Performance Assessment of Municipal Solid Waste Management for A City in England: Past and Future Perspectives

By

Dan WANG

A thesis submitted to the University of Nottingham

for the degree of Doctor of Philosophy

December 2019

The University of Nottingham

UNITED KINGDOM · CHINA · MALAYSIA

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Abbreviations

ANN	Artificial neural network
BMW	Biodegradable municipal waste
CA	Civic Amenity
CHP	Combined heat and power
DEFRA	Department for Environment, Food & Rural Affairs
DECC	Department of Energy & Climate Change
DBEIS	Department for Business, Energy & Industrial Strategy
EC	European Commission
EEA	European Environmental Agency
EfW	Energy from Waste
GHG	Greenhouse gas
GPC	Waste generation per capita
GWP	Global warming potential
KCS	Kerbside collection service
LATS	Landfill Allowance Trading Scheme
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
LCR	Landfill rate
LCI	Life cycle inventory
LHV	Lower heating value
LSOA	Lower Layer Super Output Areas
MAPE	Mean absolute percentage error
MBT	Mechanical biological treatment
MFA	Materials flow analysis
MSW	Municipal solid waste
MRF	Material recovery facility
NCC	Nottingham City Council
RCR	Recycling rate
RECR	Recovery rate
RDF	Refuse derived fuel
SDR	Separate delivery rate
SEE	Standard error of the estimate
WRAP	Waste and Resource Action Programme
WEEE	Waste Electrical and Electronic Equipment

Abstract

Sustainable municipal solid waste (MSW) management is the key to achieve urban sustainability and circular economy via recycling materials and recovering energy from waste. It also contributes to mitigating global climate change and reducing impacts on human health and the environment. To combat the environmental impacts caused by MSW management, especially the impacts from landfills, EU Landfill Directive was introduced in 1999 to reduce the landfilled biodegradable municipal waste (BMW). After that, a series of waste directives have been put into enforcement to facilitate the establishment of sustainable MSW management in European countries. These waste directives initiated and drove the evolution of MSW management in the UK, but their realistic effects on the improvement of MSW management has not to date been investigated in detail.

This study depicted the transition and assessed the performance of MSW management in Nottingham since the Waste Strategy 2000 published to respond to the EU Landfill Directive (throughout the period from April 2001 to March 2017) using the methods materials flow analysis (MFA) and life cycle assessment (LCA). To quantitatively estimate the potential energy that can be recovered from MSW, predictive models were built to estimate the lower heating value (LHV) of MSW using the two model building methods, multiple regression analysis and artificial neural network (ANN), based on 151 datasets which record the wet physical components of MSW and their measured LHV worldwide.

Improvements in waste reduction, material recycling, energy recovery, and landfill prevention were observed in Nottingham because of the voluntary preventive actions and the introduction of new waste management strategies such as kerbside collection,

material recovery facility (MRF) and production of residual derived fuel (RDF). MSW management in Nottingham transformed from a relatively simple model combining landfilling and incineration with energy recovery to a more complex model integrating source separation, recycling, composting, pre-treating landfilled waste and incineration with energy recovery.

During the period from 2001/02 to 2016/17, annual waste generation in Nottingham reduced from 463 kg/Ca to 361 kg/Ca, the recycling and composting share increased from 4.6% to 44.4%, and the landfill share reduced from 54.7% to 7.3%. These improvements resulted in a significant reduction of the greenhouse gas (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. These signs of progress are believed to be driven by the EU waste directives and the associated policies and waste management targets established at both national and local levels.

LHV predictive models generated using both methods (multiple regression analysis and ANN) exhibited acceptable and compatible levels of performance, based on the diagnostic tests on residuals (range and standard deviation), the standard error of the estimate (SEE) and mean absolute percentage error (MAPE). ANN models proved to be more robust in their handling of datasets of diverse quality. Models developed from both methods indicated a negative contribution of the wet food waste to LHV. Supported by the strong and significant correlation between food waste and moisture content, we concluded that the negative impact of the component with high moisture content on LHV outweighed its calorific value. This reveals that separating food waste or any other waste with high moisture content from the MSW to be incinerated can be the key to improved energy recovery efficiency.

Based on the experience in analysing historical MSW management scenarios and building LHV predictive models, an alternative future scenario was proposed that assumed food waste and textile were separated at source, separately collected organic waste was treated using anaerobic digestion and residual waste was pre-treated in MRF before incineration. Based on the assessment using MFA and LCA, this future scenario could fulfil the high target to recycle 55% of household waste set by the local authority, and GHG emission from MSW management sector in Nottingham could be further reduced to as low as – 142.3 kg CO_2-eq./t of MSW (or – 40.2 kg CO_2-eq./Ca). To realise this future scenario, enhanced source separation is especially important. Therefore, public education and supporting facilities on waste separation at the sources should be strengthened in the future.

Acknowledgements

First of all, I would like to express my sincere thanks to my supervisors, Professor Jun He, Dr. Yu-Ting Tang, Dr. Gavin Long and Professor Darren Robison for their constant guidance, invaluable suggestions, constructive comments, critical revision and consistent encouragements throughout my research journey and thesis writing. They provided me enough freedom and support to focus on the topics I was interested in. All they did for me made it possible for me to complete this thesis.

Next, I would like to thank the International Doctoral Innovation Centre (IDIC), Ningbo Education Bureau, Ningbo Science and Technology Bureau (Innovation Team Project (2017C510001)), and the University of Nottingham for offering me the financial support to pursue the Ph.D. programme. Special thanks go to all staff of FoSE, IDIC and graduate school, and all members in Professor Jun He's research group for their excellent support in my work and life.

I would also like to extend my sincere thanks to Professor David Higgitt of Lancaster University College at Beijing Jiaotong University, Professor Luo Guo of Minzu University of China, and Dr. Andy Allen for their invaluable comments and suggestions.

This work is dedicated to my beloved husband Mr. Wei Wang, my parents and sisters. They have always been there, believing in me, encouraging me and providing financial and emotional support to me. Without their love, support and encouragement, I could not complete this long journey.

Last, I would like to thank all my dearest friends for their help and companionship during this journey. Their support made my time in Ningbo very unforgettable and joyful.

Chapter 1 Introduction

Cities house near 55% of the global population with a yearly growth rate of 1.996%. Cities consume over 75% of global resources, generate 80% of anthropogenic CO_2 emission, and produce the majority of solid waste (EFCA, 2013; World Bank, 2017a; World Bank, 2017b). Cities often have regional, even global, hinterland for resource supply. The resources and energy inputs into and waste outputs from cities are still growing as a consequence of the combined effects of population growth, urbanization, industrialisation, improved living standards and consequently increased consumption per capita in cities. The increasing material and energy exchange across the urban boundaries uncovers critical worldwide challenges to accommodate a growing population with increasing demands of resources, and to safely dispose of growing and ever diversified waste generation. This leads to the development of new strategies that reform the way cities function to maintain their sustainability.

Current urban metabolisms are mostly functioning in linear style with large amounts of inputs of energy and materials, such as fuels, natural gas, foods, water and steel, and large volumes of outputs of solid waste, sewage and airborne emissions (Figure 1.1). This type of metabolism places great burdens not only on the local environment but also on the global environment. Comparatively, natural ecosystems are self-sustaining and they metabolise circularly through materials reusing and recycling (Chapin III et al., 1996). By learning from the ecosystems' metabolisms, academia, industries and governments are working toward new theories, frameworks, technologies and policies that help cities to transfer their linear urban metabolisms to more circular ones (Figure 1.1). This type of transformation can be made in the area of waste management. Methodologies, strategies, technologies, frameworks and policies can be developed to prevent unnecessary waste generation, to improve the material recycling and energy recovery from waste, and to enhance the exchange of recovered/recycled resources among cities, hence to improve the urban sustainability (Figure 1.1). Thus, waste is increasingly regarded as a secondary resource that can be exploited to support resource conservation, and waste management is considered as an anchor for the circular economy (Bergeron, 2017).

