fact, there is a marginal worsening in environmental performance in all categories apart from toxicity impacts, with this slight toxicity benefit due to reductions in metal discharges in landfill leachate.

4.3. Energy recovery efficiency

Fig. 2 demonstrates the importance of maximising the efficiency of energy recovery. An increase in net efficiency from 21% (conventional EfW) to 29% (high efficiency EfW) has significant benefits for climate change, resource depletion and acidification. These effects are all due to the reduction in the burning of fossil fuels to generate electricity and the associated emissions. The increase in net benefits is proportionately greater than the increase in thermal efficiency, due to the parasitic load (the power required to operate the EfW) being largely independent of plant efficiency. In the case of eutrophication, increasing efficiency shows a transformation from an overall burden to an overall benefit. This is because the NOx emissions from the EfW (the main source of eutrophication burdens) are largely independent of energy efficiency,

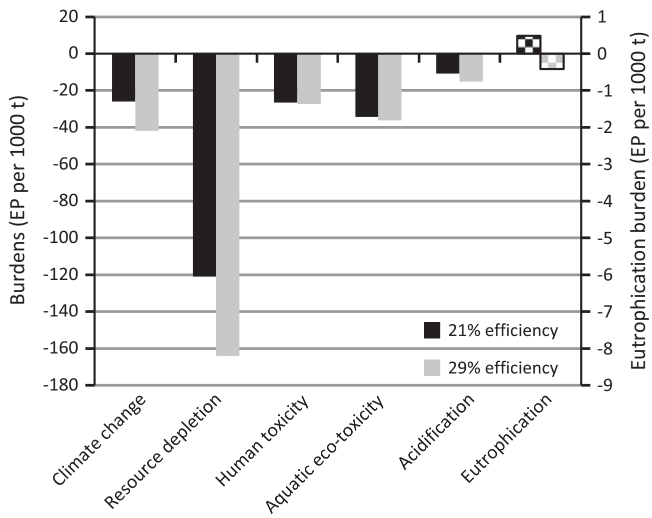

Fig. 2. Effect of electricity generation efficiency on baseline scenario burdens (Europersons equivalent per 1000 tonnes of residual waste burned – note eutrophication burdens – hatched columns – refer to the right hand axis).

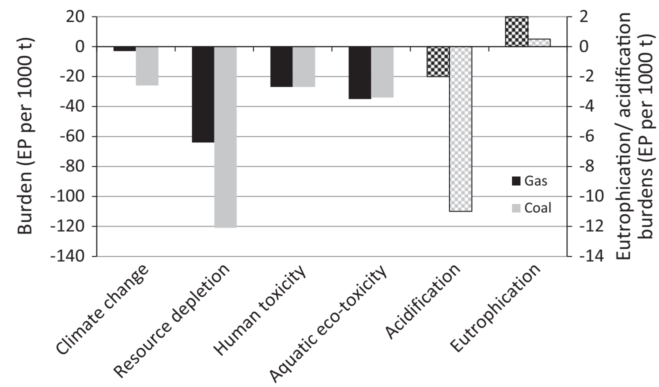

Fig. 3. Effect of conventional power source displaced on baseline scenario burdens (Europersons equivalent per 1000 tonnes of residual waste burned – note acidification and eutrophication burdens – hatched columns – refer to the right hand axis).

whereas the avoided burdens from conventional fuel power generation are directly proportional to the energy efficiency of the EfW. The toxicity benefits are largely independent of the efficiency of the energy recovery stage because these benefits are almost entirely due to displacement of metal extraction from ores (particularly aluminium) by the metal recovery processes as discussed in Section 4.2 above.

High efficiency EfW relies on achieving higher initial steam temperatures that increase the boiler superheater tube temperatures. This will lead to increased boiler tube corrosion rates and possibly a need to increase the frequency of tube replacements or to make them from more costly materials. Further work is required to determine fully the financial and environmental benefits of operation at higher efficiencies over extended time periods.

Similar environmental improvements would be possible through the adoption of the relatively more efficient CHP EfW as demonstrated by Rigamonti et al. (2012). However, this requires a reliable and constant demand for process/space heating which is not always available, particularly in countries without a developed heat distribution infrastructure. Even where this infrastructure exists, CHP load factors can be low. For example, the Carbon Trust (2008) estimates that a CHP scheme providing space heating for offices in the UK would have a load factor (percentage of the time that the plant was operating averaged over a year) of 20% and a scheme providing process heat would still only achieve a load factor of 60%, restricting the EfW operation to power-only mode for at least 40% of the time.

4.4. Conventional fuel displacement

The most significant factor affecting the climate change and abiotic resource consumption impacts of EfW is the assumption made regarding the type of power displaced as shown in Fig. 3. This illustrates the effect of electricity from EfW displacing high-carbon electricity produced from coal and lower carbon electricity produced from combined cycle gas turbines. In terms of toxicity, there is no difference between the two cases owing to the overwhelming effect of metal recovery as discussed above.

The UK electricity sector is wholly in private ownership, but UK government policy assumes that new generating capacity will be supplied by building CCGT plant and therefore this is the marginal fuel that would be displaced by power generated from EfW, as discussed in Section 2. The authors do not contend this assumption for decisions to build new electricity capacity, but it should be noted that no one faced with a potential shortfall in electricity supply would countenance waste as a prime fuel. It has a low calorific value, is highly heterogeneous and has a high potential for causing pollution. Furthermore, UK EfW plant outputs are typically 10–20 MW, a scale of generation that is one or two orders of magnitude lower than conventional fossil fuel electricity generation.

