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1 **Life cycle assessment of municipal solid waste management in Nottingham, England: Past**

2 **and future perspectives**

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14 **Abstract**

15 Since the enforcement of the EU Landfill Directive, EU waste directives were successively

16 enforced in EU member states to facilitate the establishment of sustainable MSW management.

17 Various changes have been made in England to reduce the global impact of its MSW

18 management, but the effectiveness of these changes on mitigating the global warming potential

19 (GWP) from MSW management has never been investigated in detail. This study assessed the

20 historical GWP of MSW management in Nottingham throughout the period from April 2001

21 to March 2017 through life cycle assessment (LCA). The LCA results indicate continuous

22 reductions in greenhouse gas (GHG) emissions from MSW management during the study

23 period due to improvements in waste collection, treatment and material recycling, as well as

24 waste prevention. These improvements resulted in a net reduction of GHG emission from

25 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

26 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

27 (or -40.2 kg CO_2–eq./Ca) could be achieved by separating food waste from incinerated waste,

28 treating organic waste via anaerobic digestion and by pretreating incinerated waste in a material

29 recovery facility.

30 *Keywords*: EU waste directives; municipal solid waste; evolution; life cycle assessment; global

31 warming potential; Nottingham.

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37   **1.   Introduction**

38   Climate change is one of the most serious of current international concerns, to which

39   municipal solid waste (MSW) management is a significant contributor, through greenhouse

40   gases (GHG) emissions (Turner et al., 2016, Kaza et al., 2018), such as methane resulting from

41   the decomposition of biodegradable municipal waste (BMW) (El-Fadel et al., 1997). MSW and

42   landfills are the third largest anthropogenic source of global CH_4 emission (Das et al., 2019).

43   In 2016, the greenhouse gas (GHG) emissions from the waste management sector were 1.6

44   billion tons of CO_2-eq., accounting for 5% of global emissions (Kaza et al., 2018). To mitigate

45   the global warming potential (GWP) of MSW management, the EU Landfill Directive (EU

46   Directive 99/31/EC) was introduced in 1999 to reduce the quantity of BMW sent to landfill,

47   and setting a target of lowering the amount of landfilled BMW to 35% of that in 1995 by 2016

48   (EC, 1999). Subsequently, regulations have been successively introduced to divert waste from

49   landfill to more environmental friendly treatment options such as recycling, composting and

50   energy recovery, with corresponding management targets (Table S1). EU member states were

51   legally obligated to establish and enforce regional policy instruments to meet these targets.

52   Furthermore, the EU Waste Framework Directive (EU Directive 2008/98/EC) established the

53   "waste management hierarchy" to guide the practice of sustainable waste management. These

54   EU Directives have gradually promoted the establishment of sustainable MSW management,

55   which has the ability to harness resource from waste in the form of materials and energy (Liang

56   and Zhang, 2012, Cobo et al., 2018). To achieve the targets set in EU Directives, a variety of

57   strategies, technologies and techniques aiming at material recycling and energy recovery from

58   waste, as well as waste prevention, have been introduced in the last two decades in England,

59   but their realistic effects on the improvement of the performance of MSW management has not

60   to date been investigated.

61   A number of studies have been conducted to assess the evolution of MSW management,

62   and the pros and contras of the corresponding policies and strategies. Uyarra and Gee (2013)

63   investigated the transformation of waste management in Greater Manchester from a simple

64   landfill model to a complex, multi-technology waste solution based on intensive recycling and

65   composting, and sustainable energy usage. Pomberger et al. (2017) assessed the performance

66   of MSW management concerning the rate of landfilling, incineration, recycling and

67   composting, from 1995 to 2014 in Europe. Castillo-Giménez et al. (2019) assessed the

68   performance and convergence in the treatment of MSW by the EU-27 during the period 1995-

69   2016, by country and year. However, these studies focused on the final destinations of waste,

70   paying less attention to the environmental impacts of changing MSW management practices

71   from a life cycle perspective. This latter is of interest, since it has the potential to show for

72   example that landfill could be a desirable waste treatment option when landfill gas to energy is

73   considered (Khandelwal et al., 2019). Besides, waste prevention, which ranks at the top of the

74   waste management hierarchy, has seldom been considered as an indicator in evaluating the

75   performance of MSW management.

76   Life cycle assessment (LCA) has been extensively applied to evaluate environmental

77   burdens associated with MSW management (Fernández-Nava et al., 2014, Yay, 2015,

78   Milutinović et al., 2017, Coelho and Lange, 2018). But in addition to quantifying the

79   environmental impacts and burdens associated with waste management options, LCA can also

80   be used to explore opportunities for improvements (Cherubini et al., 2009). It also helps to

81   expand the perspective beyond the waste management system. This makes it possible to take

82   the significant environmental benefits that can be obtained through alternative waste

83   management options into account; for example, energy-from-waste (EfW) reduces the

84   consumption of energy from fossil fuels; recycled materials replace part of virgin materials;

85   and the compost from biological treatment substitutes the production of chemical fertilizers

86   (Franchetti and Kilaru, 2012, Jeswani et al., 2013, Turner et al., 2016). On the other hand, LCA

87   results can be affected by multiple factors such as the definition of system boundary, the

88   assumptions in life cycle inventory (LCI), and the methodologies or software adopted for

89   calculation (Yadav and Samadder, 2018, Zhou et al., 2018a, Khandelwal et al., 2019). There

90   are a number of impact assessment methods (e.g. CML, EDIP, IPCC 2013) and more than 50

91   LCA software (e.g. SimaPro, Gabi, WASTED) available to aid the performing of LCA (Yadav

92   and Samadder, 2018). Winkler and Bilitewski (2007) pointed out that the LCA results

93   calculated by different models showed high variation and not negligible, even led to

94   contradictory conclusions in some cases. Therefore, sensitivity analysis is often included in the

95   assessment to inform the robustness of the LCA results and the potential for improvement

96   (Khandelwal et al., 2019).

97   Most LCA studies have focused on the environmental impacts associated with the present

98   and possible future MSW management at specific sites, with less attention paid to the evolution

99   of an MSW management system in a historical context. Habib et al. (2013) assessed the GWP

100   of MSW management in Aalborg, Denmark from 1970 to 2010, with the focus on the effect of

101   EfW. Zhou et al. (2018b) evaluated the environmental performance evolution of MSW

102   management in Hangzhou, China, focusing on the treatment technologies and source-separated

103   collection. Evaluation of the environmental impacts over time reveals and documents the trend

104   in environmental impacts of a given waste management system for the study site, or whether

105   there has actually been progress towards a more environmentally friendly waste management

106   strategy (Poulsen and Hansen, 2009).

107   On the basis of the research gaps identified above, this study attempts to evaluate how the

108   implementation of new waste management options and regulations over time has affected the

109   GWP of MSW management at a selected city by quantifying the GHG emissions from MSW

110   management scenarios at different stages of development using LCA. Nottingham in Eastern

111   England was chosen as it has changed its MSW management strategy several times since the

112   implementation of the EU Landfill Directive, beginning with combined landfilling and

113   incineration with energy recovery and ending at present with a combination of source

114   separation, recycling, composting and incineration with energy recovery, and ambitious MSW

115   management targets have been set. The balance for GHG has been evaluated for three specific

116   years: 2001/02, 2006/07 and 2016/17, and a future scenario which would potentially reach the

117   2025 recycling target and 2030 landfill target set by Nottingham City Council (Section 2.1).