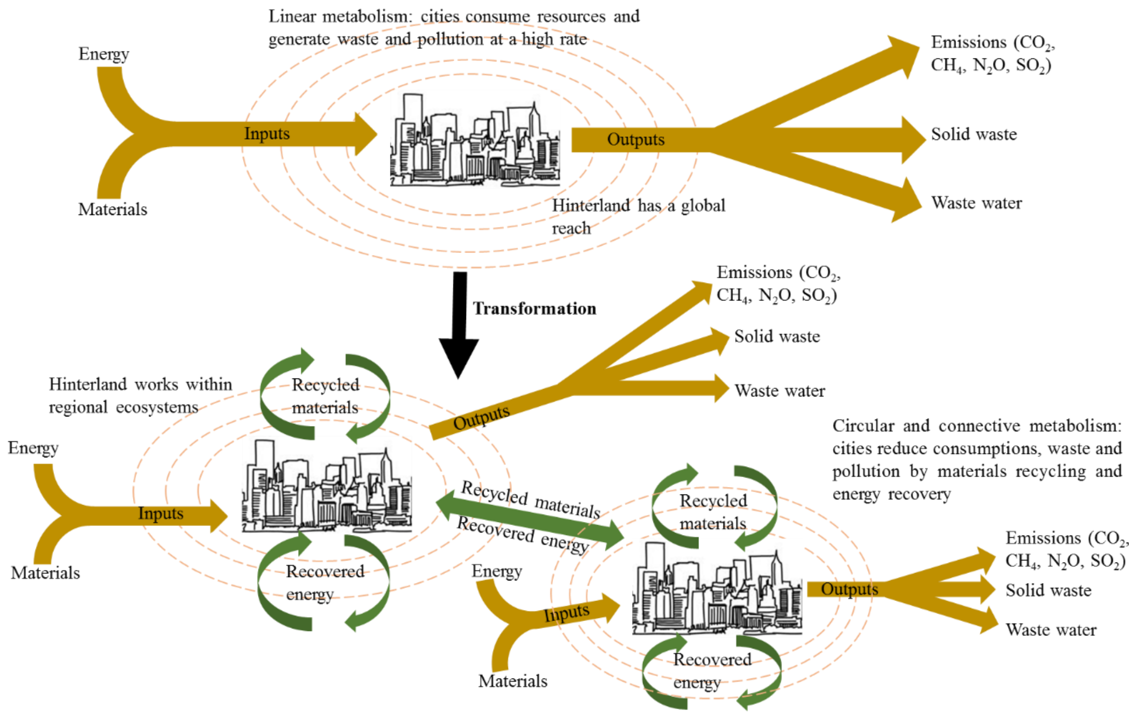


Figure 1.1. Transformation from linear urban metabolism to circular urban metabolism through material recycling and energy recovery from waste and enhanced exchange of recovered/recycled resources among cities (source: Girardet, 2010).

In 2016, globally generated municipal solid waste (MSW) in total was approximately 2.01 billion tonnes, and this is expected to grow to 3.40 billion tonnes by 2050 (Kaza et al., 2018). Most MSW was disposed of in the way of landfilling and dumping. Landfilling and dumping causes severe environmental consequences such as climate change (methane emission because of the decomposition of biodegradable municipal waste (BMW)), human health damages (e.g. exposure to chemical, bacteria and particles from landfills), ecosystem damages (e.g. leachate and gas emission from landfills to atmosphere, water and soil), unexpected disasters (e.g. fires and explosions resulted from gas and leachate migration from landfills), to name but a few (El-Fadel et al., 1997; Laurent et al., 2014a). Thus, the landfill is commonly regarded as the least preferable MSW disposal method, but these adverse impacts from landfills can be diminished by adopting more sustainable MSW management strategies such as material recycling, materials and energy recovery (Laurent et al., 2014b; Brunner and Rechberger, 2015).

For handling the increased quantity and ever diversified composition of MSW and diminishing the adverse impacts of MSW, especially the adverse impacts from landfilled MSW, as well as enhancing urban sustainability, a series of EU waste directives had been published by European Commission to facilitate the establishment of sustainable MSW management. MSW management objectives and targets were outlined in these directives and EU member states were legally obligated to establish and enforce policy instruments to meet these targets (Chapter 4). Targets and obligations in the EU Landfill Directive for diverting waste from landfill initiated the development and improvement of waste management in European countries.

To achieve the targets set in EU waste directives, various strategies, technologies and techniques aiming at improving the recycling and recovery from waste, as well as waste prevention, have been adopted in England in the last two decades. For example, four waste management strategies in compliance with the EU waste directives were published to fulfil the requirements by EU, and progress has been made during the last two decades. This will be illustrated in detail in Chapter 4. Kerbside collection service (KCS) was initiated to separate recyclable materials at sources. Residual derived fuel (RDF) was produced and exported for energy recovery in response to the rising gate fee of landfill in England (DEFRA, 2013b). As a result, MSW management in England has been significantly changed since the enforcement of the EU Landfill Directive.

1.1. Problem Statement

Many studies have been conducted to identify the challenges and gaps of meeting the targets set in the EU waste directives (Price, 2001; Lasaridi, 2009; Stanic-Maruna and Fellner, 2012; Závodská et al., 2014), to analyse the influences EU waste directives on local waste management legislations and practices (Taşeli, 2007; Stanic-Maruna and Fellner, 2012; Závodská et al., 2014; Scharff, 2014), and to evaluate the potential environmental impacts of waste management technologies or scenarios (Pires et al., 2007; Emery et al., 2007; Ionescu et al., 2013; Závodská et al., 2014). However, less attention has been paid to the process of how EU waste directives have driven the evolution of MSW management and the extent to which the performance of MSW management has been improved under the guidance of these directives. Examining the evolution of MSW management is essential for evaluating the drawbacks and effectiveness of the various strategies, so as to provide supports for improvements.

The evolution of MSW management driven by the EU waste directives can be measured by tracking the change of waste management legislations and/or strategies responding to these directives and comparing the historical and current waste flows and the performance of an MSW management system via appropriately selected indicators. Uyarra and Gee (2013) investigated the transformation of sustainable waste management and the processes leading to the transformation in Greater Manchester. Pomberger et al. (2017) assessed the MSW management performance concerning the rate of recycling, composting, incineration, and landfilling, from 1995 to 2014 in all 28 EU member states. Castillo-Giménez et al. (2019) assessed the performance of MSW treatment in 27 EU member states during the period 1995-2016 at the country and year level. However, these studies focused on the destination of waste and the achievements of targets, less attention has been paid on the change of pathways of waste streams and the associated environmental impacts from a life cycle perspective.

Waste prevention which ranks at the top of the “waste management hierarchy” is considered as the most important strategy to be integrated into any MSW management because it avoids expending energy at the post-consumer states to recycle, to treat and to dispose waste thus improves net energy balances and material efficiency (Tseng et al., 2018). Waste prevention (or reduction) targets are sometimes set by local authorities. Policies and regulations on waste prevention, such as charging for single-use carrier bags, have been enforced in England. Campaigns are also launched to educate the public to prevent unnecessary waste generation. However, the performance of waste prevention and its associated impact has seldom been assessed (Ekvall et al., 2007; Habib et al., 2013; Matsuda et al., 2018).

In this vein, this study have evaluated and compared the historical performance of MSW management in Nottingham after the enforcement of the EU Landfill Directive (from April 2001 to March 2017) to assess the effectiveness of the EU waste directives and to evaluate how the implementation of new waste management strategies and regulations over time has affected the performance of MSW management at a selected city using material flow analysis (MFA) and life cycle assessment (LCA) against appropriately selected indicators. Nottingham was chosen as the study city because it has changed its MSW management strategy several times to respond to the EU and national waste regulations since 2000, and ambitious MSW management targets (e.g., recycling 55% of household waste, and achieving “zero waste to landfill”) have been set by Nottingham City Council.

Energy recovery is an essential part of a sustainable MSW management system, and accurately estimating the heating value of MSW is crucial for the design and operation of energy from waste (EfW) facilities, especially the waste incineration plant. Empirical models were often used to estimate the heating value of MSW; however, models built in previous studies cannot apply at the regional or global scale because the datasets collected to build predictive models were usually restricted to a small region and had not been validated using regional or global data. Therefore, to successfully estimate the heating value of incinerated MSW, lower heating value (LHV) predictive models were built to aid the assessment of the performance of MSW management.