One needs therefore to draw a distinction between decisions relating to energy policy, waste policy and investment in energy and waste management infrastructure. A government may have a policy to encourage a switch from landfill to EfW for residual waste management and this could provide an additional 400 MW of renewable electricity generation capacity in the UK. Although related to energy and contributing to meeting renewable energy targets, this would be a waste policy that has an impact on the energy market, not an energy policy. Clearly, a decision to build an EfW plant is not a decision about how additional power is generated; it is a decision about how some residual waste is managed in a way that reduces the overall environmental burden. We know that demand for electricity varies throughout the day and between days, therefore if EfW is to displace any electricity generation, it should be that generated by the fuel that would be switched off as no longer required.

With certain exceptions, electricity generation plant is selected to meet the forecast demand in half hourly periods on the basis of prices bid-in by producers. Of the available electricity sources and fuels, the authors assumed that nuclear, wind and 'other' (eg solar) generation always ran if available, the interconnectors (linking France and England, the Netherlands and England) were not marginal.

The short-run marginal plant is the plant that would have run had demand been higher or that would not have run had a cheaper plant or a, renewable source (for example EfW) been available. In the latter case, this would have been the generating unit with the highest bid-in price (for utilised plant) in that half hour period. Such price data are confidential and were not available to the authors. Therefore to approximate the short-run marginal source, half hourly data for electricity supplied by type were analysed. These data covered three month periods for summer and autumn/winter: July to September 2014 and mid-October 2014 to mid-January 2015. The maximum amount of electricity supplied by each plant type in any half hour during the period was found and this maximum supplied was assumed to be the capacity of that plant type throughout the period considered. For each half hour period the difference between the amount supplied by each plant

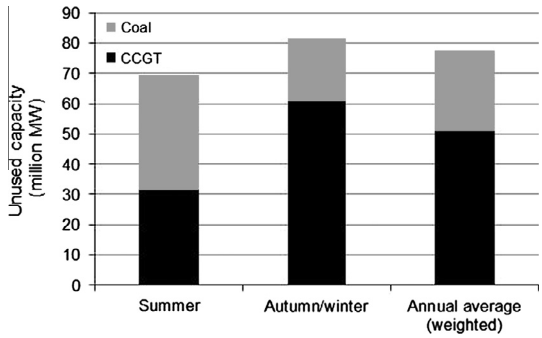

Fig. 4. Unused power generation capacity in the UK (million MW) (note nuclear, renewable and minor sources are excluded).

Table 7
Effect of source of conventionally-produced power on baseline and landfill scenarios (Europersons per 1000 tonnes residual waste burned).

Fuel displaced	Coal		Gas	
	EfW	Landfill	EfW	Landfill
Climate change	−33	10	−1.9	21
Resource depletion	−140	−40	−54	−13
Acidification	−13	−2.6	−1.1	1.4
Eutrophication	0.31	8.0	3.0	8.6
Human toxicity	−27	−0.4	−26	−0.2
Aquatic ecotoxicity	−35	0.4	−33	0.7

type and the maximum was calculated. This was in effect the total of plant capacity of each type that was available for electricity generation but was not used for each half hour in the period. The overall marginal mix for the period was therefore the relative amounts of the totals for each fuel type available for generation but not used in the 3 months.

Fig. 4 shows that the calculated, marginal mix for the summer period (55:45 coal and CCGT fired plant is very different from that in the autumn/winter period (27:73 coal and CCGT fired plant). This may be due to several factors but is most likely a combination of much lower demand in summer and generating units being off-line due to planned maintenance. The annual average marginal mix was calculated assuming that the summer marginal mix was representative of four months, with the autumn/winter mix accounting for the balance. Based on this assumption, the annual average marginal mix is 35:65 coal and CCGT.

Therefore it is this mix that should be used as the offset for electricity displaced by EfW plant and not CCGT, which is the long-run marginal. Regardless of the assumptions made about the fuel displaced, the comparison to be made when assessing EfW is with the alternative waste management options. In the case of non-recyclable residual waste the comparison should be with landfill. As shown in Table 7, EfW is the better option for the displacement of coal or gas.

5. Conclusions and recommendations

This study has demonstrated that, under UK conditions, EfW is a significantly more environmentally beneficial method of managing residual municipal waste than is landfill.

The benefits of EfW increase with higher thermal efficiency and with the amount of metal, particularly aluminium, reclaimed from the residues.

The overall environmental impacts of EfW are highly dependent on the fuel for the power displaced by the electricity generated by the EfW. It is recommended that, to provide consistency, future LCA studies should include appropriate low and high carbon intensity fuels as part of the sensitivity analysis.

All LCA studies are dependent on the processes and impacts that are included in the study and on the assumptions made. However, the consistency of the results from the different studies discussed in the literature survey and the agreement between WRATE and EASETECH lends support to the use of LCA for the purposes of assessing waste management options.

One of the purposes of LCA is to identify areas for improving the environmental performance of the different technologies. To aid this, the chemical species responsible for the key impacts should be clearly identified. For example, this research has identified the importance of controlling the emissions of oxides of nitrogen to reduce both acidification and eutrophication impacts.

To obtain a clearer picture of the benefits, EfW should be compared with several forms of pre-treatment followed by landfill – the alternative management route for residual waste.

Further research is required to establish the long-term costs and environmental impacts of high efficiency EfW facilities.

There is an urgent need for clarity as to future generation policy, not in terms of what type of generation capacity will be built but rather for how long coal-fired power stations will remain operational and what will be the marginal electricity fuel, rather than the marginal electrical generation build.

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