118   The results provide an insight into how the waste management policies and regulations drive

119   the improvement of waste management, and hence support local policy and decision making

120   by identifying the areas where the enforcement of policies, regulations, strategies and

121   technologies can be strengthened in the future development of MSW management, as reference

122   to other similar cities.

123   **2.   Methodology**

124   *2.1.   Study city*

125   Nottingham is one of the Core Cities in England, located in the central UK (52° 57' N and

126   1° 09' W) (Fig. 1). It covers an area of 7,5378 hectares and had an estimated population of

127   329,200 in 2017 (Nottingham Insight, 2018). Since the start of the new millennium, new waste

128   management strategies, measurements and technologies were adopted in Nottingham to divert

129   waste from landfill, as well as to prevent unnecessary waste generation. As a result, the

130   quantities of waste generated and landfilled were significantly reduced (Fig. S1). A kerbside

131   collection service (KCS) was introduced in Nottingham in 2002, separating at source recyclable

132   materials including paper, cardboard, cans, mixed plastics, mixed glass, as well as garden waste.

133   Advance booking is required for bulky waste collection. One Civic Amenity (CA) site (also

134   known as a Household Waste Recycling Center) and dozens of bring sites (also known as Mini

135   Recycling Centers) are also located across the city for the further collection of recyclables.

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

137   and flats.

138   Nottingham is the pioneer regarding EfW and waste minimization in England. With a

139   capacity of 170,000 tons/year, the Eastcroft EfW was built in the early 1970s, and upgraded in

140   1998 to cogenerate combined heat and power (CHP) from waste. Recovered power and heat

141   are supplied to National Grid and for heating city center buildings via a district heating scheme,

142   respectively. Refuse-derived fuel (RDF) is also produced from a material recovery facility

143   (MRF) for improved energy recovery. Nottingham City Council has also introduced ambitious

144   MSW management targets for 2025: 1) to reduce household waste generation to 390 kg per

145   person, 2) to recycle 55% of household waste; and for 2030: 1) to reduce the residual household

146   waste generation to less than 200 kg per person, 2) to achieve "zero waste to landfill" (NCC,

147   2010). Waste prevention measures have been introduced to reduce waste generation. Per capita

148   MSW generation had been reduced from 463 kg in 2001/02 to 361 kg in 2016 (Fig. S1), which

149   was much lower than the average value in England (412 kg) and the EU (487 kg) in that year

150   (Eurostat, 2017, DEFRA, 2018). The reduction target for 2025 seems has been achieved in

151   advance.

152   *2.2.   Goal and scope*

153   The goal of this study was to quantify and compare the GWP of three historical MSW

154   management strategies at three development stages in Nottingham, and a future scenario in

155   response to the EU directives. MSW is defined as the solid waste arising from household

156   sources, for consistency with targets set in waste regulations and available data. The functional

157   unit is defined as the treatment of one ton of MSW, to ensure that the presented scenarios are

158   comparable to each other. To assess the influence and importance of waste prevention on

159   establishing sustainable MSW management, GHG emissions from managing MSW generated

160   by each person were also quantified.

161   *2.3.   System boundary*

162   The spatial boundary of the MSW management system is the administrative boundary of

163   Nottingham City Council. The overall system addressed in the present study is illustrated in

164   Fig. 2. It contains all waste management processes including the collection, transport, treatment

165   and disposal of waste. All possible future emissions were accounted for the year when the

166   MSW was managed. This is necessary to ensure that the calculations for all MSW management

167   scenarios comparable. The major sources of emissions were determined as follows:

168   • Fuel and power used in MSW management processes, but excluding emissions from

169   upstream activities such as mining and transport. Due to the evolution of energy mix, the

170   emission factors of electricity production were estimated to be 0.45kg CO_2 eq./kWh in 2002,

171   0.47 CO_2 eq./kWh in 2007 and 0.35 CO_2 eq./kWh in 2017.

172   • Waste collection.

173   • Transport to/between treatment facilities.

174   • Direct emissions from waste; for example, CO_2 from waste incineration.

175   • Avoided GHG emissions due to materials recycling and energy recovery.

176   • Environmental burdens from the operation of the CA and bring sites were excluded due to

177   data deficiency.

178   *2.4.   Scenarios*

179   In total, four MSW management scenarios including three historical scenarios and a future

180   scenario have been developed and assessed in this study (Fig. 3). The statistical year in the UK

181   is the period from April to the following March; for example, April 2016 – March 2017, so that

182   the years to our MSW management scenarios are expressed to cross two years, i.e. 2001/02.

183   The selection of scenarios was based on the enforcement time of EU waste directives and data

184   availability. The scenarios are discussed in detail in the following sub-sections.

185   *2.4.1. Description of Scenario S1: 2001/02*

186   This scenario relates to MSW management as at 2001/02, when the EU Landfill Directive

187   began to be enforced in Nottingham, and is the earliest year for which complete data is available.

188   In this scenario, weekly house-to-house collection without separation was provided by the local

189   authority (Parfitt et al., 2001). A transfer station was used to store and transfer waste to landfill.

190   MSW was disposed in landfills (54.7%) and incinerated at the Easrcroft EfW facility (40.7%)

191   (NCC, 2005). Under these circumstances, the compositions of incinerated and landfilled MSW

192   were assumed to be the same (Table 1 and 2). 3.4% and 1.2% of MSW were recycled and

193   composted (NCC, 2005). Materials were recycled at the CA site and bring sites. Recycled

194   materials were assumed to be paper, glass and metal (estimated at 50%, 25% and 25% of

195   recycled materials, respectively) (Data.Gov, 2018). Garden waste was composted via open

196   windrow composting. Pretreatment before incineration/landfill and methane collection systems

197   at the landfill were unavailable. Bottom ash from incineration (BAI) was landfilled.

198   *2.4.2. Description of Scenario S2: 2006/07*

199   S2 corresponds to the year 2006/07, before the enforcement of the EU Waste Framework

200   Directive. It is the earliest year of documented waste flows. In this scenario, new waste

201   management initiatives, such as the KCS, bespoke bulky waste collection and MRF, had been

202   introduced but were not fully implemented (Fig. 2). A transfer station was still used, but now

203   to store and transfer waste to MRF. Landfilling rate was reduced to 32.7% because of the

204   improved recycling (17.5%) and composting (8.6%) rates. 41.2% of waste sent for EfW. Metal

205   from BAI was recycled. The compositions of MSW and incinerated waste are illustrated in

206   Table 1 and 2.

207   *2.4.3.   Description of Scenario S3: 2016/17*

208   S3 corresponds to the year of 2016/17 and represents the most recent full year for which

209   data was available for our analysis (Fig. 2). KCS was further strengthened to serve all

210   households in Nottingham, which led to increased recycling and composting rates of 31.5 %

211   and 12.9%, respectively. Production of RDF was also introduced. BAI was recycled for

212   aggregates. Landfill became the least favorable waste disposal method with 7.3% of MSW

213   landfilled. 57.6% of MSW was incinerated for energy recovery.