1.2. Purpose of This Study

The purpose of this study was to gain a better understanding of the evolution of MSW management in a British city under the guidance of EU and national waste directives and strategies with the aim of identifying the problems and key issues affecting the transition of MSW management in the city, as well as the positive and negative effectiveness in relation to waste strategies/policies, so as to identify the possible solutions to the problems and further improvement. In line with this, the specific objectives guiding this study were:

- To review EU, national and local waste regulations and strategies since the enforcement of the EU Landfill Directive in1999. → Addressed in Chapter 4.
- To describe the MSW management situations in Nottingham. →Addressed in section 3.4.
- To identify the pathways of waste streams at different stages of the evolution of MSW management. → Addressed in Chapter 6.
- To assess the performance of MSW management scenarios. → Addressed in Chapter 6 and Chapter 7.
- To identify the successes and failures of all MSW management scenarios. → Addressed in Chapter 6 and Chapter 7.
- To identify ways to improve the MSW management in Nottingham. → Addressed in Chapter 6 and Chapter 7.
- To build LHV predictive model which can be used at global scale. → Addressed in Chapter 5.

1.3. Impact of This Study

The impact and novelty of this study can be summarized from seven aspects as follows:

- This study identified the changes of MSW management in a historical context. It gives an insight into how waste management policies and regulations drive the evolution of MSW management. → Addressed in Chapter 4 and Chapter 6.
- This study identified the gaps where the enforcement of policies, regulations, strategies and technologies can be strengthened. → Addressed in Chapter 6 and Chapter 7.

- The results of this study provide information to and support the local authority for planning and decision making. → Addressed in Sections 6.3 and 7.3.
- The experiences in the case city could provide a reference to the cities alike. → Addressed in Sections 8.2 and 8.3.
- This study built LHV predictive models that could be used at the regional or global scale. → Addressed in Chapter 5.
- The experiences of building the LHV predictive model using two different methods, regression analysis and artificial neural network (ANN) could give reference to researchers when choosing a method to build an LHV predictive model. → Addressed in Chapter 5.
- This study also provides a way to quantitatively assess the effectiveness waste prevention actions. → Addressed in Chapter 6 and Chapter 7.

1.4. Outline of the Thesis Report

Chapter 1: Introduction

Chapter 1 introduces the context and backgrounds of the present study. Research gaps, aims, objectives and contributions of this study are also presented in this chapter. Besides, the structure of the whole thesis report is outlined in this chapter.

Chapter 2: Literature Review

Chapter 2 presents a review of the literature related to conceptual issues addressed in this thesis. These include the definitions, principle and concept related to MSW management, the technologies utilized for waste treatment and disposal, the methodologies used for evaluating the performance of MSW management. A brief review of the LHV predictive model is also presented in this chapter.

Chapter 3: Methodologies and Materials

System boundaries and MSW management scenarios in the present study are defined in this chapter. Chapter 3 depicts the study city, data and data sources, and methodologies used in this study.

Chapter 4: Waste Management Policies and Progresses

This chapter systematically reviewed the main directives, regulations, policies, strategies and measures concerning MSW management undertaken by EU, England and Nottingham. Progress achieved so far is summarized as well.

Chapter 5: Generalised Models to Predict the LHV of MSW

This chapter presents the LHV predictive models built using multiple regression analyses and ANN. The predictive performance of models were evaluated and compared using appropriately selected indicators. The application of LHV predictive models in MSW management were discussed as well.

Chapter 6: MFA of MSW Management in Nottingham

This chapter depicts and assesses the waste flows and performances of three historical MSW management scenarios and a future MSW management scenario using MFA against appropriately selected indicators. The successes and drawbacks of historical MSW management and the potential improvements are identified in this chapter.

Chapter 7: LCA of MSW Management in Nottingham

This chapter quantifies the global warming potential (GWP) of all MSW management scenarios in Nottingham using LCA based on the results of MFA in Chapter 6 and the LHV predictive model built in Chapter 5.

Chapter 8: Discussions and Conclusions

This chapter summarizes the major findings and results, as well as contributions of this study. MSW management experiences in Nottingham and findings of this study that can be referred by other cities, especially Chinese cities are addressed. The limitations and uncertainties in this research are analysed. Recommendations in future research are proposed.

Chapter 2 Literature Review

There is no optimal MSW management system could universally applied for providing sustainable waste management for all municipalities because of the geographical variations in waste characteristics, available resources, availability of technologies and techniques, economic development, public participation and acceptance, and the market size of secondary products (Mendes et al., 2004; Yay, 2015). Even though, there are basic concepts and principles available for guiding the establishment of sustainable MSW management. There are also a bunch of technologies and techniques that can be integrated into MSW management, and some methodologies are available to assess the performance of the whole management system or a single management option for further improvement. Thus, this chapter gives a review of the aspects related to the MSW management mentioned above.

2.1. Definition of MSW

The definition of MSW has been various. It can be taken to refer to urban solid waste, household waste, residential or domestic solid waste (Buenrostro and Bocco, 2003; Masebinu et al., 2017; Tang and Huang, 2017). The classification of MSW may depend on its sources of generation, or the territorial limits of a municipality responsible for its collection. European Commission defines MSW as waste from households, as well as other waste has similar nature or composition to waste from the households (EC, 1999). However, MSW defined among EU members of states or their municipalities may not be consistent. Indeed, the ambiguity and inconsistency of the definition may affect the way the EU directive is implemented and comparison among municipalities or countries (Buenrostro et al., 2001; Buenrostro and Bocco, 2003; Masebinu et al., 2017).

Based on the definitions of MSW collectively acquired from the literature, as depicted in Figure 2.1a, MSW can in general be defined as the solid waste collected by (or on behalf of) a local authority from all the households and part of the industrial, commercials and institutional entities, so long as the waste produced by these sources is of a similar nature and composition as household waste (Burnley, 2001;Shekdar, 2009; Masebinu et al., 2017). In England, MSW used to be defined based on the waste collection operation rather than its sources or composition (Burnley, 2001; Samiha, 2013). It means that MSW includes household waste and any other wastes collected by a waste collection authority, or its agents, or managed by the waste disposal authority (Figure 2.1b) (NCC, 2010). This definition is still adopted by some authorities in England, for instance, Nottingham City Council. Currently, MSW in England is defined as "regular waste from non-industrial sources, such as residential homes, restaurants, retail centres, and office buildings (DEFRA, 2016b)." Practically, this means that only the commercial or industrial waste having a similar composition to and collected along with household waste is classified as MSW; otherwise this same commercial or industrial waste, if collected by a separate collection mechanism, would be defined as commercial or industrial waste instead of MSW (Burnley, 2001). Separately collected healthcare waste and hazardous waste are normally excluded from the scope of MSW in all definitions.

