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Life cycle assessment of municipal solid waste management in Nottingham, England: Past and future perspectives

Dan Wang [a], Jun He [a]*, Yu-Ting Tang [b]*, David Higgitt [c], Darren Robinson [d]

[a] International Doctoral Innovation Centre, Department of Chemical and Environmental Engineering, University of Nottingham Ningbo China, Ningbo, Zhejiang, PR China

[b] School of Geographical Sciences, University of Nottingham Ningbo China, Ningbo, Zhejiang, PR China

[c] School of Architecture, University of Sheffield, Sheffield, United Kingdom

[d] Lancaster University College at Beijing Jiaotong University, Weihai, Shandong, China

* Correspondence to:

Dr Jun He, email: jun.he@nottingham.edu.cn;

Dr Yu-Ting Tang, email: yu-ting.tang@nottingham.edu.cn;

Abstract

Since the enforcement of the EU Landfill Directive, EU waste directives were successively enforced in EU member states to facilitate the establishment of sustainable MSW management. Various changes have been made in England to reduce the global impact of its MSW management, but the effectiveness of these changes on mitigating the global warming potential (GWP) from MSW management has never been investigated in detail. This study assessed the historical GWP of MSW management in Nottingham throughout the period from April 2001 to March 2017 through life cycle assessment (LCA). The LCA results indicate continuous reductions in greenhouse gas (GHG) emissions from MSW management during the study period due to improvements in waste collection, treatment and material recycling, as well as waste prevention. These improvements resulted in a net reduction of GHG emission from 1076.0 kg CO_2–eq./t of MSW (or 498.2 kg CO_2–eq./Ca) in 2001/02 to 211.3 kg CO_2–eq./t of MSW (or 76.3 kg CO_2–eq./Ca) in 2016/17. A further reduction to –142.3 kg CO_2–eq./t of MSW (or –40.2 kg CO_2–eq./Ca) could be achieved by separating food waste from incinerated waste, treating organic waste via anaerobic digestion and by pretreating incinerated waste in a material recovery facility.

Keywords: EU waste directives; municipal solid waste; evolution; life cycle assessment; global warming potential; Nottingham.

1. Introduction

Climate change is one of the most serious of current international concerns, to which municipal solid waste (MSW) management is a significant contributor, through greenhouse gases (GHG) emissions (Turner et al., 2016, Kaza et al., 2018), such as methane resulting from the decomposition of biodegradable municipal waste (BMW) (El-Fadel et al., 1997). MSW and landfills are the third largest anthropogenic source of global CH_4 emission (Das et al., 2019). In 2016, the greenhouse gas (GHG) emissions from the waste management sector were 1.6 billion tons of CO_2-eq., accounting for 5% of global emissions (Kaza et al., 2018). To mitigate the global warming potential (GWP) of MSW management, the EU Landfill Directive (EU Directive 99/31/EC) was introduced in 1999 to reduce the quantity of BMW sent to landfill, and setting a target of lowering the amount of landfilled BMW to 35% of that in 1995 by 2016 (EC, 1999). Subsequently, regulations have been successively introduced to divert waste from landfill to more environmental friendly treatment options such as recycling, composting and energy recovery, with corresponding management targets (Table S1). EU member states were legally obligated to establish and enforce regional policy instruments to meet these targets. Furthermore, the EU Waste Framework Directive (EU Directive 2008/98/EC) established the "waste management hierarchy" to guide the practice of sustainable waste management. These EU Directives have gradually promoted the establishment of sustainable MSW management, which has the ability to harness resource from waste in the form of materials and energy (Liang and Zhang, 2012, Cobo et al., 2018). To achieve the targets set in EU Directives, a variety of strategies, technologies and techniques aiming at material recycling and energy recovery from waste, as well as waste prevention, have been introduced in the last two decades in England, but their realistic effects on the improvement of the performance of MSW management has not to date been investigated.

A number of studies have been conducted to assess the evolution of MSW management, and the pros and contras of the corresponding policies and strategies. Uyarra and Gee (2013) investigated the transformation of waste management in Greater Manchester from a simple landfill model to a complex, multi-technology waste solution based on intensive recycling and composting, and sustainable energy usage. Pomberger et al. (2017) assessed the performance of MSW management concerning the rate of landfilling, incineration, recycling and composting, from 1995 to 2014 in Europe. Castillo-Giménez et al. (2019) assessed the performance and convergence in the treatment of MSW by the EU-27 during the period 1995-2016, by country and year. However, these studies focused on the final destinations of waste, paying less attention to the environmental impacts of changing MSW management practices from a life cycle perspective. This latter is of interest, since it has the potential to show for example that landfill could be a desirable waste treatment option when landfill gas to energy is considered (Khandelwal et al., 2019). Besides, waste prevention, which ranks at the top of the waste management hierarchy, has seldom been considered as an indicator in evaluating the performance of MSW management.

Life cycle assessment (LCA) has been extensively applied to evaluate environmental burdens associated with MSW management (Fernández-Nava et al., 2014, Yay, 2015, Milutinović et al., 2017, Coelho and Lange, 2018). But in addition to quantifying the environmental impacts and burdens associated with waste management options, LCA can also be used to explore opportunities for improvements (Cherubini et al., 2009). It also helps to expand the perspective beyond the waste management system. This makes it possible to take the significant environmental benefits that can be obtained through alternative waste management options into account; for example, energy-from-waste (EfW) reduces the consumption of energy from fossil fuels; recycled materials replace part of virgin materials; and the compost from biological treatment substitutes the production of chemical fertilizers

(Franchetti and Kilaru, 2012, Jeswani et al., 2013, Turner et al., 2016). On the other hand, LCA results can be affected by multiple factors such as the definition of system boundary, the assumptions in life cycle inventory (LCI), and the methodologies or software adopted for calculation (Yadav and Samadder, 2018, Zhou et al., 2018a, Khandelwal et al., 2019). There are a number of impact assessment methods (e.g. CML, EDIP, IPCC 2013) and more than 50 LCA software (e.g. SimaPro, Gabi, WASTED) available to aid the performing of LCA (Yadav and Samadder, 2018). Winkler and Bilitewski (2007) pointed out that the LCA results calculated by different models showed high variation and not negligible, even led to contradictory conclusions in some cases. Therefore, sensitivity analysis is often included in the assessment to inform the robustness of the LCA results and the potential for improvement (Khandelwal et al., 2019).

Most LCA studies have focused on the environmental impacts associated with the present and possible future MSW management at specific sites, with less attention paid to the evolution of an MSW management system in a historical context. Habib et al. (2013) assessed the GWP of MSW management in Aalborg, Denmark from 1970 to 2010, with the focus on the effect of EfW. Zhou et al. (2018b) evaluated the environmental performance evolution of MSW management in Hangzhou, China, focusing on the treatment technologies and source-separated collection. Evaluation of the environmental impacts over time reveals and documents the trend in environmental impacts of a given waste management system for the study site, or whether there has actually been progress towards a more environmentally friendly waste management strategy (Poulsen and Hansen, 2009).

On the basis of the research gaps identified above, this study attempts to evaluate how the implementation of new waste management options and regulations over time has affected the GWP of MSW management at a selected city by quantifying the GHG emissions from MSW management scenarios at different stages of development using LCA. Nottingham in Eastern

England was chosen as it has changed its MSW management strategy several times since the implementation of the EU Landfill Directive, beginning with combined landfilling and incineration with energy recovery and ending at present with a combination of source separation, recycling, composting and incineration with energy recovery, and ambitious MSW management targets have been set. The balance for GHG has been evaluated for three specific years: 2001/02, 2006/07 and 2016/17, and a future scenario which would potentially reach the 2025 recycling target and 2030 landfill target set by Nottingham City Council (Section 2.1). The results provide an insight into how the waste management policies and regulations drive the improvement of waste management, and hence support local policy and decision making by identifying the areas where the enforcement of policies, regulations, strategies and technologies can be strengthened in the future development of MSW management, as reference to other similar cities.