214   *2.4.4.   Description of Scenario S4: Future scenario*

215   Based on our experience in analysing historical MSW management scenarios, an

216   alternative future scenario is proposed, to further improve the material and energy recovery

217   capability of the MSW management system in Nottingham. This scenario was constructed

218   based on the same quantity and quality of waste in 2016/17. Food waste is separately collected.

219   Anaerobic digestion (AD) replaces open windrow composting for treating food and garden

220   waste. Biogas from AD is utilized for power and heat generation. Regularly collected residual

221   waste is pre-treated in the residual MRF for material recycling before incineration.

222   *2.5.   Life cycle inventories*

223   *2.5.1.   Collection, transfer and transport*

224   Detailed estimations of the travel distance and LCI for MSW collection and transport are

225   presented in Appendix Section S1. Electricity and diesel consumption due to the transfer station

226   was assumed to be 4 kWh/t and 0.84 kg/t, respectively (Turner et al., 2016).

227   *2.5.2.   Landfill*

228   1.8 kg/t diesel and 8 kWh/t electricity were assumed to be consumed for operating landfill

229   (Turner et al., 2016). The amount of methane emitted from landfill can be estimated based on

230 equations reported by Fong et al. (2015) (Presented in SI Section S2). This method calculates

231 the total mass of methane potentially generated based on the mass and composition of landfilled

232 waste as listed in Table 1.

233 *2.5.3. Incineration with energy recovery*

234 The flue gas emitted from the incinerator fed by MSW after treatment mainly contains CO_2,

235 but also some trace gases including CO, SO_2, NOx and N_2O, etc. Given that CO_2 capture is not

236 in place in most waste incineration plants worldwide, the quantity of CO_2 emitted from the

237 incinerator could be calculated based on the mass and composition of the incinerated waste

238 (Table 2) using equations provided by the IPCC (2006) (Presented in SI Section S2). Air

239 pollution control equipment, such as selective noncatalytic reduction (SNCR) for the reduction

240 of nitrogen oxides, was installed by Eastcroft EfW to control the emission of air pollutants

241 (FCC Environment, 2015). After treatment, the concentrations of methane and NO_x emitted

242 from the incinerator was under the emission limit values set by the EU (EC, 2000, WRG, 2008,

243 FCC Environment, 2015). Thus, the GWP of methane and NO_x emitted from MSW combustion

244 were ignored.

245 Eastcroft EfW could harness 89% of the LHV of MSW to produce steam (FCC

246 Environment, 2015). This steam is sent to an energy generation facility for electricity and hot

247 water production with conversion efficiencies of 17.2% and 31.7%, respectively (FCC

248 Environment, 2015). 62 kWh/t of recovered electricity and 3.76 kg/t fuel oil were consumed in

249 operating the incineration plant (WRG, 2008). The LHV of incinerated waste was estimated

250 through physical composition based empirical model (Eq. 1), developed by the authors using

251 151 datasets collected from 47 cities in 12 countries.

252 $LHV \ (kJ/kg) = \ -72.42Pr + 83.20Pa + 67.90Pl + 7669.08$ \hfill (1)

253   Where *Pr* is the percentage of putrescible including food waste and garden waste, *Pa* is the

254   percentage of paper; and *Pl* is the percentage of plastics. The value of percentage is within the

255   range between 0 and 100.

256   Recovered heat from waste was assumed to substitute the equivalent heat generated from

257   gas boilers, as these dominate home heating in England, due to insufficient district heating

258   networks (Euroheat & Power, 2017, DECC, 2013). The majority of boilers available on the

259   British market have efficiencies in the range of 88 % and 89.7 % (Knight, 2018). Hence, 89 %

260   was uded in this study. The LHV of natural gas is 47.82 MJ/kg with a GHG emission factor of

261   2.72 kg CO_2-eq./kg (DEFRA, 2016). Based on these assumptions, the quantity of natural gas

262   and associated GHG emission saved by EfW were quantified.

263   *2.5.4. Recycling*

264   Avoided emissions by material recycling were modeled based on the England Carbon

265   Metric Report (DEFRA, 2012).

266   *2.5.5. Composting*

267   GHG emissions from composting were calculated after excluded the 36% non-compostable

268   fraction (NCC, 2013). Details of LCI for composting are presented in Table 3. The produced

269   compost was used to substitute inorganic N, P and K fertilizers. Hill et al. (2011) reported that

270   GHG emission from production 1 kg of inorganic N, P and K fertilizer were 6.8 kg CO_2-eq.,

271   1.2 kg CO_2-eq. and 0.5 kg CO_2-eq. respectively.

272   *2.5.6. Material recovery facility*

273   There are two types of MRF. One is designed to process comingled collected recyclables

274   for the recovery of paper, glass, plastics and cans. Diesel and electricity consumption in this

275   MRF are 2 kg/t and 35 kWh/t, respectively (Turner et al., 2016). The other is Residual MRF,

276 which is designed to recover materials from bulky waste, street waste and residual waste from

277 a CA site. Diesel and electricity consumption in a Residual MRF are 44 kWh/t and 2 kg/t,

278 respectively (Pressley et al., 2015, Turner et al., 2016).

279 *2.5.7. Production and incineration of RDF with energy recovery*

280 Burnley et al. (2011) recommended that electricity consumption in a facility with a yield

281 of RDF in the range of 14 – 22% was 40 kWh/t. The RDF yields in both types of MRF in

282 Nottingham were around 20%. RDF was assumed to be incinerated in a power plant to generate

283 electricity only. The efficiency of a dedicated RDF incineration plant was assumed to be higher

284 than the EfW plant; at 25% on an LHV basis (Burnley et al., 2011). The LHV of standard UK

285 MSW derived RDF is 25 MJ/kg with a fossil carbon content of 32% by weight (Materazzi et

286 al., 2015, IPCC, 2006). Emissions from RDF combustion could thus be calculated based on the

287 equations provided by IPCC (2006) (Presented in SI Section S2) .

288 *2.5.8. AD*

289 Biogas production with a yield of 20% by weight of which 63% is methane in an AD

290 process, was assumed (Zaccariello et al., 2015, Turner et al., 2016). The LHV of biogas is 30

291 MJ/kg (DEFRA, 2016). Biogas is used for electricity and heat production on site using the CHP

292 engine. Energy recovery efficiencies of 31% and 49% for electricity and heat were assumed

293 (Turner et al., 2016). A detailed LCI for the AD process is presented in Table 4.

294 *2.6. Impact assessment*

295 The life cycle impact assessment was characterized by GWP at a 100 year period (GWP_{100})

296 based on the results of the inventories using the IPCC 2013 GWP 100a method (IPCC, 2013).

297 This method provides a comprehensive methodology to calculate GWP_{100}, associated with

298 amount of GHG emission and its equivalency factor. The total GWP of the MSW management

299   is the sum of GWPs of all GHGs. The GHGs of interest in MSW management include carbon

300   dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). These GHGs account over 90% of total

301   GHG emissions from MSW management (Bogner et al., 2007). According to IPCC guidelines

302   on GHG inventories, only CO_2 from fossil origins is regarded to have a GWP (IPCC, 2006).