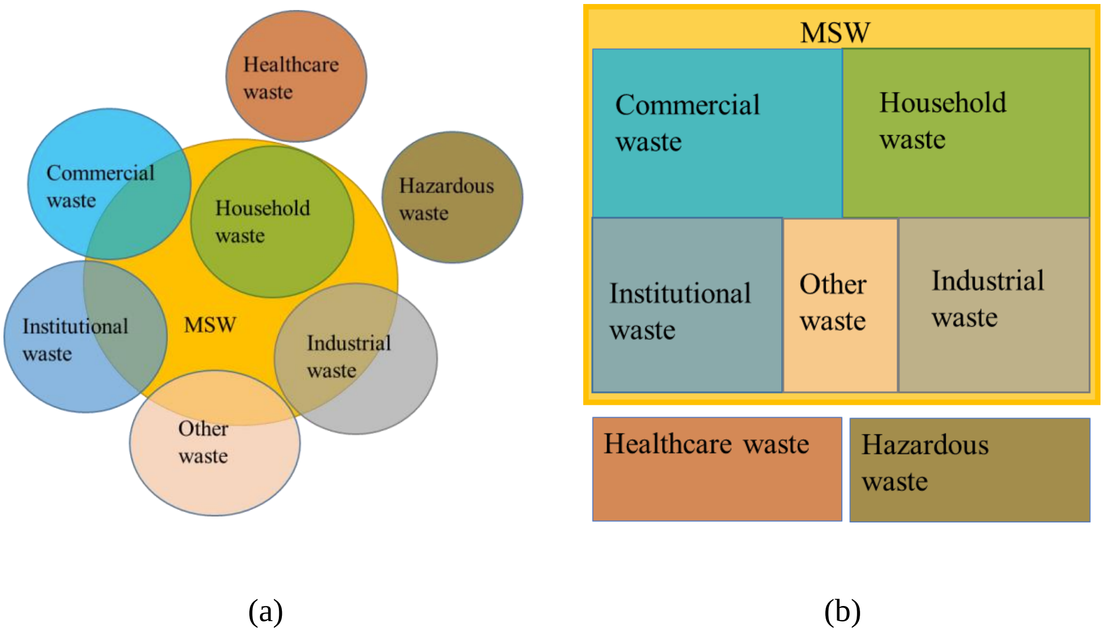


Figure 2.1. Framework of the definitions of MSW (source: Wang, 2020): (a) the broadly adopted definition of MSW; (b) the definition of MSW was adopted by England and currently adopted by Nottingham

2.2. Waste Management Hierarchy

One of the most widely adopted principles guiding sustainable MSW management is the "waste management hierarchy", which places the following strategies in descending order of priority: prevention, recycling, recovery and landfill (Figure 2.2). It was initiated about 40 years ago (Pires and Martinho, 2019), then clearly defined in the Community Strategy for Waste Management in 1989 (EC, 1989; Papargyropoulou et al., 2014) and circulated in the Waste Framework Directive (EU Directive 2008/98/EC) as guiding principle for practicing sustainable waste management in European countries (EC, 2008). In 2015, European Commission emphasized the important role of waste management hierarchy in the Circular Economy Strategy by taking valuable resources back into the economy and leading to the best overall environmental benefits (Minelgaitė and Liobikienė, 2019; Pires and Martinho, 2019).

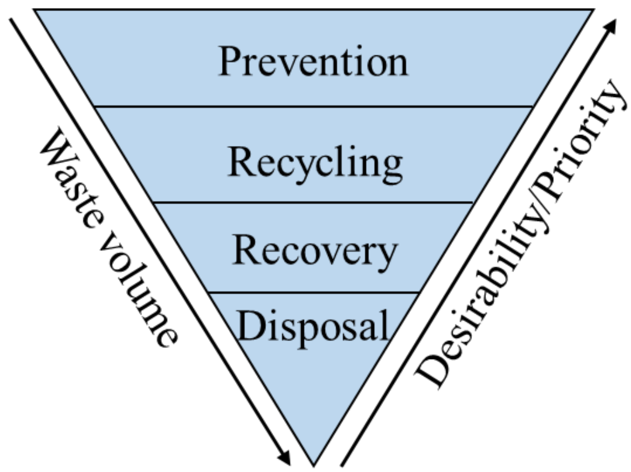


Figure 2.2. The structure of waste management hierarchy (source: Meystre and Silva, 2012).

2.2.1. Prevention

Prevention (or reduction in some studies) is ranked at the top of the waste management hierarchy (Figure 2.2). It is defined as the prevention of waste at sources taken before a substance, material or product has become waste (Zorpas et al., 2014). In 2016, waste prevention was included in the 12th Sustainable Development Goals of the 2030 Agenda for Sustainable Development and has been adopted by the 193 United Nations countries (United Nations, 2016). Waste prevention contributes to both environmental and social benefits. It reduces 1) the quantity of waste; 2) the content of hazardous substances in materials and products; 3) the adverse impacts on human and environment; 4) the energy, materials and labour inputs at the post-consumption states to recycle, to treat and to dispose of waste; and 5) the energy, materials and labour inputs into upstream processes to mine and processing virgin materials (EC, 2008; Tseng et al., 2018). Waste prevention through product reuse even potentially contributes to all three dimensions of sustainability by offering goods to low-income people with affordable prices, creating job opportunities for marginalized people outside the labour market, and reducing environmental impacts through replacing consumption of new products (Zacho and Mosgaard, 2016). Thus, waste prevention plays a key role in sustainable MSW management and should be integrated into any waste management strategy (Zorpas et al., 2014).

Prevention measures can be taken in all life cycle phases of a product (Zacho and Mosgaard, 2016). These measurements include reducing materials usage through appropriate product designing, consuming less by changing life behaviours, reusing products and composting organic waste at home. Based on the environmental performance of prevention actions, their priorities are ranking as Figure 2.3 illustrated. Avoiding and reducing waste generation are principally preferable than reuse because reuse does not reduce the daily household discard and second-hand products do not replace in a 1:1 proportion of the new products (Zacho and Mosgaard, 2016). Home composting reduces the waste entering the downstream treatment and disposal processes, but it produces leachate and greenhouse gas (GHG) emissions when organic waste decomposed, and these leachate and emissions are usually discharged into surrounding environment without any treatment.

The promotion of waste prevention is limited and challenged by many factors. One of them is the complexity of prevention behaviours. Prevention behaviours are complex because most prevention actions are voluntary. Voluntary behaviours are always affected by many internal and external factors including the public concerns for environment (Ferrara and Missios, 2012), altruistic attitudes (Cecere et al., 2014), moral obligations

(Graham-Rowe et al., 2014), inconvenience (Bortoleto et al., 2012), consumerism, availability of skills, tools and knowledges (Barr et al., 2013). Unlike other waste management behaviours such as recycling, prevention behaviour is not provoked by political or social pressure (Zacho and Mosgaard, 2016). Other limitations and challenges come from the perspective of definitions, monitoring methods and indicators, the linkage between policies and stakeholder, and quantifying the effects of waste prevention (Yano and Sakai, 2016).

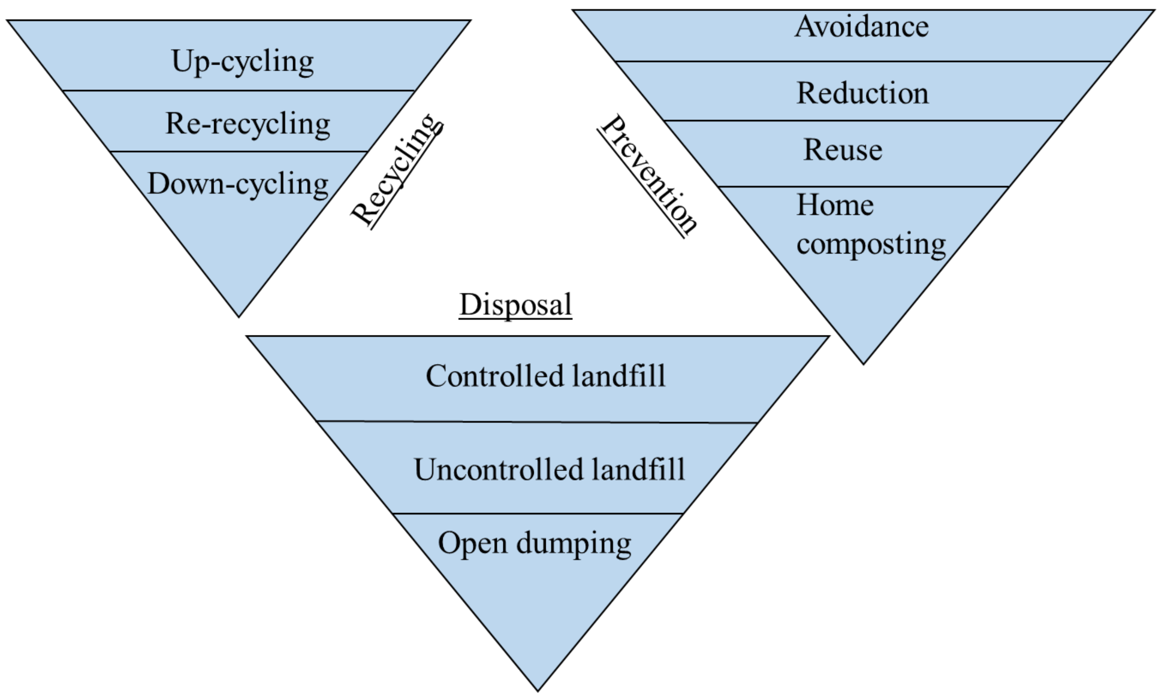


Figure 2.3. Hierarchies of waste management options in the same layer of waste management hierarchy (source: Wang, 2020).