2. Methodology

2.1. Study city

Nottingham is one of the Core Cities in England, located in the central UK (52° 57' N and 1° 09' W) (Fig. 1). It covers an area of 7,5378 hectares and had an estimated population of 329,200 in 2017 (Nottingham Insight, 2018). Since the start of the new millennium, new waste management strategies, measurements and technologies were adopted in Nottingham to divert waste from landfill, as well as to prevent unnecessary waste generation. As a result, the quantities of waste generated and landfilled were significantly reduced (Fig. S1). A kerbside collection service (KCS) was introduced in Nottingham in 2002, separating at source recyclable materials including paper, cardboard, cans, mixed plastics, mixed glass, as well as garden waste. Advance booking is required for bulky waste collection. One Civic Amenity (CA) site (also known as a Household Waste Recycling Center) and dozens of bring sites (also known as Mini Recycling Centers) are also located across the city for the further collection of recyclables.

Orange recycling bags are provided to homes that cannot use bins, such as communal dwellings and flats.

Nottingham is the pioneer regarding EfW and waste minimization in England. With a capacity of 170,000 tons/year, the Eastcroft EfW was built in the early 1970s, and upgraded in 1998 to cogenerate combined heat and power (CHP) from waste. Recovered power and heat are supplied to National Grid and for heating city center buildings via a district heating scheme, respectively. Refuse-derived fuel (RDF) is also produced from a material recovery facility (MRF) for improved energy recovery. Nottingham City Council has also introduced ambitious MSW management targets for 2025: 1) to reduce household waste generation to 390 kg per person, 2) to recycle 55% of household waste; and for 2030: 1) to reduce the residual household waste generation to less than 200 kg per person, 2) to achieve "zero waste to landfill" (NCC, 2010). Waste prevention measures have been introduced to reduce waste generation. Per capita MSW generation had been reduced from 463 kg in 2001/02 to 361 kg in 2016 (Fig. S1), which was much lower than the average value in England (412 kg) and the EU (487 kg) in that year (Eurostat, 2017, DEFRA, 2018). The reduction target for 2025 seems has been achieved in advance.

2.2. Goal and scope

The goal of this study was to quantify and compare the GWP of three historical MSW management strategies at three development stages in Nottingham, and a future scenario in response to the EU directives. MSW is defined as the solid waste arising from household sources, for consistency with targets set in waste regulations and available data. The functional unit is defined as the treatment of one ton of MSW, to ensure that the presented scenarios are comparable to each other. To assess the influence and importance of waste prevention on establishing sustainable MSW management, GHG emissions from managing MSW generated by each person were also quantified.

2.3. System boundary

The spatial boundary of the MSW management system is the administrative boundary of Nottingham City Council. The overall system addressed in the present study is illustrated in Fig. 2. It contains all waste management processes including the collection, transport, treatment and disposal of waste. All possible future emissions were accounted for the year when the MSW was managed. This is necessary to ensure that the calculations for all MSW management scenarios comparable. The major sources of emissions were determined as follows:

- Fuel and power used in MSW management processes, but excluding emissions from upstream activities such as mining and transport. Due to the evolution of energy mix, the emission factors of electricity production were estimated to be 0.45kg CO_2 eq./kWh in 2002, 0.47 CO_2 eq./kWh in 2007 and 0.35 CO_2 eq./kWh in 2017.
- Waste collection.
- Transport to/between treatment facilities.
- Direct emissions from waste; for example, CO_2 from waste incineration.
- Avoided GHG emissions due to materials recycling and energy recovery.
- Environmental burdens from the operation of the CA and bring sites were excluded due to data deficiency.

2.4. Scenarios

In total, four MSW management scenarios including three historical scenarios and a future scenario have been developed and assessed in this study (Fig. 3). The statistical year in the UK is the period from April to the following March; for example, April 2016 – March 2017, so that the years to our MSW management scenarios are expressed to cross two years, i.e. 2001/02.

The selection of scenarios was based on the enforcement time of EU waste directives and data availability. The scenarios are discussed in detail in the following sub-sections.

2.4.1. Description of Scenario S1: 2001/02

This scenario relates to MSW management as at 2001/02, when the EU Landfill Directive began to be enforced in Nottingham, and is the earliest year for which complete data is available. In this scenario, weekly house-to-house collection without separation was provided by the local authority (Parfitt et al., 2001). A transfer station was used to store and transfer waste to landfill. MSW was disposed in landfills (54.7%) and incinerated at the Easrcroft EfW facility (40.7%) (NCC, 2005). Under these circumstances, the compositions of incinerated and landfilled MSW were assumed to be the same (Table 1 and 2). 3.4% and 1.2% of MSW were recycled and composted (NCC, 2005). Materials were recycled at the CA site and bring sites. Recycled materials were assumed to be paper, glass and metal (estimated at 50%, 25% and 25% of recycled materials, respectively) (Data.Gov, 2018). Garden waste was composted via open windrow composting. Pretreatment before incineration/landfill and methane collection systems at the landfill were unavailable. Bottom ash from incineration (BAI) was landfilled.

2.4.2. Description of Scenario S2: 2006/07

S2 corresponds to the year 2006/07, before the enforcement of the EU Waste Framework Directive. It is the earliest year of documented waste flows. In this scenario, new waste management initiatives, such as the KCS, bespoke bulky waste collection and MRF, had been introduced but were not fully implemented (Fig. 2). A transfer station was still used, but now to store and transfer waste to MRF. Landfilling rate was reduced to 32.7% because of the improved recycling (17.5%) and composting (8.6%) rates. 41.2% of waste sent for EfW. Metal from BAI was recycled. The compositions of MSW and incinerated waste are illustrated in Table 1 and 2.

2.4.3. Description of Scenario S3: 2016/17

S3 corresponds to the year of 2016/17 and represents the most recent full year for which data was available for our analysis (Fig. 2). KCS was further strengthened to serve all households in Nottingham, which led to increased recycling and composting rates of 31.5 % and 12.9%, respectively. Production of RDF was also introduced. BAI was recycled for aggregates. Landfill became the least favorable waste disposal method with 7.3% of MSW landfilled. 57.6% of MSW was incinerated for energy recovery.

2.4.4. Description of Scenario S4: Future scenario

Based on our experience in analysing historical MSW management scenarios, an alternative future scenario is proposed, to further improve the material and energy recovery capability of the MSW management system in Nottingham. This scenario was constructed based on the same quantity and quality of waste in 2016/17. Food waste is separately collected. Anaerobic digestion (AD) replaces open windrow composting for treating food and garden waste. Biogas from AD is utilized for power and heat generation. Regularly collected residual waste is pre-treated in the residual MRF for material recycling before incineration.

2.5. Life cycle inventories

2.5.1. Collection, transfer and transport

Detailed estimations of the travel distance and LCI for MSW collection and transport are presented in Appendix Section S1. Electricity and diesel consumption due to the transfer station was assumed to be 4 kWh/t and 0.84 kg/t, respectively (Turner et al., 2016).

2.5.2. Landfill

1.8 kg/t diesel and 8 kWh/t electricity were assumed to be consumed for operating landfill (Turner et al., 2016). The amount of methane emitted from landfill can be estimated based on equations reported by Fong et al. (2015) (Presented in SI Section S2). This method calculates the total mass of methane potentially generated based on the mass and composition of landfilled waste as listed in Table 1.