303   *2.7.   Interpretation*

304   Interpretation relates to the presentation of results and associated sensitivity analysis. LCA

305   results were presented in two ways: the GWP_{100} of managing 1 ton of MSW (expressed as

306   GWP_{100} per ton of MSW), and the GWP_{100} of managing MSW generated by each citizen

307   (expressed as GWP_{100} per capita). Sensitivity analysis is a crucial step in assessing the

308   reliability and robustness of LCA results, by understand how they are affected by changes in

309   certain parameters, such as waste composition and the adopted calculation models. In this study,

310   two sensitivity analyses were carried out. Sensitivity analysis 1 was carried out by varying the

311   DOC in landfilled waste, the content of N, P, K in composted organic waste (Table S9), and

312   the LHV and fossil carbon of RDF. Sensitivity analysis 2 was carried out by using another

313   LHV predictive model to estimate the LHV of incinerated MSW.

314   **3.   Results and discussions**

315   *3.1.Historical GWP_{100} of MSW management*

316   *3.1.1. GWP_{100} per ton of MSW*

317   The LCA results are presented in Fig. 4 – 5 and Table 5. Fig. 4a clearly illustrates that the

318   GWP_{100} of MSW management has significantly decreased from 1076.0 kg CO_2-eq./t of MSW

319   in 2001/02 to 211.3 kg CO_2-eq./t of MSW in 2016/17. This is mainly due to the diversion of

320   waste from landfill to more sustainable management options such as recycling, composting and

321   incineration. S1 has the highest GWP_{100} amongst all historical scenarios, because over half of

322 MSW was landfilled without any methane recovery, which made landfill the major emitter of

323 GHG, accounting for 82.5% of the total GWP_{100} in S1.

324 In S2, the GWP_{100} reduced to 487.9 kg CO_2-eq./t of MSW, less than 50% of that of S1. A

325 further reduction to half of that in S2 was achieved in S3 (Fig. 4a). This was because more

326 materials such as paper, plastics, glass and metal were recycled, more garden waste was

327 composted and RDF was produced. The fully implemented KCS improved the separate

328 delivery rate, so as to enhance the quantity and quality of recycled materials. Recycled

329 materials compensate the equivalent GWP_{100} from the consumption of virgin materials and

330 fossil fuels.

331 Materials recycling was the only waste management practice that consistently resulted in

332 GWP_{100} savings in all historical scenarios. A significant reducing trend of GWP_{100} achieved by

333 materials recycling was observed from 2001/02 to 2006/07. This is mainly because the

334 introduction of KCS and MRF greatly improved the material recycling rate. However, GWP_{100}

335 contributed by materials recycling increased by 5.8 kg CO_2-eq./t of MSW in 2016/17 as

336 compared to that in 2006/07. The reason is that producing products from secondary materials

337 (recycled or recovered materials from waste) does not always cause less global warming impact

338 than from virgin resources (Björklund and Finnveden, 2005). DEFRA (2012) reported that it

339 produced more GHG to recycle food and beverage cartons than to produce it from virgin

340 materials in the UK. Alternative treatment options should be considered to treat these materials,

341 which could cause greater GWP to recycle it, or to improve the efficiency of recycling and

342 reprocessing. As Fig. 5 depicted, GWP_{100} saved by recycling varies among materials.

343 Recycling metals followed by recycling paper, saved the most GHG emission in all historical

344 scenarios. The quantity of recycled paper was far more than for other recycled materials in both

345 2006/07 and 2016/17, but the GWP_{100} saved by recycling paper was less than metal recycling

346 because chemical and fossil fuel consumption in paper recycling was greater (Habib et al.,

347   2013), and the substituted CO_2 emission from steel manufacturing from virgin material was

348   relatively higher (Rankin, 2012, Burchart-Korol, 2013, Laurijssen, 2013).

349   Composting of garden waste was a contributor of GWP_{100} in all historical MSW

350   management scenarios, because open windrow composting was applied, through which GHGs

351   were directly emitted to the ambient atmosphere and no energy was recovered. The detailed

352   LCA result for the composting process indicates that the production of organic fertilizer

353   avoided the utilization of inorganic fertilizers (N, P, K) and cut the overall GWP_{100}, but the

354   GHG emission from decomposition and facility operation was more than the saved amount.

355   The gross GWP_{100} of composting was 122.5 kg CO_2-eq./t of garden waste, while the saved

356   GWP_{100} by inorganic fertilizer avoidance was only 20.4 kg CO_2-eq./t of garden waste.

357   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-

358   eq./t of MSW in S1, S2 and S3, which accounted for 18.1%, 55.9% and 81.8% of GWP_{100} in

359   these scenarios, respectively. The energy recovery efficiency in Nottingham was 15.3% for

360   electricity and 28.2% for heat, which appeared to be lower than other cases reported in the

361   literature. Reimann (2012) reported that average energy recovery efficiency in European EfW

362   plants was 26.1% in the case of electricity production only, 77.2% in case of heat production

363   only and 52.1% in case of CHP. Habib et al. (2013) reported that the gross energy recovery

364   efficiency of EfW reached 28% for electricity and 85% for heat in Aalborg, Denmark, which

365   made MSW management in that city a GHG saver. Therefore, upgrading the EfW facility to

366   improve the energy recovery efficiency is recommended as a possible solution to improve the

367   future environmental performance of the waste management system in Nottingham.

368   The quantity and share of GWP_{100} contributed by collection and transport were lower

369   compared to other processes, but an obvious increasing trend has been observed during the

370   period of study. As MSW management options were shifted to upper layers of the waste

371   management hierarchy, the GWP_{100} generated by transport increased significantly from 4.7 kg

372   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}

373   from collection stayed relatively stable with a gentle declining trend during the same period

374   (Table 5). The reduction in GWP_{100} from collection is due to the amount of waste collected at

375   bring sites and street cleaning was reduced due to the introduction of KCS. Generally, a

376   relatively longer distance was traveled to collect recyclables from distributed bring sites and to

377   clean streets than to collect waste through KCS. The GWP_{100} of transporting recycled materials

378   to reprocessing facilities increased significantly (Table 5), due to two factors: more materials

379   were recycled, and reprocessing facilities were usually located some distance from Nottingham.

380   For example, recycled glass and paper was transported 173 km and to overseas for reprocessing,

381   respectively. GWP_{100} of transporting recycled materials to reprocessing facilities in S3 was

382   nearly 44 times and 9 times more than those in S1 and S2, respectively. The increased GWP_{100}

383   by transport led to the increase of overall GWP_{100} from materials recycling. Similar result was

384   observed by Turner et al. (2016) and they suggested that promoting domestic reprocessing of

385   secondary materials was a possible solution to reduce the GWP_{100} from transport and

386   eventually enhance the overall environmental benefits from materials recycling.

387   *3.1.2. GWP_{100} per capita*

388   Similarly, GWP_{100} per capita significantly reduced from 498.2 kg CO_2-eq. in 2001/02 to

389   76.3 kg CO_2-eq. in 2016/17, a nearly sevenfold reduction (Fig. 4b). This is due to the

390   improvements in MSW management discussed in section 3.1.1, as well as efforts in waste

391   prevention. MSW generation per capita decreased from 463 kg to 361 kg during the same

392   period (Fig. S1). GWP_{100} added by collection and transport increased significantly from 0.4 kg

393   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

394   also been detailed in section 3.1.1.