Owing to the complexity of prevention behaviours, monitoring and quantitatively assessing the effectiveness of waste prevention actions is difficult. The evaluation of prevention was often based on the assumption of prevented waste generation, thus the results could not be used as an indicator in the assessment of the effectiveness of prevention measures (Matsuda et al., 2018). Matsuda et al. (2018) evaluated the impact of waste prevention by quantifying GHG emission through three perspectives: relative change from baseline year, absolute change from potential waste generation and the absolute amount of activities. Waste generation per capita is often used as an indicator to estimate the waste prevention in the comparative analysis (Makarichi et al., 2018; Eurostat, 2019).

2.2.2. Recycling

Recycling valuable proportions of waste are the most widely applied strategy to reduce the volume of disposed waste, so as to reduce the adverse impacts of MSW management (Das et al., 2019). For example, recycling e-waste could shift the e-waste waste management sector from a GHG emitter to a saver when compared to landfill (Clarke et al., 2019). It also an effective way to turn waste into resources, hence to facilitate resource conservation and the circular economy by closing the circle of material flow (Pomberger et al., 2017; EC, 2019). Therefore, recycling targets are often set in waste management regulations and plans, and many efforts have been put into recycling programmes.

Recycling is important but not as effective as prevention strategies in achieving sustainability. Material and energy inputs, as well as financial and labour investments, are needed for recycling materials (Gertsakis and Lewis, 2003; Minelgaitė and Liobikienė, 2019; Samiha, 2013). Sometimes, recycling material consumes more energy and/or generates more pollutants than disposal of it. DEFRA (2012) reported that it produced more GHG to recycle food and beverage cartons than to dispose of them in the landfill in the UK. Recycled materials are usually transported long distances to reach their end-user, this will reduce the overall environmental benefits of material recycling (Turner et al., 2016; Cobo et al., 2018). Besides, one MSW management system having a good performance in material recycling does not mean this system performs well as a whole because it might potentially generate more waste per capita, which means it has poor performance in terms of waste prevention (Castillo-Giménez et al., 2019). Minelgaitė and Liobikienė (2019) pointed out that recycling behaviour has a significantly positive correlation with waste generation. Bassi et al. (2017) pointed out there was no clear correlation between the environmental performance of household waste management and the recycling rate. Thus, the sustainability of an MSW management system does not necessarily correlate with the recycling rate.

Furthermore, hazardous substances disperse and accumulate in the recycled materials, and this 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., 2017b). Thus, recycling activates should not only focus on where the waste is routed but also the quality and utilization of the recycled materials (Bassi et al., 2017). As Figure 2.3 depicted, up-cycling is the most preferable options because materials are recycled from source separation, and recycled materials are infinitely recyclable, such as glass and metal (Tonn et al., 2014; Pires and Martinho, 2019). Materials recycled through re-recycling process do not replace the virgin material one-to-one, such as paper/cardboard and packaging waste (Rigamonti et al., 2009; Pires and Martinho, 2019), while materials recycled through down-cycling process has low quality because they are often recycled from and contaminated by mixed/residual waste (Fortelný et al., 2004; Dieterle et al., 2018; Pires and Martinho, 2019).

Particular attention should also be paid to the potential risks of recycling. Recycling put potential hazards on human health because human is exposed to pollutants and toxic compounds during the recycling process (Cobo et al., 2018). Especially in developing countries, labour-intensive informal recycling often occurs. Pickers and scavengers are exposed easily in unsafe and unhealthy working conditions at street or uncontrolled dumpsites. For example, 48 people were killed by a rubbish landslide in Addis Ababa, Ethiopia; these people made livings by scavenging at the landfill site and some even resided at the dumpsite permanently (BBC, 2017b).

2.2.3. Recovery

Recovery normally refers to energy recovery from waste through thermal or biological EfW options such as incineration and anaerobic digestion. Sometimes, it refers to recovering materials from MSW, such as compost from biological treatment. EfW, especially incineration with energy recovery, are increasingly adopted worldwide as a more sustainable waste treatment option over the landfill. EfW has advantages in waste volume reduction, environment protection and energy recovery. Political requirements and energy shortage are also the driving forces for increased application of EfW (Horttanainen et al., 2013). It has been assumed that 50% of the energy content of MSW is renewable and the GHG emission from the incineration of MSW (about 40 kg CO_2-eq./GJ) is much lower than that from coal (98 – 108 kg CO_2-eq./GJ) or peat (97 – 106 kg CO_2-eq./GJ) (Horttanainen et al., 2013). Therefore, recovery energy from waste is not only helping to tackle the problem of energy shortage but also contributes to the mitigation of global change.

However, from the perspective of the circular economy, incineration should be the least preferable option because materials cannot be used again in products (Pires and Martinho, 2019). Therefore, energy recovery should be undertaken after recycling has been done. Besides, thermal EfW facilities are normally more costly, requiring staff having higher skills, and producing more air pollutants than landfills if emission control is not applied (Bogner et al., 2007).

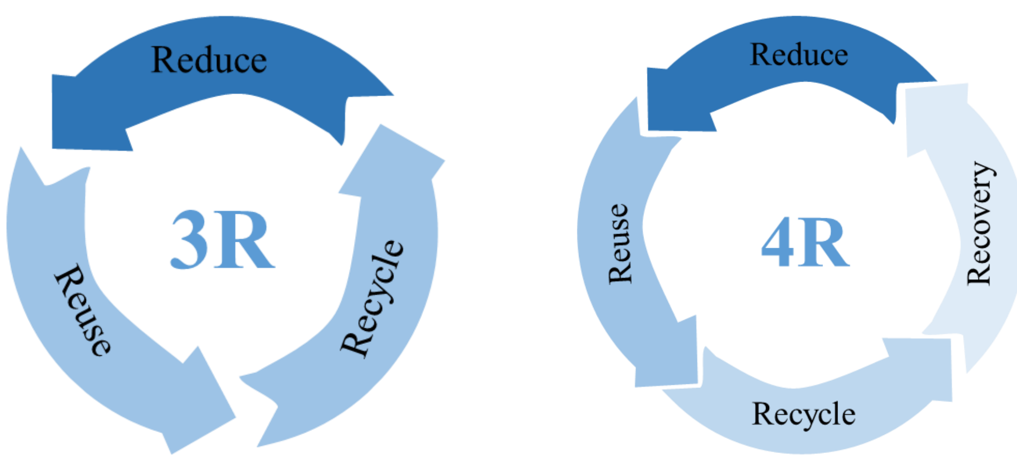
2.2.4. Disposal

Disposal of waste is regarded as the final step of waste management that further process of treatment is not needed (Moeinaddini et al., 2010; Othman et al., 2013). Landfilling or dumping is usually the final disposal of waste (their priorities are listed in Figure 2.3). Among these methods, open dumping is the least preferable one because it has the highest contamination but a big part (33%) of world waste is disposed of in this way (Kaza et al., 2018). Lower-income countries are especially relying on this method to dispose of about 93% of their waste (Kaza et al., 2018).

Even though landfill is not a preferable option for waste management, a sanitary landfill cannot be totally replaced no matter how successful an application of waste management technique is (Othman et al., 2013). This is because all the other subsystems in a waste management system generate a certain amount of residue that cannot be valorised by current technologies (Cobo et al., 2018; Cossu, 2012). Hence, landfill should be one of the vital options whenever environmental and ecological solutions are provided or modelled (Othman et al., 2013). Besides, landfill could be a preferable waste treatment option when landfill gas is recovered, leachate and pollutant emissions are controlled and processed (Khandelwal et al., 2019). Appropriately managed landfills can also work as a pool to conserve valuable materials which cannot be properly separated or recycled by current technologies (Cobo et al., 2018). The stored materials could be mined and reprocessed in the future until the pertinent technologies have been developed (Cobo et al., 2018).