2.5.3. Incineration with energy recovery

The flue gas emitted from the incinerator fed by MSW after treatment mainly contains CO_2, but also some trace gases including CO, SO_2, NOx and N_2O, etc. Given that CO_2 capture is not in place in most waste incineration plants worldwide, the quantity of CO_2 emitted from the incinerator could be calculated based on the mass and composition of the incinerated waste (Table 2) using equations provided by the IPCC (2006) (Presented in SI Section S2). Air pollution control equipment, such as selective noncatalytic reduction (SNCR) for the reduction of nitrogen oxides, was installed by Eastcroft EfW to control the emission of air pollutants (FCC Environment, 2015). After treatment, the concentrations of methane and NO_x emitted from the incinerator was under the emission limit values set by the EU (EC, 2000, WRG, 2008, FCC Environment, 2015). Thus, the GWP of methane and NO_x emitted from MSW combustion were ignored.

Eastcroft EfW could harness 89% of the LHV of MSW to produce steam (FCC Environment, 2015). This steam is sent to an energy generation facility for electricity and hot water production with conversion efficiencies of 17.2% and 31.7%, respectively (FCC Environment, 2015). 62 kWh/t of recovered electricity and 3.76 kg/t fuel oil were consumed in operating the incineration plant (WRG, 2008). The LHV of incinerated waste was estimated through physical composition based empirical model (Eq. 1), developed by the authors using 151 datasets collected from 47 cities in 12 countries.

$$LHV\ (kJ/kg) = -72.42Pr + 83.20Pa + 67.90Pl + 7669.08 \quad (1)$$

Where *Pr* is the percentage of putrescible including food waste and garden waste, *Pa* is the percentage of paper; and *Pl* is the percentage of plastics. The value of percentage is within the range between 0 and 100.

Recovered heat from waste was assumed to substitute the equivalent heat generated from gas boilers, as these dominate home heating in England, due to insufficient district heating networks (Euroheat & Power, 2017, DECC, 2013). The majority of boilers available on the British market have efficiencies in the range of 88 % and 89.7 % (Knight, 2018). Hence, 89 % was uded in this study. The LHV of natural gas is 47.82 MJ/kg with a GHG emission factor of 2.72 kg CO_2-eq./kg (DEFRA, 2016). Based on these assumptions, the quantity of natural gas and associated GHG emission saved by EfW were quantified.

2.5.4. Recycling

Avoided emissions by material recycling were modeled based on the England Carbon Metric Report (DEFRA, 2012).

2.5.5. Composting

GHG emissions from composting were calculated after excluded the 36% non-compostable fraction (NCC, 2013). Details of LCI for composting are presented in Table 3. The produced compost was used to substitute inorganic N, P and K fertilizers. Hill et al. (2011) reported that GHG emission from production 1 kg of inorganic N, P and K fertilizer were 6.8 kg CO_2-eq., 1.2 kg CO_2-eq. and 0.5 kg CO_2-eq. respectively.

2.5.6. Material recovery facility

There are two types of MRF. One is designed to process comingled collected recyclables for the recovery of paper, glass, plastics and cans. Diesel and electricity consumption in this MRF are 2 kg/t and 35 kWh/t, respectively (Turner et al., 2016). The other is Residual MRF, which is designed to recover materials from bulky waste, street waste and residual waste from a CA site. Diesel and electricity consumption in a Residual MRF are 44 kWh/t and 2 kg/t, respectively (Pressley et al., 2015, Turner et al., 2016).

2.5.7. Production and incineration of RDF with energy recovery

Burnley et al. (2011) recommended that electricity consumption in a facility with a yield of RDF in the range of 14 – 22% was 40 kWh/t. The RDF yields in both types of MRF in Nottingham were around 20%. RDF was assumed to be incinerated in a power plant to generate electricity only. The efficiency of a dedicated RDF incineration plant was assumed to be higher than the EfW plant; at 25% on an LHV basis (Burnley et al., 2011). The LHV of standard UK MSW derived RDF is 25 MJ/kg with a fossil carbon content of 32% by weight (Materazzi et al., 2015, IPCC, 2006). Emissions from RDF combustion could thus be calculated based on the equations provided by IPCC (2006) (Presented in SI Section S2) .

2.5.8. AD

Biogas production with a yield of 20% by weight of which 63% is methane in an AD process, was assumed (Zaccariello et al., 2015, Turner et al., 2016). The LHV of biogas is 30 MJ/kg (DEFRA, 2016). Biogas is used for electricity and heat production on site using the CHP engine. Energy recovery efficiencies of 31% and 49% for electricity and heat were assumed (Turner et al., 2016). A detailed LCI for the AD process is presented in Table 4.

2.6. Impact assessment

The life cycle impact assessment was characterized by GWP at a 100 year period (GWP_{100}) based on the results of the inventories using the IPCC 2013 GWP 100a method (IPCC, 2013). This method provides a comprehensive methodology to calculate GWP_{100}, associated with amount of GHG emission and its equivalency factor. The total GWP of the MSW management is the sum of GWPs of all GHGs. The GHGs of interest in MSW management include carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). These GHGs account over 90% of total GHG emissions from MSW management (Bogner et al., 2007). According to IPCC guidelines on GHG inventories, only CO_2 from fossil origins is regarded to have a GWP (IPCC, 2006).

2.7. Interpretation

Interpretation relates to the presentation of results and associated sensitivity analysis. LCA results were presented in two ways: the GWP_{100} of managing 1 ton of MSW (expressed as GWP_{100} per ton of MSW), and the GWP_{100} of managing MSW generated by each citizen (expressed as GWP_{100} per capita). Sensitivity analysis is a crucial step in assessing the reliability and robustness of LCA results, by understand how they are affected by changes in certain parameters, such as waste composition and the adopted calculation models. In this study, two sensitivity analyses were carried out. Sensitivity analysis 1 was carried out by varying the DOC in landfilled waste, the content of N, P, K in composted organic waste (Table S9), and the LHV and fossil carbon of RDF. Sensitivity analysis 2 was carried out by using another LHV predictive model to estimate the LHV of incinerated MSW.

3. Results and discussions

3.1. Historical GWP_{100} of MSW management

3.1.1. GWP_{100} per ton of MSW

The LCA results are presented in Fig. 4 – 5 and Table 5. Fig. 4a clearly illustrates that the GWP_{100} of MSW management has significantly decreased from 1076.0 kg CO_2-eq./t of MSW in 2001/02 to 211.3 kg CO_2-eq./t of MSW in 2016/17. This is mainly due to the diversion of waste from landfill to more sustainable management options such as recycling, composting and incineration. S1 has the highest GWP_{100} amongst all historical scenarios, because over half of

MSW was landfilled without any methane recovery, which made landfill the major emitter of GHG, accounting for 82.5% of the total GWP_{100} in S1.

In S2, the GWP_{100} reduced to 487.9 kg CO_2-eq./t of MSW, less than 50% of that of S1. A further reduction to half of that in S2 was achieved in S3 (Fig. 4a). This was because more materials such as paper, plastics, glass and metal were recycled, more garden waste was composted and RDF was produced. The fully implemented KCS improved the separate delivery rate, so as to enhance the quantity and quality of recycled materials. Recycled materials compensate the equivalent GWP_{100} from the consumption of virgin materials and fossil fuels.