395   *3.2.GWP_{100} in the future scenario (S4)*

396   MSW management in S4 becomes a net saver of GHG emissions, due to improvements in

397   material recycling and waste treatment. Both GWP_{100} per ton of MSW and GWP_{100} per capita

398   reduce to just –142.3 kg CO_2-eq. (Fig. 4a) and –40.2 kg CO_2-eq (Fig. 4b), respectively. AD

399   reduces GWP_{100}, because of energy recovery from biogas. 81.3 kg CO_2-eq./t of MSW will be

400   saved when garden waste and food waste are treated by AD. Incineration will be another saver

401   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

402   materials recycling will be further improved to 257.5 kg CO_2-eq./t of MSW because more

403   materials are recycled from residual waste. However, EfW and combustion of RDF will

404   consistently be GHG emitters, if no more advanced technology is applied to improve the EfW's

405   energy recovery efficiency. GWP_{100} from transport in S4 will increase, since more materials

406   are transported for recycling (Table 5).

407   In addition to improving the recycling/composting rate and upgrading the biological

408   treatment technology to reduce GWP from MSW management, attention should also be paid

409   to the quality of secondary products from recycled materials and compost. Accumulation of

410   hazardous substances in recycled materials reduces the quality of products made up of

411   secondary materials and increases the release potential of hazardous substances (Kral et al.,

412   2013). An apparent example is found in the steel industry where copper contaminates the steel

413   cycle (Kral et al., 2013). The accumulation of copper hardens steel and decreases steel quality

414   (Haupt et al., 2017). Recycling material from mixed residual waste could improve the recycling

415   rate, but also introduce contaminates to recycled materials, and this will reduce the quality of

416   secondary products made from them. Production of RDF might be an alternative option. The

417   suitability of compost from bio-treatment as fertilizer is influenced by the quality of feedstock

418   (proteins, minerals, and presence of undesirable materials) which depends mainly on the source

419 separation (Kumar and Samadder, 2017). Thus, enhancing source separation and public

420 participation will be crucial to improve the quality of secondary products.

421 *3.3. Sensitivity analysis*

422 As presented in Table 6 and Fig.6, sensitivity analysis results indicate that the variations in

423 waste composition and the LHV prediction model affect the estimated GWP_{100} values, but not

424 the downwards trend.

425 The DOC (Table 1), N, P and K (Table S9) contents in organic waste varied within a range

426 due to the diversified compositions within this category (Boldrin et al., 2009). Furthermore, the

427 LHV and fossil carbon of RDF in the UK vary in the ranges $13 - 25$ MJ/kg and $21.7 - 32.0$ %,

428 respectively, depending on its composition (Burnley et al., 2011, Materazzi et al., 2015). All

429 these variations in waste composition affect the total GWP_{100} of MSW management. Table 6

430 illustrates the minimum and maximum GHG emission from managing 1 ton of MSW when the

431 variations in waste composition are taken into consideration.

432 To assess the sensitivity of LCA results affected by the LHV predicting model, the model

433 developed by Khan and Abu-Ghararah (1991) (Eq. 2), using global data collected and the same

434 explanatory variables as Eq. 1, was used to predict LHV of incinerated waste in S1, S3 and S4

435 (the LHV of incinerated waste in S2 was measured using a bomb calorimeter). As Fig. 6

436 illustrated, both the LHVs and associated GWP_{100} of incinerated waste in all three scenarios

437 change significantly when using Eq. 2. However, this model was developed 30 years ago, and

438 so may not be suitable for estimating the LHV of modern waste, because the characteristics of

439 MSW have changed dramatically during this period. Therefore, the updated model (Eq. 1) is

440 recommended to estimate the LHV of MSW. Nevertheless, the GWP_{100} of MSW management

441 in Nottingham is estimated to have reduced during the study period, irrespective of the model

442 adopted.

443 $$LHV \ (kJ/kg) = 53.5 \ (F + 3.6 \ Pa) + 372.16 \ Pl \tag{2}$$

444 **4.   Conclusions**

445   To assess the effectiveness of waste regulations and the evolution of MSW management

446 under the guidance of these regulations, in this study, LCA was carried out to estimate and

447 compare the GWP_{100} of three historical MSW management scenarios in Nottingham, since the

448 enforcement of the EU Landfill Directive. A further future scenario designed to meet the local

449 2025 recycling target and 2030 landfill target was also evaluated and compared with the

450 historical scenarios. The results indicate that both GWP_{100} per ton of MSW and GWP_{100} per

451 capita in Nottingham have reduced significantly during the last 16 years. Waste regulations

452 effectively incentivised the shifting of MSW management from a landfill centered mode to a

453 more environmentally friendly management approach. The results also indicate the importance

454 of waste prevention in mitigating the GWP of MSW management. In future works, other

455 environmental impacts in addition to GWP and sustainability at social and economic

456 dimensions of MSW management can be assessed to comprehensively assess the effectiveness

457 of waste regulations.

458   MSW management system in Nottingham is still a net emitter of GHGs, partly because of

459 the low energy recovery efficiency in EfW facility and increased emissions due to the transport

460 of materials for recycling. Thus, improving the energy recovery efficiency in EfW by

461 upgrading its technology and promoting domestic reprocessing of secondary materials are

462 recommended to mitigate GHG emission from MSW management. The LCA results of the

463 future-looking scenario indicate that separating food waste at source and treating it via AD,

464 pretreating residual waste before incineration and replacing open windrow composting by AD

465 could turn the MSW management system into a net saver of GWP_{100}. To achieve the future-

466 looking scenario, public participation also need to be enhanced to ensure the source separation.

467 Besides, attention should be paid to the quality of recycled and recovered materials.

468 **Acknowledgements**

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640   **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	-

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

644

645   **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]	-	-	-

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

649

650 **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)

651 **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)

652

653   **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

654

655   **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

656

657   **Fig.1.** The location of Nottingham in Nottinghamshire and the UK, and Lower Layer Super

658   Output Areas (LSOA) within Nottingham.

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

660   **Fig. 3.** Schematic illustration of MSW management in all scenarios assessed in the current

661   study. Newly introduced processes and changed waste flows are identified by different colors.

662   BAI represents bottom ash from the incineration plant.

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

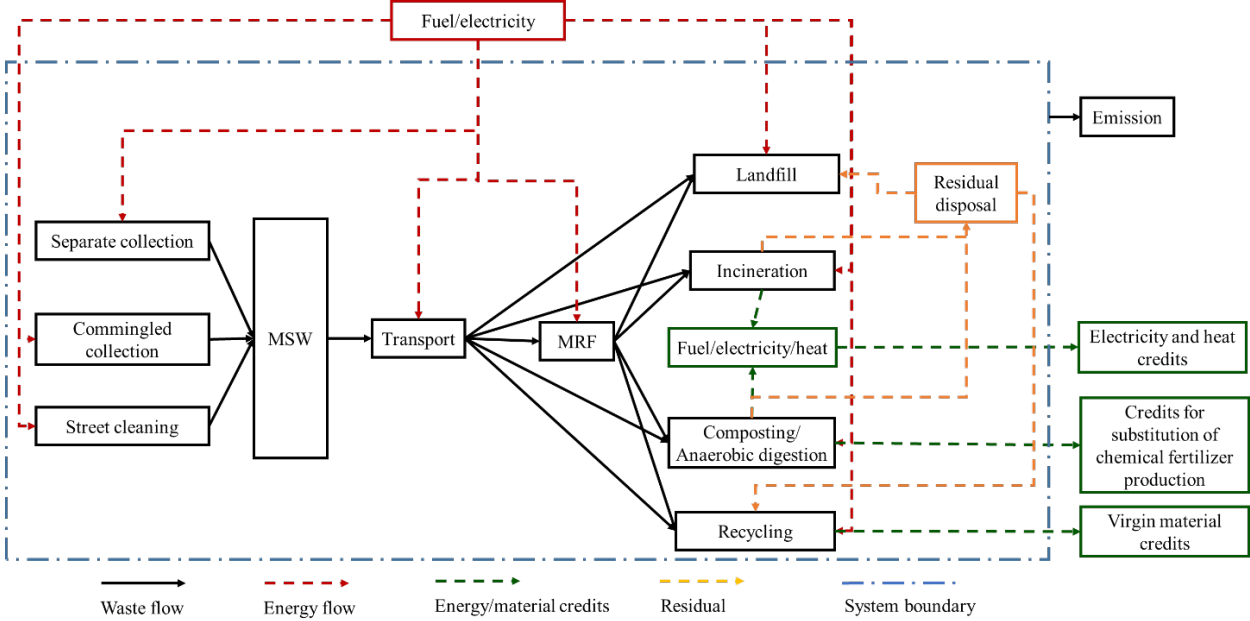664   MSW. (b): GWP100 per capita.