2.3. 3Rs Principle

Another widely applied principle for practicing sustainable MSW management is the "3Rs" principle (Figure 2.4a). 3Rs principle was initiated in 2004 and promoted in countries across Asia, USA and many countries in Europe (Ray, 2008; Sakai et al., 2011; Pandey et al., 2018; Minelgaitė and Liobikienė, 2019). It encompasses the options of reducing, reusing and recycling (Figure 2.4a) (Yoshida et al., 2007; Shekdar, 2009; Sakai et al., 2011). In practical, the 3Rs principle promotes waste prevention ahead of disposal, then its reuse and recycling, and last the optimization of its final disposal. (Ramachandra et al., 2018; Minelgaitė and Liobikienė, 2019). Sometimes, "recovery" is added to and makes up the "4Rs" principle (Figure 2.4b).



(a) 3Rs principle (b) 4Rs principle

Figure 2.4. 3Rs and 4Rs principle for waste management (source: Wang, 2020).

3Rs promotes the sustainable development from three directions via helping public towards sustainable living and encouraging us to make lifestyle decisions to reduce the waste we create and reduce the impact on the environment (Sakai et al., 2011; Samiha, 2013). The first direction is to reduce and prevent the environmental impact and cost from final disposal in landfills, as well as to save land space, this is especially important for the countries where disposal sites are difficult to obtain (Sakai et al., 2011; Samiha, 2013). In the second direction, 3Rs save natural resources, thus help to secure resource demands. This is important for the countries experiencing rapid economic growth, such as India and China (Sakai et al., 2011; Samiha, 2013). In the third direction, developing an integrated policy centred on the 3Rs principle helps to achieve the collaboration of waste management, protection of natural resources, and reduction of GHG emission (Sakai et al., 2011).

2.4. Integrated MSW management

MSW management is complex because of the complex characteristic of MSW and the complex interaction of stakeholders in the processes of waste management. Instead of treating MSW as a homogenous mass, academia argued that MSW consists of different materials that should be treated differently: some should be prevented, some should be reused, some should be recycled or composted, some should be combusted and others buried (Gertsakis and Lewis, 2003). Therefore, an effective and sustainable MSW management system should integrate multiple management options such as reducing, reusing, recycling, recovering, and finally incinerating and/or landfilling (Othman et al., 2013). The selection of treatment technology and technique for various waste streams should base on their physical and chemical characteristics (Papargyropoulou et al., 2014).

Integrated MSW management is regarded as the best solution towards sustainability that goes beyond the safe disposal of waste (Yay, 2015). It provides an answer to transferring a society or an industry to a circular economy by linking waste treatment and resource processing (Cobo et al., 2018). An integrated MSW management is expected to produce not only materials through materials recycling and recovery but also energy and nutrient through EfW and composting or anaerobic digestion. In this way, the environmental impacts of MSW management are expected to be mitigated. For example, the production of per tonne of secondary aluminium from scrap instead of produced from the mineral ore could avoid the emission of greenhouse gas up to 19 tonnes of equivalent CO_2 (Damgaard et al., 2009).

Integrated MSW management encompasses the functions of waste generation, storage, collection, transfer and transport, processing, recycling, recovery and disposal in an environmentally friendly manner, and provides support to economic productivity in the light of the best principles of public health, economics, engineering, techniques, technologies, conservation, aesthetics, public attitude and other environmental considerations (Tanskanen, 2000; Muller and Hoffman, 2001; Henry et al., 2006; Uyarra and Gee, 2013; Ramachandra et al., 2018). Thus, the waste flows and streams within an integrated MSW management system are highly complex (Yay, 2015). Taking Greater Manchester, the UK, as an example, the MSW management in the city integrates multiple technologies and strategies from collection to final disposal. These strategies and technologies include types of source separation such as kerbside collection and collection at household waste recycling centre, material recovery facility, mechanical biological treatment, in-vessel composting and a series of EfW plants such as anaerobic digestion and thermal recovery facilities (GMCA, 2018). To realize these complex processes in integrated MSW management, the participation and cooperation of all stakeholders are required. Stakeholders incorporated in MSW management includes households, governments, companies, enterprise, city planners and all others who are engaged in various MSW management activities (Joseph, 2006; Caniato et al., 2014). Identification of these stakeholders and their interests and characteristics is difficult but important in involving them in various waste management activities (Joseph, 2006; Caniato et al., 2014).

2.5. Waste Collection

Waste collection is the first essential step of MSW management through which mixed or segregated wastes enter the downstream management processes. In most municipalities, local governments are responsible for MSW collection, but collection by the private sector, informal waste picking or scavenging activities usually happens, especially in developing countries (Manaf et al., 2009). The informal collection activities benefit the poor in the cities and contribute to the mitigation of environmental impacts via recycling valuable materials (Kassim and Ali, 2006; Botello-Álvarez et al., 2018).

MSW used to be collected in the mix in European countries and is still collected as a mixture worldwide. However, mixed waste obstacles the utilization of resources from waste thus reduces urban sustainability. The mixture reduces the quantity and quality of materials that can be recycled or composted because recyclable and compostable materials are contaminated in the collection process and no longer suitable for recycling or composting even if they can be sorted after their collection (Kumar and Samadder, 2017; Rumyantseva et al., 2018). Besides, The LHV of the mixed waste is reduced by moisturized components such as food waste (Zhang et al., 2010; Bai et al., 2012). The low LHV of MSW reduces combustion efficiency and increases the potential of pollutant production (McKay, 2002; Tsai and Chou, 2006). Furthermore, the mixed collection increases the costs, energy and labour inputs, as well as environmental impacts, at the downstream processes to sort, treat and to dispose of the mixture (Rumyantseva et al., 2018).

Therefore, the separate collection is the key to improve the sustainability of MSW management by improving the recycling potential and reducing energy inputs. The separate collection also modifies the residual MSW that can become similar to RDF (Rada, 2013). The improved residual MSW opens direct co-combustion opportunities in existing industrial plants, and thus decreases the cost of solutions alternative to direct combustion in dedicated incineration plants (Rada, 2013).

By realizing the importance of separate collection in sustainable MSW management, regulations, policies and strategies have been put into enforcement to make source separation an obligation for local governments. Kerbside collection is often adopted to separate waste at source in European countries. It has been demonstrated to be the most efficient and sustainable separate collection scheme (Tucker et al., 1998; Larsen et al., 2010). It improves the quantity and quality of materials left to recycling (Dahlén et al., 2007; Gallardo et al., 2010), and reduces the cost of MSW management (Cimpan et al., 2015).

However, effective source separation always cannot be guaranteed (Fei et al., 2018). The success of separate collection (kerbside collection) lies mostly on citizens' participation (Gallardo et al., 2010), which is affected by both intrinsic personal factors and external incentives such as social and political instruments. Knowledge, skills, the belief of consequences and capability, action planning, role clarification, feedback, moral values and motivation are the intrinsic personal factors affecting public participation (Ghani et al., 2013; Dai et al., 2015; Xiao et al., 2017). Therefore, improving public awareness through public education is essential to enhance separate collection (Ibáñez-Forés et al., 2018). Accessible supportive facilities, available information and knowledge on waste separation, good situational factors such as storage convenience, suitable distances to containers and collection schedule also encourage public's participation in the separate collection (Ghani et al., 2013; Gallardo et al., 2010; Bernad-Beltrán et al., 2014). Therefore, local authority plays an important role in enhancing public involvement in source separation by appropriately designing the collection system, providing sufficient investments and facilities, and conducting public education campaigns.

2.6. Waste Management Techniques and Technologies

Waste management technologies can be classified into three categories based on their purposes: i) pre-treatment of feedstocks for recycling, recovering or final disposal; ii) recovering energy from waste in any form; iii) disposal of residues. Reprocessing of secondary materials is not included in the scope of waste management because these materials already enter the production process.

2.6.1. Pre-treatment technologies

As effective source separation always cannot be guaranteed, especially in developing country (Fei et al., 2018), waste mixture makes it difficult to ensure the quality of recycled materials and by-products derived from waste such as compost, and the stable operating conditions of waste treatment such as anaerobic digestion and combustion (Fei et al., 2018).