Materials recycling was the only waste management practice that consistently resulted in GWP_{100} savings in all historical scenarios. A significant reducing trend of GWP_{100} achieved by materials recycling was observed from 2001/02 to 2006/07. This is mainly because the introduction of KCS and MRF greatly improved the material recycling rate. However, GWP_{100} contributed by materials recycling increased by 5.8 kg CO_2-eq./t of MSW in 2016/17 as compared to that in 2006/07. The reason is that producing products from secondary materials (recycled or recovered materials from waste) does not always cause less global warming impact than from virgin resources (Björklund and Finnveden, 2005). DEFRA (2012) reported that it produced more GHG to recycle food and beverage cartons than to produce it from virgin materials in the UK. Alternative treatment options should be considered to treat these materials, which could cause greater GWP to recycle it, or to improve the efficiency of recycling and reprocessing. As Fig. 5 depicted, GWP_{100} saved by recycling varies among materials. Recycling metals followed by recycling paper, saved the most GHG emission in all historical scenarios. The quantity of recycled paper was far more than for other recycled materials in both 2006/07 and 2016/17, but the GWP_{100} saved by recycling paper was less than metal recycling because chemical and fossil fuel consumption in paper recycling was greater (Habib et al., 2013), and the substituted CO_2 emission from steel manufacturing from virgin material was relatively higher (Rankin, 2012, Burchart-Korol, 2013, Laurijssen, 2013).

Composting of garden waste was a contributor of GWP_{100} in all historical MSW management scenarios, because open windrow composting was applied, through which GHGs were directly emitted to the ambient atmosphere and no energy was recovered. The detailed LCA result for the composting process indicates that the production of organic fertilizer avoided the utilization of inorganic fertilizers (N, P, K) and cut the overall GWP_{100}, but the GHG emission from decomposition and facility operation was more than the saved amount. The gross GWP_{100} of composting was 122.5 kg CO_2-eq./t of garden waste, while the saved GWP_{100} by inorganic fertilizer avoidance was only 20.4 kg CO_2-eq./t of garden waste.

GWP_{100} generated by EfW were 195.0 kg CO_2-eq./t, 272.9 kg CO_2-eq./t and 172.8 kg CO_2-eq./t of MSW in S1, S2 and S3, which accounted for 18.1%, 55.9% and 81.8% of GWP_{100} in these scenarios, respectively. The energy recovery efficiency in Nottingham was 15.3% for electricity and 28.2% for heat, which appeared to be lower than other cases reported in the literature. Reimann (2012) reported that average energy recovery efficiency in European EfW plants was 26.1% in the case of electricity production only, 77.2% in case of heat production only and 52.1% in case of CHP. Habib et al. (2013) reported that the gross energy recovery efficiency of EfW reached 28% for electricity and 85% for heat in Aalborg, Denmark, which made MSW management in that city a GHG saver. Therefore, upgrading the EfW facility to improve the energy recovery efficiency is recommended as a possible solution to improve the future environmental performance of the waste management system in Nottingham.

The quantity and share of GWP_{100} contributed by collection and transport were lower compared to other processes, but an obvious increasing trend has been observed during the period of study. As MSW management options were shifted to upper layers of the waste management hierarchy, the GWP_{100} generated by transport increased significantly from 4.7 kg CO_2-eq./t of MSW in 2001/02 to 44.2 kg CO_2-eq./t of MSW in 2016/17; whereas the GWP_{100} from collection stayed relatively stable with a gentle declining trend during the same period (Table 5). The reduction in GWP_{100} from collection is due to the amount of waste collected at bring sites and street cleaning was reduced due to the introduction of KCS. Generally, a relatively longer distance was traveled to collect recyclables from distributed bring sites and to clean streets than to collect waste through KCS. The GWP_{100} of transporting recycled materials to reprocessing facilities increased significantly (Table 5), due to two factors: more materials were recycled, and reprocessing facilities were usually located some distance from Nottingham. For example, recycled glass and paper was transported 173 km and to overseas for reprocessing, respectively. GWP_{100} of transporting recycled materials to reprocessing facilities in S3 was nearly 44 times and 9 times more than those in S1 and S2, respectively. The increased GWP_{100} by transport led to the increase of overall GWP_{100} from materials recycling. Similar result was observed by Turner et al. (2016) and they suggested that promoting domestic reprocessing of secondary materials was a possible solution to reduce the GWP_{100} from transport and eventually enhance the overall environmental benefits from materials recycling.

3.1.2. GWP_{100} per capita

Similarly, GWP_{100} per capita significantly reduced from 498.2 kg CO_2-eq. in 2001/02 to 76.3 kg CO_2-eq. in 2016/17, a nearly sevenfold reduction (Fig. 4b). This is due to the improvements in MSW management discussed in section 3.1.1, as well as efforts in waste prevention. MSW generation per capita decreased from 463 kg to 361 kg during the same period (Fig. S1). GWP_{100} added by collection and transport increased significantly from 0.4 kg CO_2-eq./Ca in 2001/02 to 17.0 kg CO_2-eq./Ca in 2016/17 (Table 5), the reason for which has also been detailed in section 3.1.1.

3.2. GWP_{100} in the future scenario (S4)

MSW management in S4 becomes a net saver of GHG emissions, due to improvements in material recycling and waste treatment. Both GWP_{100} per ton of MSW and GWP_{100} per capita reduce to just –142.3 kg CO_2-eq. (Fig. 4a) and –40.2 kg CO_2-eq (Fig. 4b), respectively. AD reduces GWP_{100}, because of energy recovery from biogas. 81.3 kg CO_2-eq./t of MSW will be saved when garden waste and food waste are treated by AD. Incineration will be another saver to reduce GWP_{100} by 0.2 kg CO_2-eq./t of MSW and 0.1 kg CO_2-eq./Ca. GWP_{100} saved by materials recycling will be further improved to 257.5 kg CO_2-eq./t of MSW because more materials are recycled from residual waste. However, EfW and combustion of RDF will consistently be GHG emitters, if no more advanced technology is applied to improve the EfW's energy recovery efficiency. GWP_{100} from transport in S4 will increase, since more materials are transported for recycling (Table 5).

In addition to improving the recycling/composting rate and upgrading the biological treatment technology to reduce GWP from MSW management, attention should also be paid to the quality of secondary products from recycled materials and compost. Accumulation of hazardous substances in recycled materials reduces the quality of products made up of secondary materials and increases the release potential of hazardous substances (Kral et al., 2013). An apparent example is found in the steel industry where copper contaminates the steel cycle (Kral et al., 2013). The accumulation of copper hardens steel and decreases steel quality (Haupt et al., 2017). Recycling material from mixed residual waste could improve the recycling rate, but also introduce contaminates to recycled materials, and this will reduce the quality of secondary products made from them. Production of RDF might be an alternative option. The suitability of compost from bio-treatment as fertilizer is influenced by the quality of feedstock (proteins, minerals, and presence of undesirable materials) which depends mainly on the source separation (Kumar and Samadder, 2017). Thus, enhancing source separation and public participation will be crucial to improve the quality of secondary products.

3.3. *Sensitivity analysis*

As presented in Table 6 and Fig.6, sensitivity analysis results indicate that the variations in waste composition and the LHV prediction model affect the estimated GWP_{100} values, but not the downwards trend.

The DOC (Table 1), N, P and K (Table S9) contents in organic waste varied within a range due to the diversified compositions within this category (Boldrin et al., 2009). Furthermore, the LHV and fossil carbon of RDF in the UK vary in the ranges 13 – 25 MJ/kg and 21.7 – 32.0 %, respectively, depending on its composition (Burnley et al., 2011, Materazzi et al., 2015). All these variations in waste composition affect the total GWP_{100} of MSW management. Table 6 illustrates the minimum and maximum GHG emission from managing 1 ton of MSW when the variations in waste composition are taken into consideration.