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

666   **Fig. 6.** Comparison between estimated LHVs (a) and GWP100 (b) of incinerated waste when

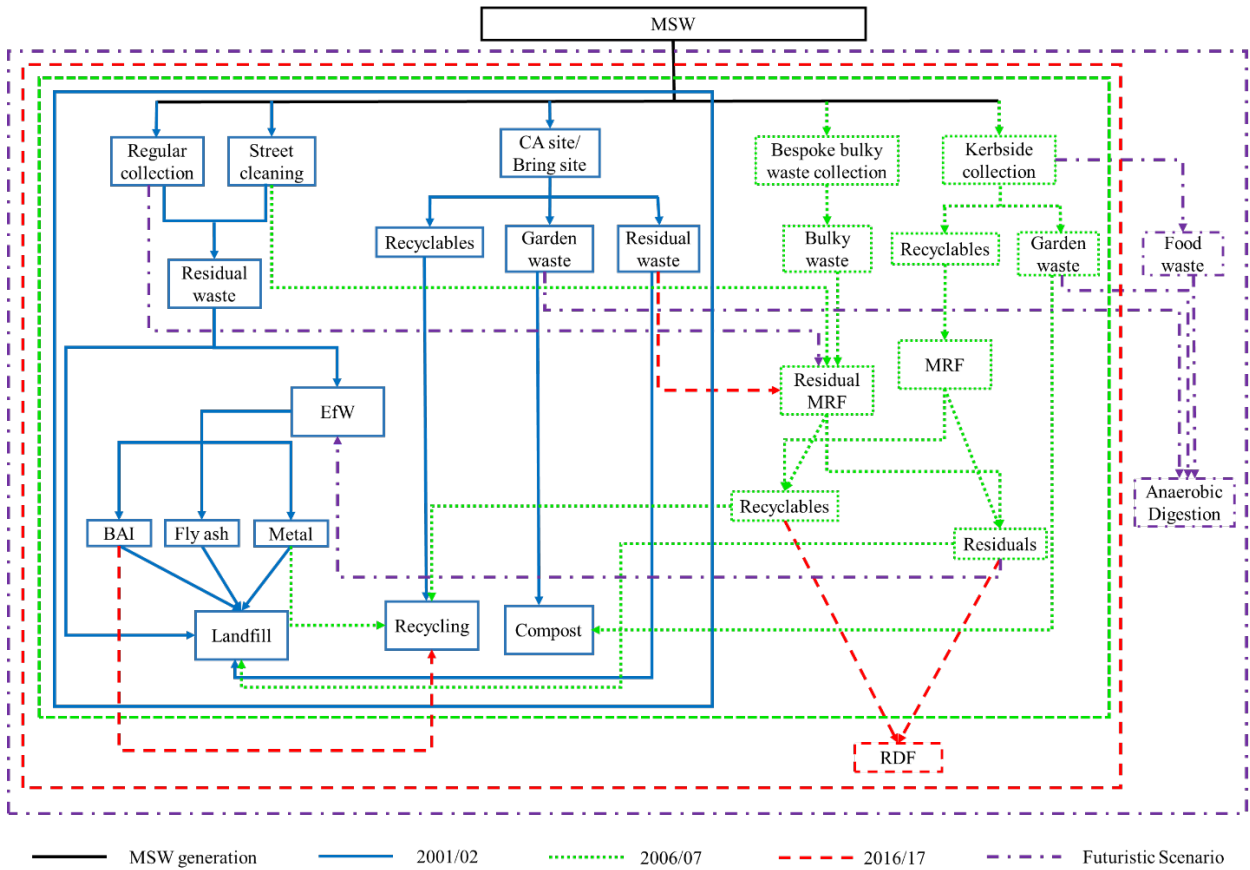