It also increases the possibility of pollutant production from incinerators and landfills (Kumar and Samadder, 2017; Browne et al., 2014). Thus, pre-treatment was introduced as a complementary of insufficient source separation to reduce the potential impacts of MSW management. It is also a possible solution for the cities to improve their recycling rate (Cimpan et al., 2015; Trulli et al., 2018), and saves fuel input in the processes of transportation due to matter reduction (Cimpan and Wenzel, 2013). Therefore, pre-treatment facilities play an important role in integrated MSW management (Cimpan et al., 2015).

Popular technologies used for waste pre-treatment are mechanical biological treatment (MBT) and material recovery facility (MRF). Initially, MBT was developed for separating and stabilizing the quickly biodegradable fraction to reduce the adverse impacts from landfills, while MRF was developed for materials recycling and recovery. Nowadays, many processes and functions of MBT and MRF overlap (Table 2.1).

Table 2.1. Brief summary of MBT and MRF (source: Wang, 2020)

	MBT	MRF
Processes	Manual sorting, mechanical sorting and stabilization	Mechanical sorting and manual sorting
Outputs	Recyclable materials, RDF (possibly), compost like products and residual waste to landfill.	Recyclable materials, RDF and residual waste to landfill.
Volume reduction	Yes	Yes

2.6.1.1. MBT

MBT was proposed to reduce waste impacts, i.e. leachate, biogas and odour, from landfill (Bayard et al., 2010; Scaglia et al., 2013), and has been widely applied in Europe and several regions around the world (Fei et al., 2018). After treated in MBT, the impact of landfilled waste can be significantly reduced by up to 30% because organic matter is stabilized (Scaglia et al., 2013; Bayard et al., 2010; Trulli et al., 2018). MBT also induces positive environmental externalities including reduced human toxicity and eco-toxicity, recovered materials and enhanced landfill efficacy such as the positive modification of leachate and landfill gas (Beylot et al., 2015; Fei et al., 2018).

The MBT consists of mechanical pre-treatment of MSW, followed by the biological treatment process (Figure 2.5). In particular, MBT involves metal separation systems for steel and aluminium recovery and mechanical screening to obtain two fractions (Norbu et al., 2005; Di Maria et al., 2012; Scaglia et al., 2013). The upper-grid fraction, which consists of a lighter fraction with higher LHV, mainly plastic and paper, is used as RDF or is landfilled without further treatment (Norbu et al., 2005; Di Maria et al., 2012; Scaglia et al., 2013). The lower-grid fraction is biologically treated to reduce its reactivity and leachate pollutant prior to landfill disposal (Norbu et al., 2005; Di Maria et al., 2012; Scaglia et al., 2013). The biological treatment section consists of a stirred, continuous flow, aerobic stabilization/composting facility (Di Maria et al., 2012).

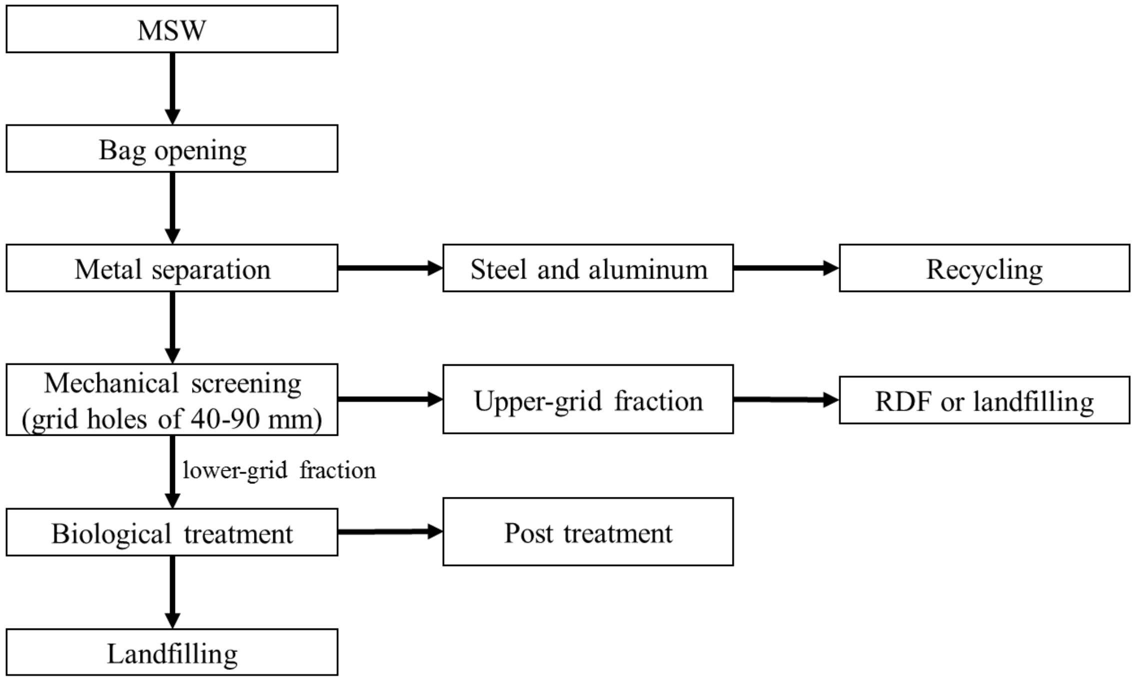


Figure 2.5. Processes to MSW in MBT (source: Trulli et al., 2018)

Two types of MBT technology are available for pre-treatment: i) mechanical-biological pre-treatment, where the organic fraction is separated and biologically stabilized prior to landfilling, recyclables and RDF are recovered from the residues, and ii) mechanical-biological stabilization or bio-drying, where composting is first applied for drying waste prior to extraction of a larger RDF fraction. Mechanical-biological pre-treatment is aiming to stabilize the organic for minimum gas and leachate emissions from the landfill while mechanical-biological stabilization aims at maximizing the materials and RDF recovery (Montejo et al., 2013).

MBT is a promising supplementary to insufficient source separation of MSW but is has weak economic performance and risk of environmental pollution (Sadhukhan et al., 2016; Fei et al., 2018). MBT's economic and environmental performance is largely influenced by the upstream separation and collection of waste and downstream management of residuals and sorted fractions from a life-cycle perspective (Trois and Simelane, 2010; Beylot et al., 2015). MBT consists of a wide variety of treatment process and has many types of output which require further effective utilization or additional disposal otherwise, there is a risk of further environmental pollution (Fei et al., 2018). In terms of economic performance, the revenue from resource recovery and product generation is a possible way to improve the economic margin of MBT. Sadhukhan et al. (2016) reported that the revenue from the by-products such as metal, energy and fertilizer can cover the waste collection fees, annual capital and operating cost.

2.6.1.2. MRF

MRF is usually a system recovering recyclable materials from multi-stream waste through mechanical and manual separation processes (Ip et al., 2018). There are two types of MRFs: single-stream commingled materials MRF and residual waste MRF (dirty MRFs or mixed MSW MRFs in North America). The former consists of a series of mechanical separation activities designed to recover recyclables from separately collected recyclable materials (Turner et al., 2016). Similar to MBT, residual waste MRF are installed to recycle materials from and stabilize residual waste (Cimpan et al., 2015).

Material separation carried out in MRFs is general mechanical separation based on the physical characteristics of waste streams; for example, screening separates waste based on its size, ballistic separator distinguishes between flat, light items and rigid, heavy items based on their particle elasticity and aerodynamic properties (Ip et al., 2018). Five main technical sections in MRF are 1) feeding and pre-conditioning, 2) conditioning waste for sorting, 3) sorting materials, 4) refining material outputs quality to market requirements, and 5) preparing and delivering products and residues to downstream processors (Cimpan et al., 2016). The latest MRF also contains bio-cells for composting organic waste and the production of RDF (Ardolino et al., 2017). The introduction of an aerobic treatment stabilizes biodegradable waste and reduces the volume of waste to be buried, thus reduces the land space required and GHG emission from landfills.