To assess the sensitivity of LCA results affected by the LHV predicting model, the model developed by Khan and Abu-Ghararah (1991) (Eq. 2), using global data collected and the same explanatory variables as Eq. 1, was used to predict LHV of incinerated waste in S1, S3 and S4 (the LHV of incinerated waste in S2 was measured using a bomb calorimeter). As Fig. 6 illustrated, both the LHVs and associated GWP_{100} of incinerated waste in all three scenarios change significantly when using Eq. 2. However, this model was developed 30 years ago, and so may not be suitable for estimating the LHV of modern waste, because the characteristics of MSW have changed dramatically during this period. Therefore, the updated model (Eq. 1) is recommended to estimate the LHV of MSW. Nevertheless, the GWP_{100} of MSW management in Nottingham is estimated to have reduced during the study period, irrespective of the model adopted.

$$LHV\ (kJ/kg) = 53.5\ (F + 3.6\ Pa) + 372.16\ Pl \quad (2)$$

4. Conclusions

To assess the effectiveness of waste regulations and the evolution of MSW management under the guidance of these regulations, in this study, LCA was carried out to estimate and compare the GWP_{100} of three historical MSW management scenarios in Nottingham, since the enforcement of the EU Landfill Directive. A further future scenario designed to meet the local 2025 recycling target and 2030 landfill target was also evaluated and compared with the historical scenarios. The results indicate that both GWP_{100} per ton of MSW and GWP_{100} per capita in Nottingham have reduced significantly during the last 16 years. Waste regulations effectively incentivised the shifting of MSW management from a landfill centered mode to a more environmentally friendly management approach. The results also indicate the importance of waste prevention in mitigating the GWP of MSW management. In future works, other environmental impacts in addition to GWP and sustainability at social and economic dimensions of MSW management can be assessed to comprehensively assess the effectiveness of waste regulations.

MSW management system in Nottingham is still a net emitter of GHGs, partly because of the low energy recovery efficiency in EfW facility and increased emissions due to the transport of materials for recycling. Thus, improving the energy recovery efficiency in EfW by upgrading its technology and promoting domestic reprocessing of secondary materials are recommended to mitigate GHG emission from MSW management. The LCA results of the future-looking scenario indicate that separating food waste at source and treating it via AD, pretreating residual waste before incineration and replacing open windrow composting by AD could turn the MSW management system into a net saver of GWP_{100}. To achieve the future-looking scenario, public participation also need to be enhanced to ensure the source separation. Besides, attention should be paid to the quality of recycled and recovered materials.

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Table 1. Composition of MSW and the landfilled waste (%)

Composition category	MSW			Landfilled waste			Degradable organic carbon (DOC) content in wet waste [c]
	2001/02 [a]	2006/07	2016/17	2001/02	2006/07 [b]	2016/17 [b]	
Paper & card	32.0	22.7	14.4	32.0	21.1	19.3	36 – 45 (40)
Putrescible [d]	21.0	33.7	36.2	21.0	37.6	2.3	8 – 20 (15)
Plastics	11.0	10.0	8.6	11.0	3.0	2.4	0
Glass	9.0	6.6	5.5	9.0	1.5	10.6	0
Metals	8.0	4.3	3.7	8.0	3.8	1.5	0
Wood	-	3.7	2.7	-	11.5	29.6	39 – 46 (43)
Textiles	2.0	2.8	5.8	2.0	4.5	1.1	20 – 40 (24)
Other	17.0	16.2	23.1	17.0	17.0	33.2	0 – 54 (0)
Total	100	100	100	100	100	100	-

a: (Burnley, 2001); b: Waste composition was estimated based on material flow analysis (Fig. S2-S4). c: sourced from IPCC (2006). d: Putrescible includes garden waste and food waste. Values in brackets () are the default values set by IPCC (2006).

Table 2. Composition of waste incinerated at Eastcroft EfW.

	2001 [a]	2006 [b]	2016 [c]	Futuristic scenario [d]	Dry matter content of wet weight [e]	Total carbon content in dry weight [e]	Fossil carbon fraction of total carbon [e]
Paper and card	32.0	20.8	10.2	2.9	90	46	1
Putrescible	21.0	25.8	34.9	12.0	40	38	-
Textiles	2.0	3.3	9.0	5.1	80	50	20
Fines (< 10mm)	7.0	3.4	0.4	1.4	90	3	100
Miscellaneous combustibles	8.0	10.9	19.2	51.7	40	70	10
Miscellaneous non-combustibles	2.0	3.2	4.7	0.5	100	-	-
Ferrous metal	6.0	3.3	2.6	2.4	100	-	-
Non-ferrous metal	2.0	1.3	0.9	2.9	100	-	-
Glass	9.0	9.4	3.2	3.8	100	-	-
Dense plastics	6.0	8.0	7.2	2.8	100	75	100
Plastics film	5.0	8.1	4.0	2.7	100	75	100
Others	0	2.7	3.7	12.4	-	-	-
Lower heating value (LHV) (MJ/kg)	9.6 [f]	8.8	6.8 [f]	7.4 [f]	-	-	-

a: Burnley (2001). b: WRL (2008). c: NCC (2013). d: Waste composition was calculated based on material flow analysis (Fig. S2-S4). e: IPCC (2006). f: LHV was calculated using the regression model built by authors based on waste composition, which would be explained in section 2.4.5.

Table 3. LCI for composting.

	Unit	Value	Reference
Pre-treatment input			
Diesel	kg/t	0.1	(Turner et al., 2016)
Electricity	kWh/t	1.1	(Turner et al., 2016)
Composting input			
Diesel	kg/t	3.07	(Fisher, 2006)
Electricity	kWh/t	0.51	(Fisher, 2006)
Process emission			
CH_4	kg/t	4	(IPCC, 2006)
N_2O	kg/t	0.24	(IPCC, 2006)
Avoided fertilizer product			
N fertilizer	kg/t	3.4	(Boldrin et al., 2009)
P fertilizer	kg/t	2.8	(Boldrin et al., 2009)
K fertilizer	kg/t	9.7	(Boldrin et al., 2009)

Table 4. Life cycle inventory data for the AD process.

	Unit	Value	Reference
Pre-treatment input			
Diesel	kg/t	0.1	(Turner et al., 2016)
Electricity	kWh/t	1.1	(Turner et al., 2016)
Process input			
Diesel	kg/t	1.3	(Fisher, 2006)
Electricity	kWh/t	20.6	(Fisher, 2006)
Process parameters			
Biogas yield rate	% by weight	20	(Zaccariello et al., 2015)
LHV	MJ/kg	30	(DEFRA, 2016)
CH_4 content of biogas	% biogas	63	(Turner et al., 2016)
Emission from incomplete combustion			
CH_4	mg /MJ $_{biogas}$	434	(Nielsen et al., 2010)
N_2O	mg /MJ $_{biogas}$	1.6	(Nielsen et al., 2010)
Process emission			
CH_4	kg/t	0.0213	(Fisher, 2006)
N_2O	kg/t	0.0115	(Fisher, 2006)
Avoided fertilizer product			
N fertilizer	kg/t	3.4	(Boldrin et al., 2009)
P fertilizer	kg/t	2.8	(Boldrin et al., 2009)
K fertilizer	kg/t	9.7	(Boldrin et al., 2009)

Table 5. GWP_{100} added by collection and transport (unit: kg CO_2-eq.)

		S1	S2	S3	S4
Per tonne of MSW	Collection	3.4	3.1	2.8	2.8
	Transport to reprocessor	1.1	4.7	42.2	44.9
	Transport between facilities	3.5	2.5	2.0	2.8
	Total	8.1	10.2	47.1	50.5
Per capita	Collection	0.2	1.4	1.0	1.0
	Transport to reprocessor	0.1	2.2	15.3	16.2
	Transport between facilities	0.2	1.1	0.7	1.0
	Total	0.4	4.8	17.0	18.2

Table 6. Effect of waste composition variation on GWP_{100} (unit: kg CO_2-eq./t MSW)

	S1		S2		S3		S4	
	Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.
Landfill	595.1	2868.5	235.1	831.8	80.2	312.1	0.3	0.3
Composting/AD	1.3	1.5	8.8	9.3	13.2	13.5	-81.4	-73.2
RDF	0.0	0.0	0.0	0.0	8.6	37.6	34.8	144.0
Total	787.6	3061.1	371.8	969.0	151.8	413.0	-250.9	-133.5

Fig.1. The location of Nottingham in Nottinghamshire and the UK, and Lower Layer Super Output Areas (LSOA) within Nottingham.