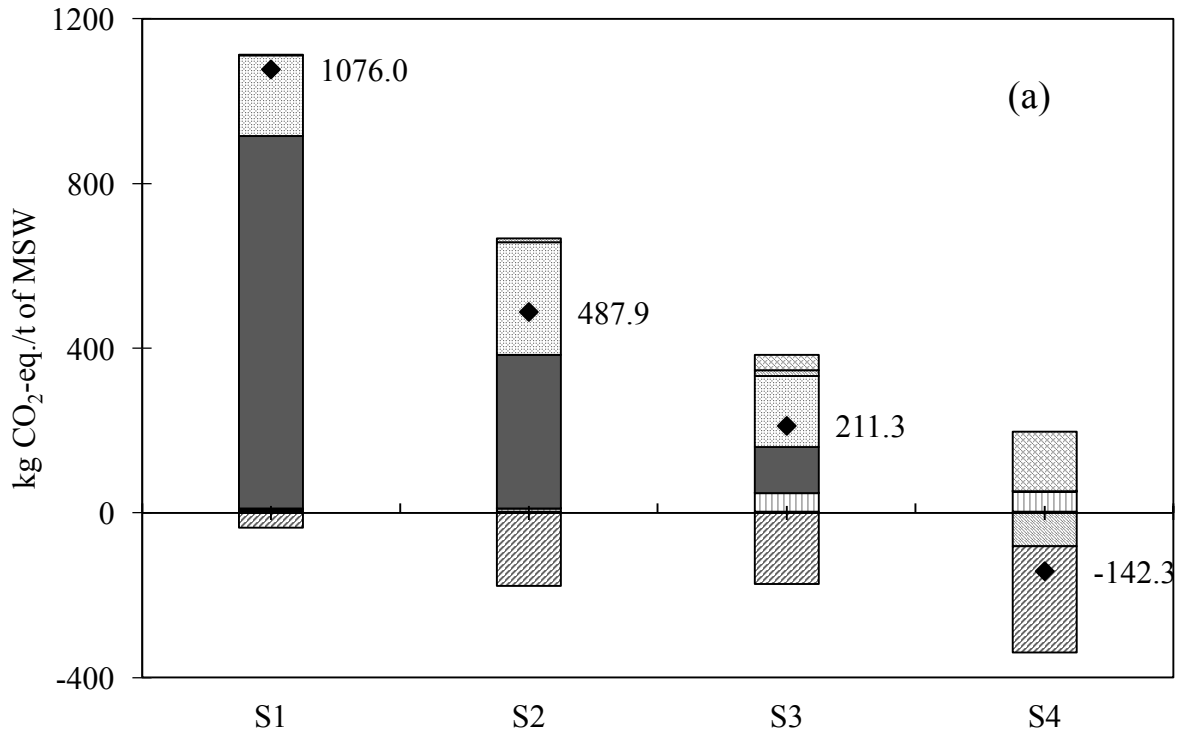667   different models were used to estimate its LHV.
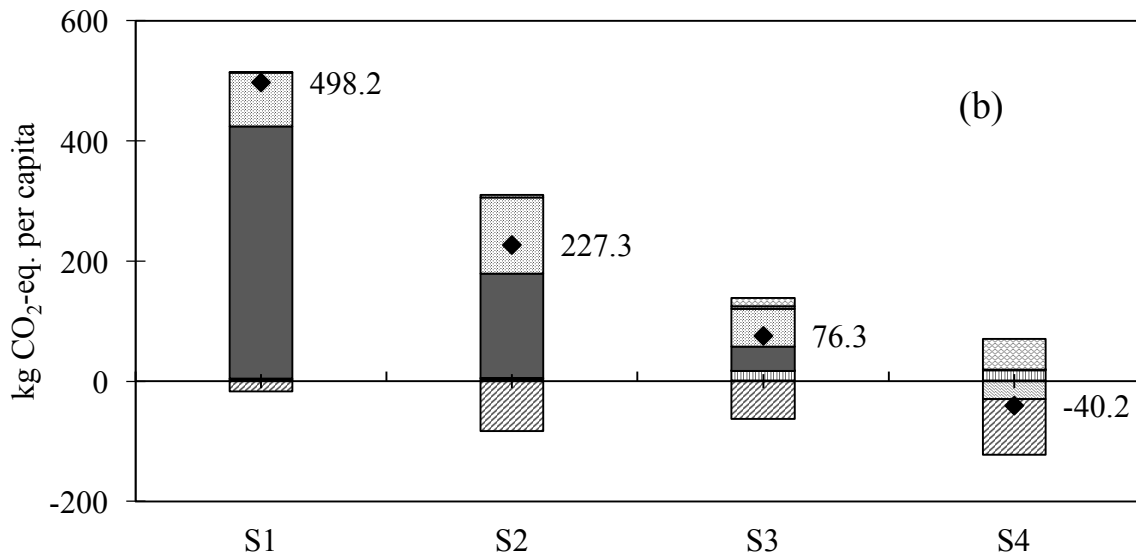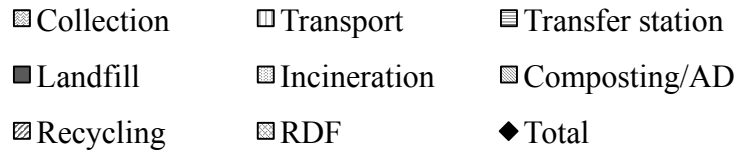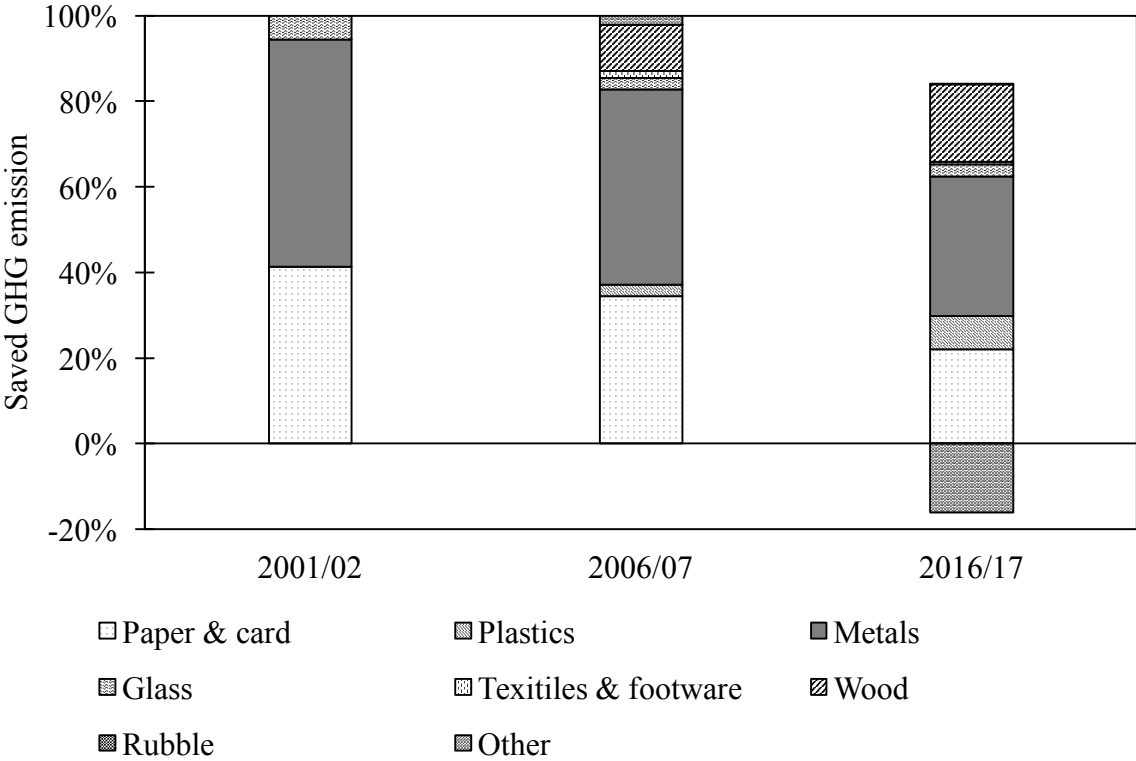
668