Fig. 2. The overall scheme of MSW management system analyzed in the present study.

Fig. 3. Schematic illustration of MSW management in all scenarios assessed in the current study. Newly introduced processes and changed waste flows are identified by different colors. BAI represents bottom ash from the incineration plant.

Fig. 4. The GWP100 of MSW management scenarios in Nottingham. (a): GWP100 per ton of MSW. (b): GWP100 per capita.

Fig. 5. The fraction of GWP100 saved by recycling different materials.

Fig. 6. Comparison between estimated LHVs (a) and GWP100 (b) of incinerated waste when different models were used to estimate its LHV.

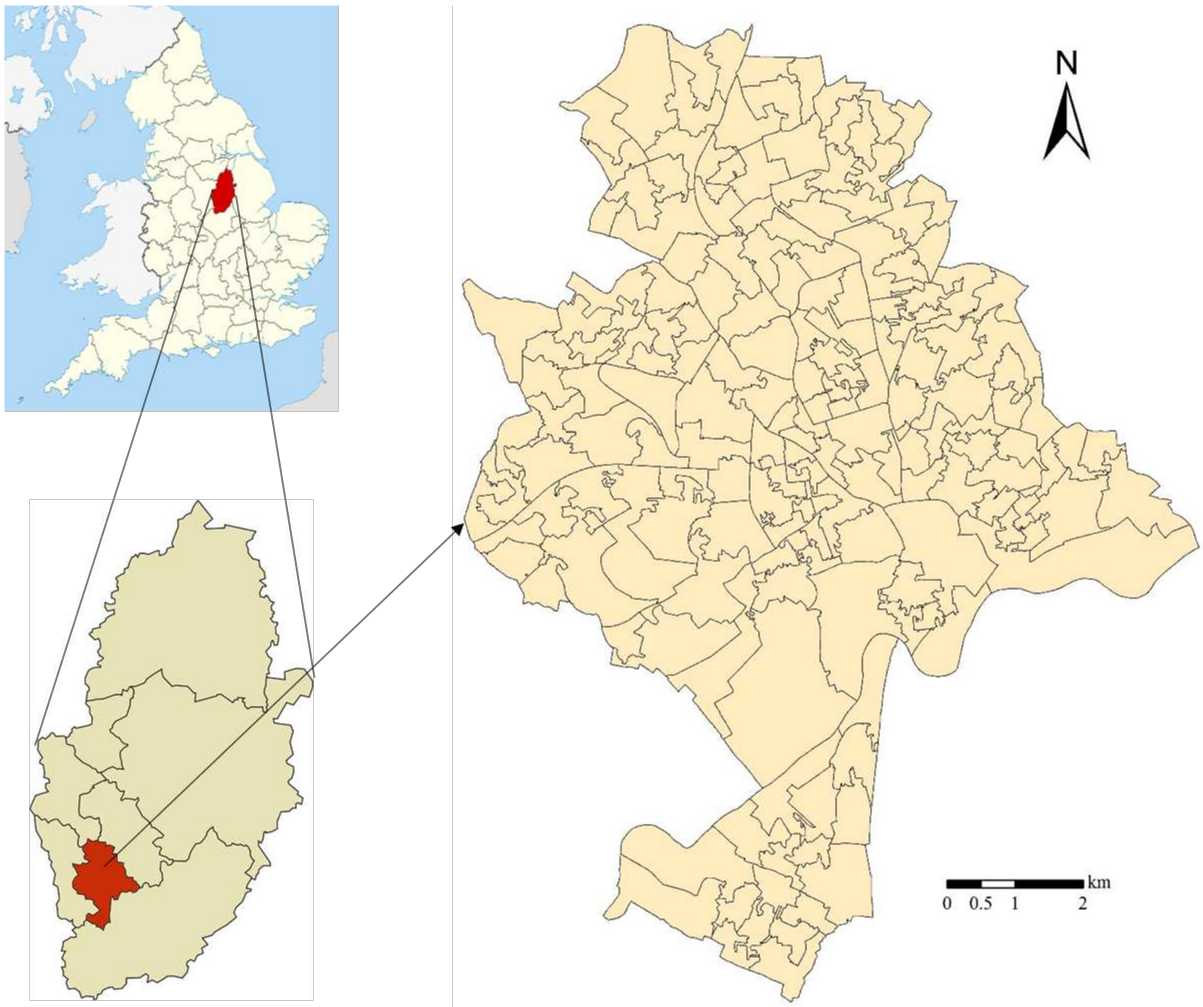


Fig.1.

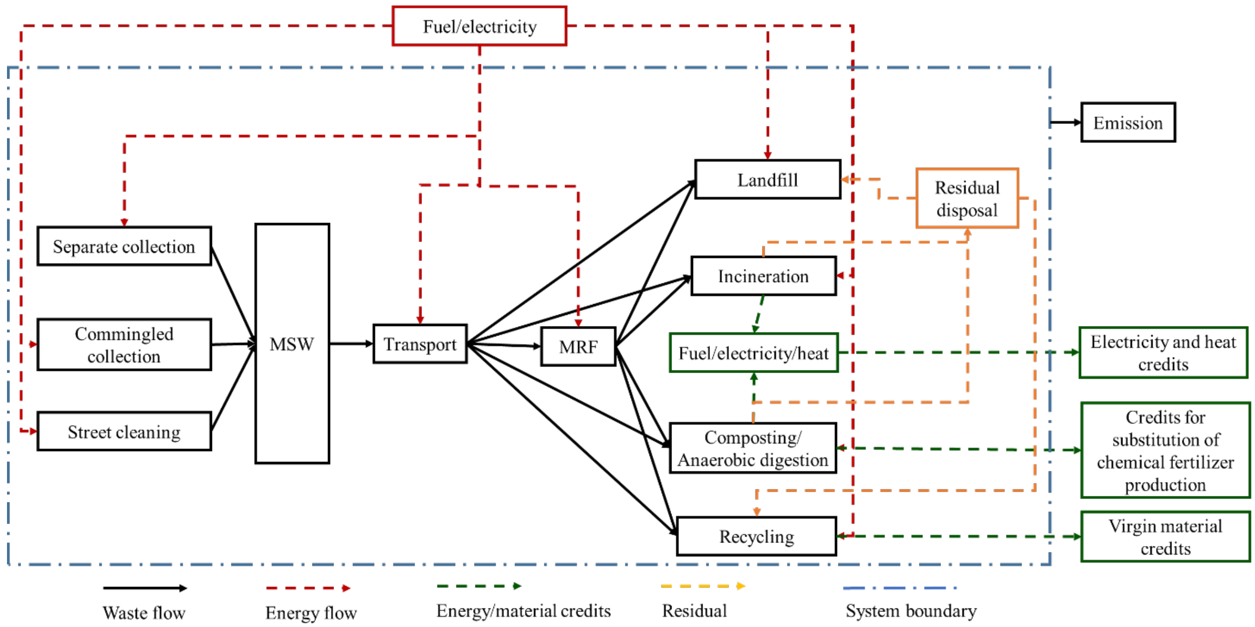

Fuel/electricity
Emission
Landfill
Residual disposal
Separate collection
Incineration
MSW
Transport
MRF
Commingled collection
Fuel/electricity/heat
Electricity and heat credits
Street cleaning
Composting/ Anaerobic digestion
Credits for substitution of chemical fertilizer production
Recycling
Virgin material credits
Waste flow
Energy flow
Energy/material credits
Residual
System boundary

Fig. 2.

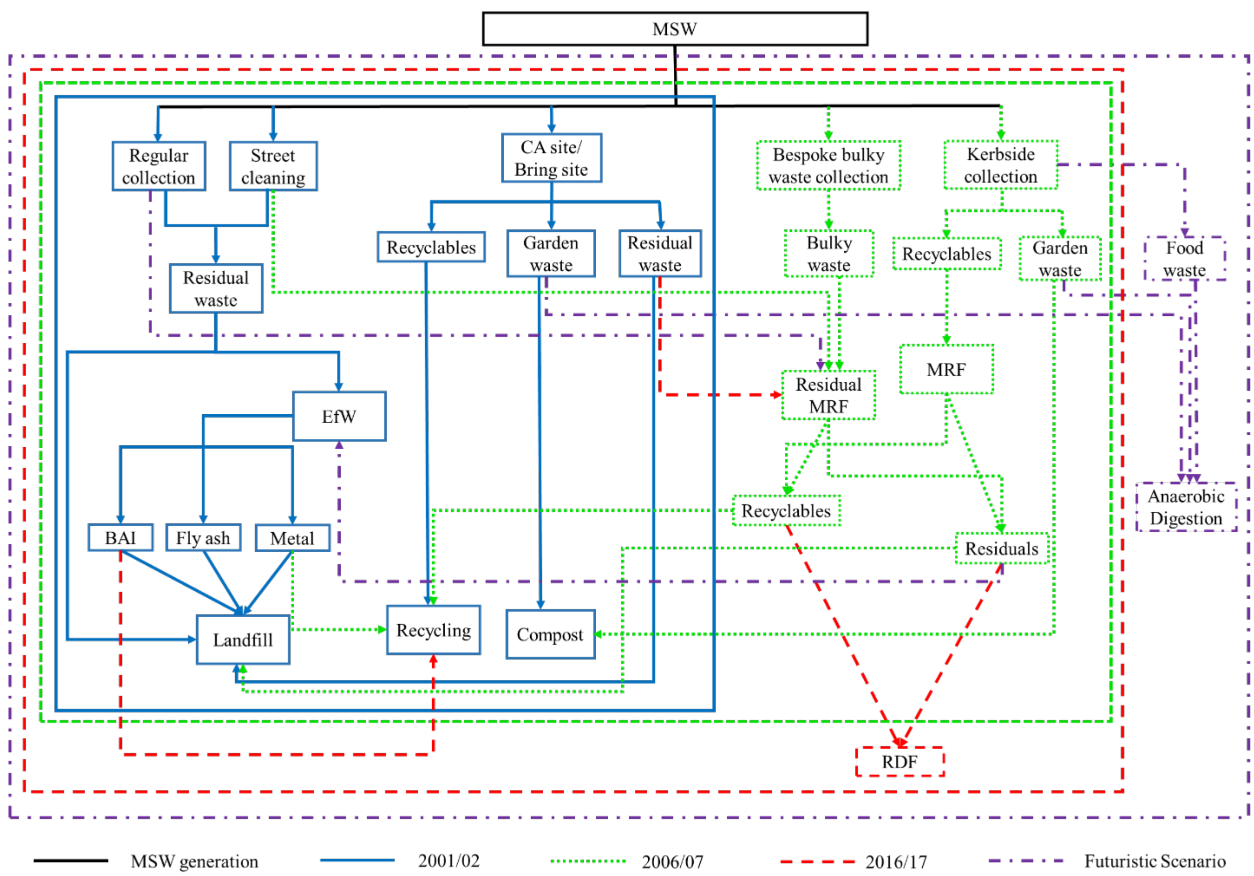

MSW
Regular collection
Street cleaning
CA site/ Bring site
Bespoke bulky waste collection
Kerbside collection
Recyclables
Garden waste
Residual waste
Bulky waste
Recyclables
Garden waste
Food waste
Residual waste
Residual MRF
MRF
EfW
Recyclables
Anaerobic Digestion
BAI
Fly ash
Metal
Residuals
Recycling
Compost
Landfill
RDF
MSW generation
2001/02
2006/07
2016/17
Futuristic Scenario

Fig. 3.

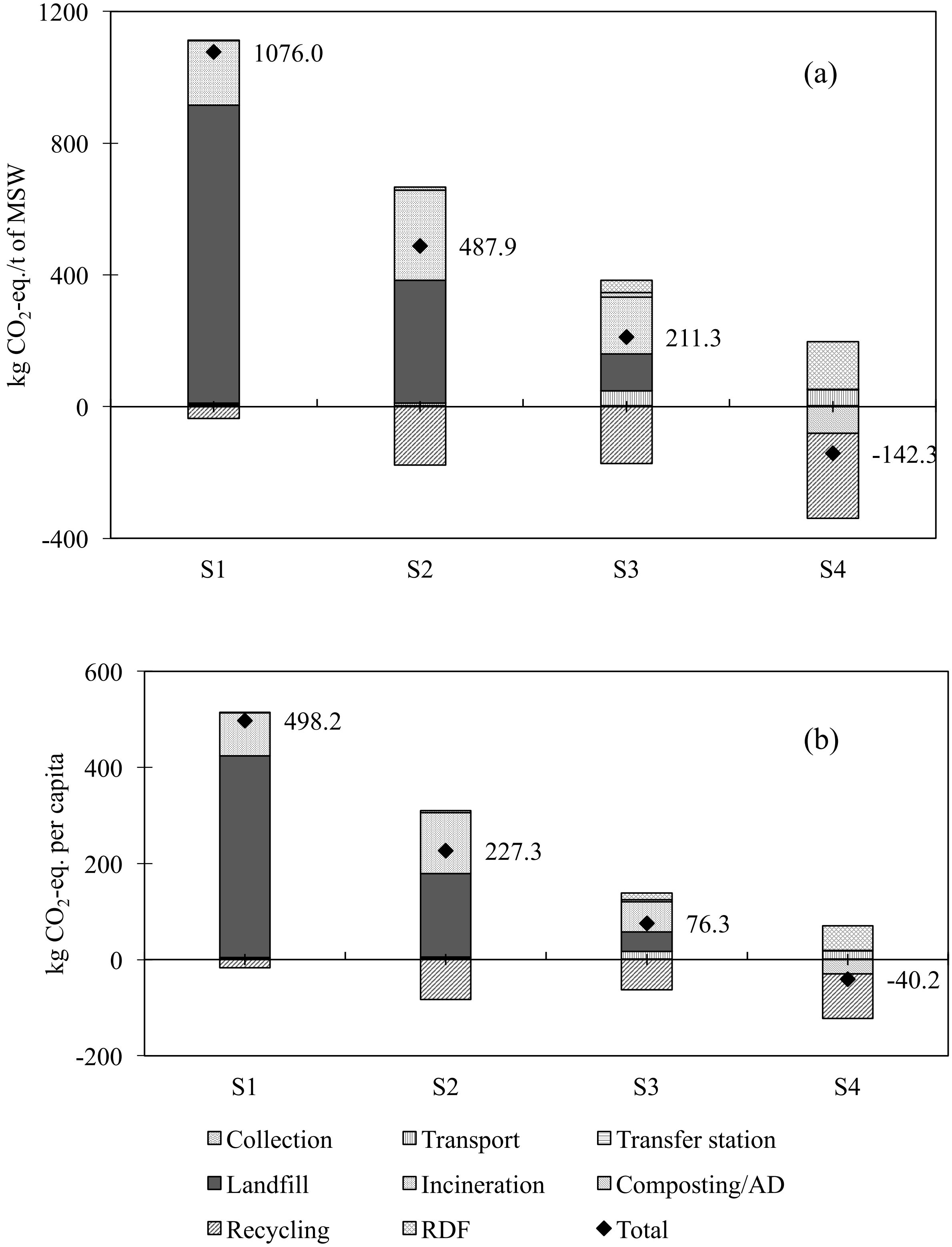
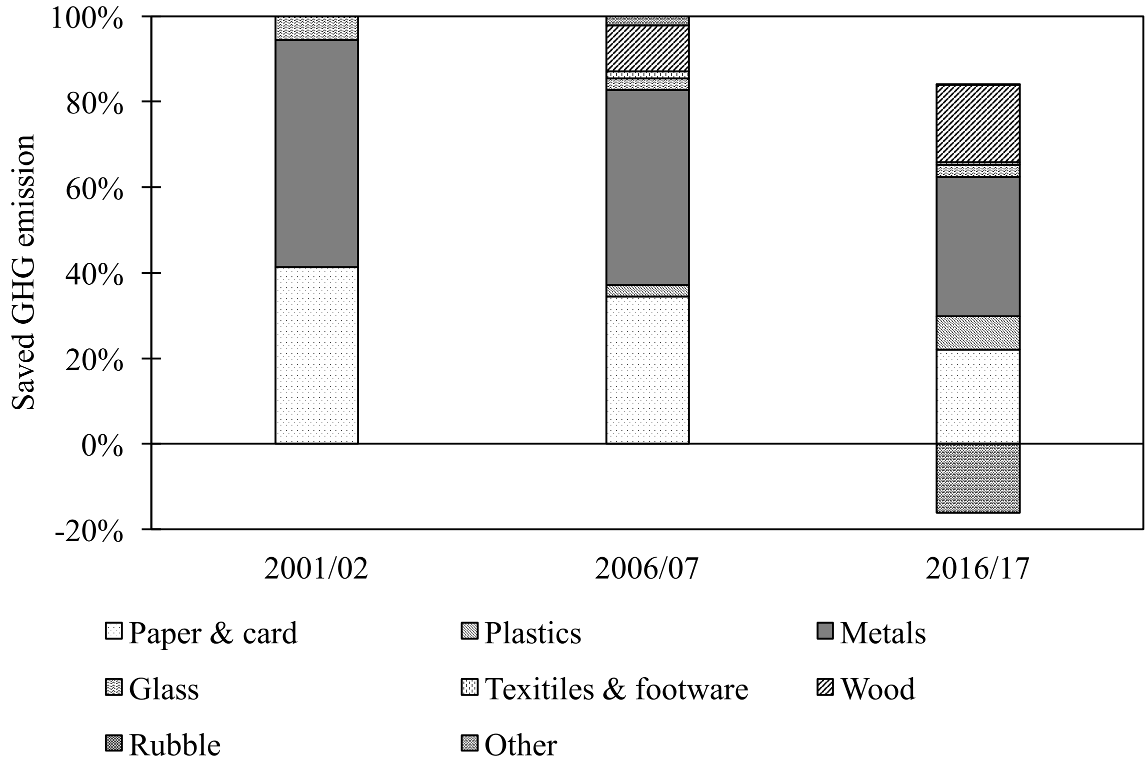