669 **Fig.1.**

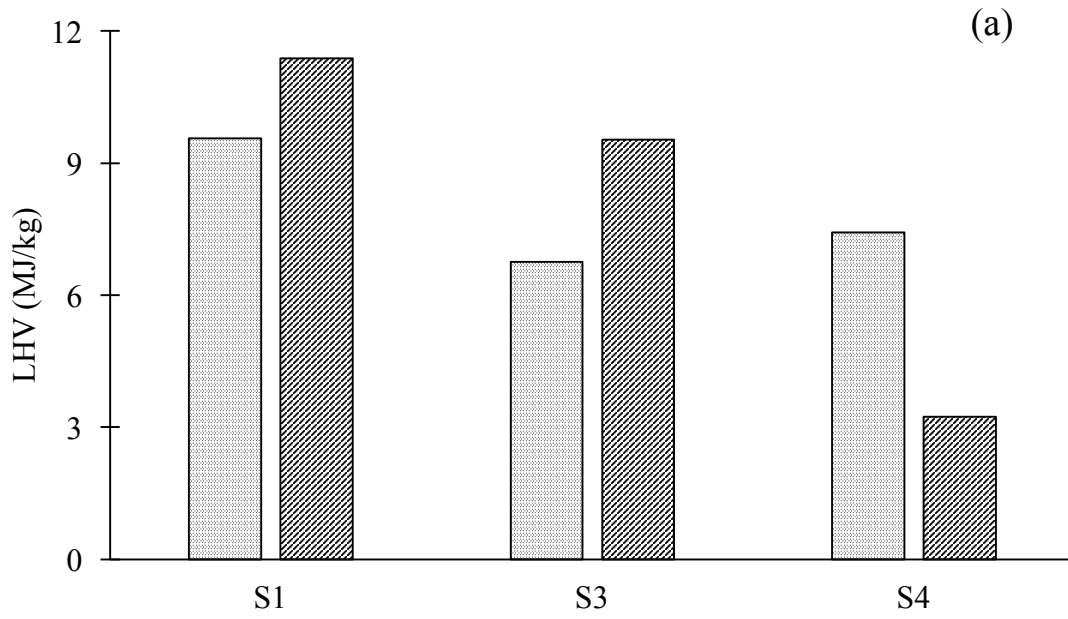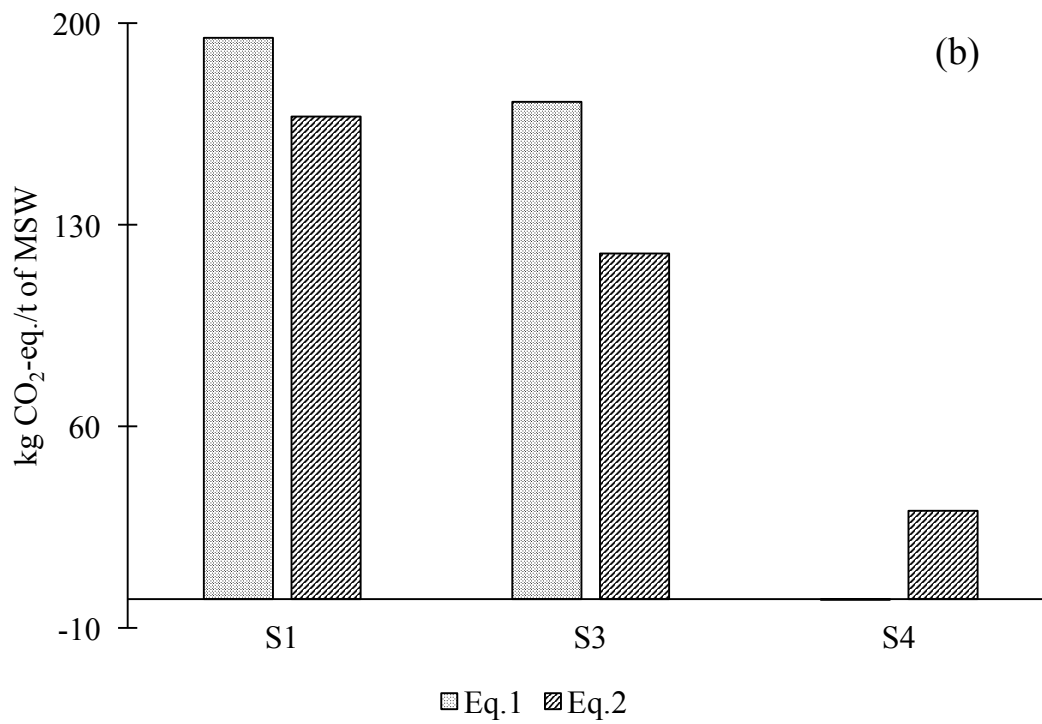670

Fig. 2.

672
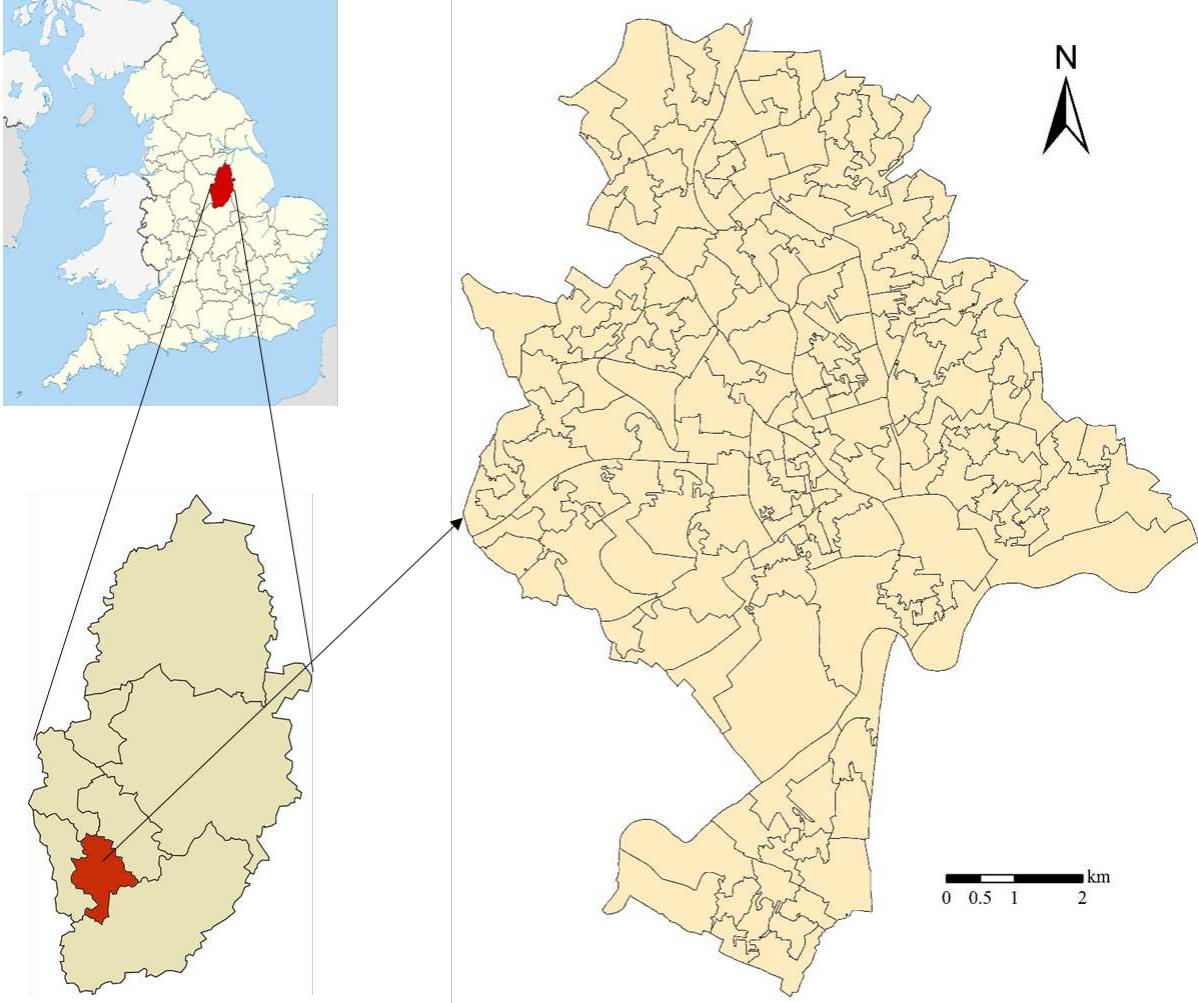
Fig. 3.

674

675

676 **Fig. 4.**

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678 **Fig. 5.**

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681

682   **Fig. 6.**