1200
800
400
0
-400
kg CO_2-eq./t of MSW
1076.0
487.9
211.3
-142.3
(a)
S1
S2
S3
S4
600
400
200
0
-200
kg CO_2-eq. per capita
498.2
227.3
76.3
-40.2
(b)
S1
S2
S3
S4
Collection
Transport
Transfer station
Landfill
Incineration
Composting/AD
Recycling
RDF
Total

Fig. 4.



100%
80%
60%
40%
20%
0%
-20%
Saved GHG emission
2001/02
2006/07
2016/17
Paper & card
Plastics
Metals
Glass
Texitiles & footware
Wood
Rubble
Other

Fig. 5.

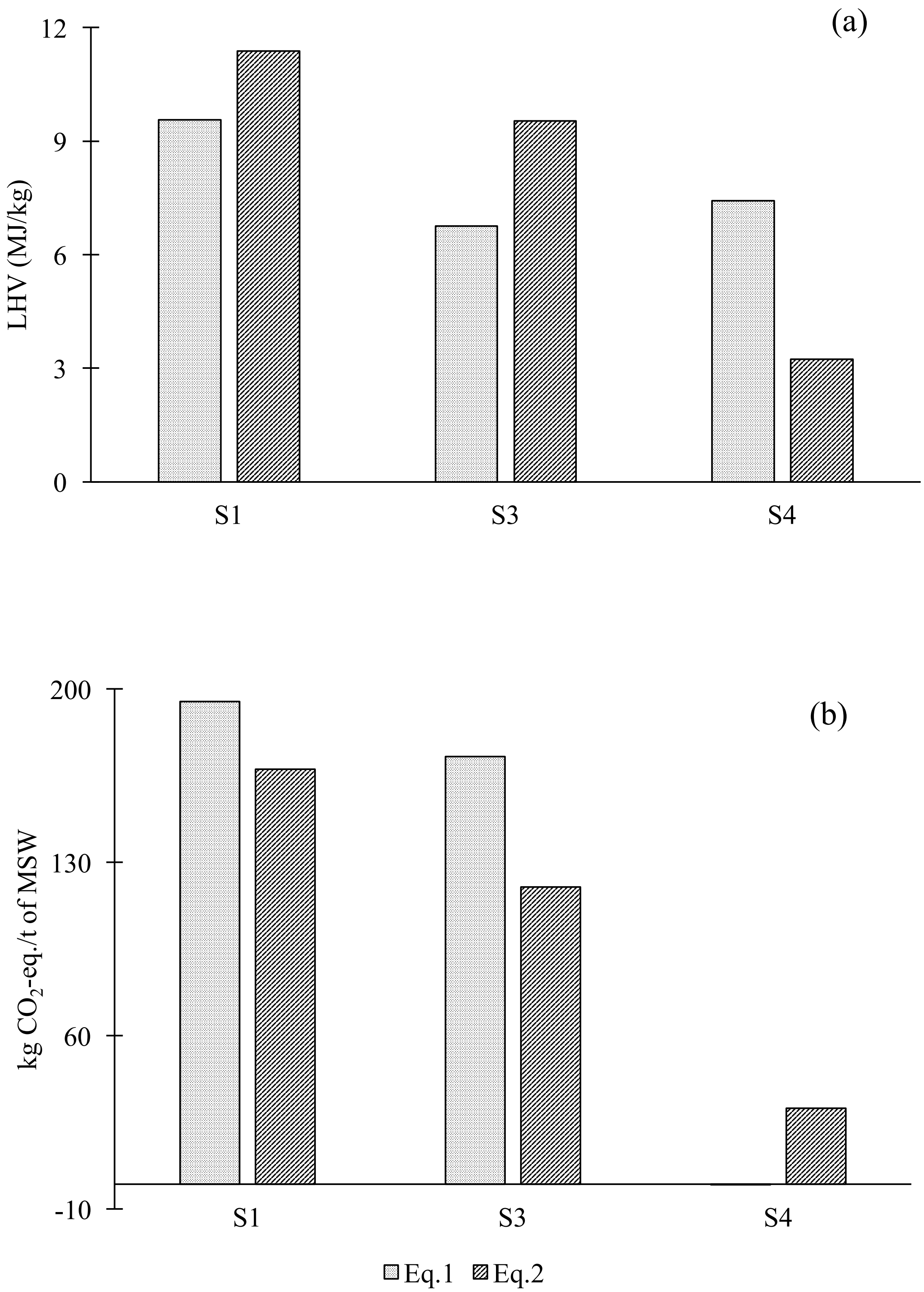

(a)
LHV (MJ/kg)
12
9
6
3
0
S1
S3
S4
(b)
kg CO2-eq./t of MSW
200
130
60
-10
S1
S3
S4
Eq.1
Eq.2

Fig. 6.