However, MRFs are facing financial challenges and challenges from the variable composition of the waste streams received by the MRFs (Gu et al., 2017; Ragaert et al., 2017; Ip et al., 2018). The revenues from sales of recovered materials cannot cover the operational and disposal costs due to the volatile market prices of secondary materials and changing buyer requirements (Gu et al., 2017; Ragaert et al., 2017; Ip et al., 2018; Kang and Schoenung, 2006; Cimpan et al., 2016). Besides, the quality of some recovered materials cannot meet the required purity (Cimpan et al., 2015). The composition of MSW is unstable and becoming ever diversified, it is difficult to appropriately separate all materials apart. Equipment and techniques need to be updated often to meet the separation requirements.

2.6.2. Energy from waste (EfW)

EfW becomes increasingly popular in managing MSW and has been greatly modernized and prioritized especially in developed countries (Kumar and Samadder, 2017). It is an essential part of integrated MSW management (Brunner and Rechberger, 2015), and an option for renewable energy supply (Levidow and Upham, 2017). EfW facilities recover energy from waste in the form of useable heat, electricity or fuel such as biogas and RDF (Zhao et al., 2016b). Available EfW technologies can be divided into three categories: 1) thermal conversion, 2) biological conversion, and 3) landfilling with gas recovery (Kumar and Samadder, 2017).

2.6.2.1. Thermal conversion technologies

Thermal conversion technologies include incineration, pyrolysis, and gasification. The characteristics of thermal treatments are listed in Table 2.2. They are generally useful for dry waste (Kumar and Samadder, 2017). Incineration is the most widely applied thermal conversion technology (Baran et al., 2016), but its initial purpose is hygiene and volume reduction. Until now, most incineration plants are not equipped with energy recovery facilities. Among the 1900 waste incineration plants in Japan, only 190 equipped with power generation facilities and 102 equipped with electricity generation facilities (Kumar and Samadder, 2017). In 2014, only 13 of the 60 incineration plants in England recovered energy from waste and only 4 of them dedicated to recovering energy from MSW (DEFRA, 2018a). Pyrolysis and gasification are still in the research phase and they are not feasible for commercial purposes at a large scale (Kumar and Samadder, 2017).

Table 2.2. Main characteristics of thermal treatments of solid waste (source: Arena, 2012).

	Incineration	Gasification	Pyrolysis
Aim of the process	Volume reduction, and maximizing waste conversion to high temperature flue gases, mainly CO_2 and H_2O.	To maximize waste conversion to high heating value fuel gases, mainly CO, H_2 and CH_4.	To maximize thermal decomposition of solid waste to gases and condensed phase.
Operation conditions			
Reaction environment	Oxidizing (oxidant amount larger then that required by stoichiometric combustion)	Reducing (oxidant amount lower than that required by stoichiometric combustion)	Total absence of any oxidant
Reactant gas	Air	Air, pure oxygen, oxygen-enriched air, steam	None
Temperature	Between 850 ℃ and 1200 ℃	Between 550 ℃ and 900 ℃ (in air gasification) and 1000 - 1600 ℃	Between 500 ℃ and 800 ℃
Pressure	Generally atmospheric	Generally atmospheric	Slight over-pressure
Process output			
Produced gases	CO_2, H_2O	CO, H_2, H_2O, CO_2, CH_4	CO, H_2, CH_4 and other hydrocarbons
Pollutants	SO_2, NO_x, HCl, PCDD/Fs, particulate	H_2S, HCl, COS, NH3, HCN, tar, alkali, particulate	H_2S, HCl, NH_3, HCN, tar, particulate
Ash	Bottom ash is treated to recover ferrous and non-ferrous metal and inert materials (to be utilized as a sustainable building material). Fly ash is generally treated and disposed in hazardous landfill.	Bottom ash are often produced as vitreous slag that can be utilized as backfilling material for road construction	Treated and deposed as industrial special waste
Gas cleaning	Treated in air pollution control units to meet the emission limits and then sent to the stack	It is possible to clean the syngas to meet the standards of chemicals production processes or those of high efficiency energy conversion devices	It is possible to clean the syngas to meet the standards of chemicals production processes or those of high efficiency energy conversion devices

Thermal treatments can vastly reduce the volume of waste and require much less land than landfill facilities for handling the same quantity of waste (Kumar and Samadder, 2017), but they require high operational and maintenance costs (Yay, 2015). Incineration might have high potentials for air pollutants production, especially in developing countries where the air pollution control system in the incineration plant is usually unavailable. Pollutants could cause health risks. Children who lives near the MSW incineration plant may suffer genetic and epigenetic modification, such as DNA damage or global DNA hypomethylation due to the MWI-emitted PCDD/Fs and other contaminants (Xu et al., 2019). Besides, in the cities where energy is generated from renewable sources, waste incineration with energy recovery does not perform better than sanitary landfilling with gas valorisation, due to the low impact of avoided electricity production (Lima et al., 2019).

2.6.2.2. Biological conversion technology

Biodegradable organic waste consists of a big fraction in MSW (44% at the globe) (Kaza et al., 2018). The inherent characteristic that is low LHV and high moisture content, of biodegradable organics makes it unsuitable for thermal treatment and best for biological treatment (Panigrahi and Dubey, 2019). Biological treatments have advantages over landfilling for volume reduction, stabilization of biodegradable organics, and production of compost. The often mentioned biological treatment technologies include composting and anaerobic digestion (Aghbashlo et al., 2019). Composting is the aerobic oxidation of organic waste while anaerobic digestion is the decomposition of organic matter under oxygen-free conditions (Jain et al., 2015; Panigrahi and Dubey, 2019).

Anaerobic digestion has gained increased attention because of its better environmental and economic performances over composting (Aghbashlo et al., 2019; Panigrahi and

Dubey, 2019). It is a reliable and attractive option to convert the complex organic materials into a clean and renewable fuel which can be directly converted into heat, electricity, and mechanical work (Panigrahi and Dubey, 2019; Aghbashlo et al., 2019). On the other hand, anaerobic digestion requires a high quality of feedstock, so as to ensure the stable operation condition and the quality of compost from it, in case of introducing heavy metals or other undesirable materials into the human food chain (Kumar and Samadder, 2017; Panigrahi and Dubey, 2019). But the quality of feedstock always cannot be guaranteed either because of the unsatisfied source separation or the contamination during the pre-treatment process in MRF (Kumar and Samadder, 2017).

Anaerobic digestion has not been widely applied in MSW management at a commercial scale. The challenges obstacle its current application include techniques issues like difficulty in starting-up the process, the lack of effective digester for handling high solid content feedstock and undeveloped hybrid system for efficient utilisation of both digested sludge and effluent; inherent characteristic of the anaerobic digestion like long stabilization period and process inhibition due to the formation of toxic and inhibitory compounds during the reaction process; and other issues such as high capital cost, high energy consumption, large amount of chemicals requirement and the undesirable separation (Panigrahi and Dubey, 2019).

2.6.2.3. Landfill with landfill gas utilization

Landfill normally costs less than other waste management options, this makes it an affordable and the most widely adopted solution (Bogner et al., 2007). However, landfill gas generated from the biological and chemical decomposition of landfilled biodegradable waste is one of the major sources of anthropogenic methane emission because of the uncontrolled landfill and the lack of landfill gas capture infrastructure aside landfills (Kumar and Samadder, 2017; Kaza et al., 2018). Landfill gas consists mainly of CH_4 (50-60%) and CO_2 (40-50%), and traces of other non-methane organic carbon including air pollutants and volatile acid (Ayodele et al., 2019). Landfill gas has a great influence on climate change if it emits directly to the atmosphere because of the high global warming potential (GWP) of methane. It is 28-36 times greater than that of CO_2 in a 100 years horizon (EPA, 2019). On the other hand, the high energy content of methane makes landfill gas a valuable fuel that can be utilized for electricity and/or heat generation if properly harnessed (Ayodele et al., 2019). Therefore, recovering energy from landfill gas is necessary not only for GHG emission reduction but also serve as a valuable renewable energy source (Ayodele et al., 2019).

Since methane is the key gas that determines the energy recovery ability of a landfill, an appropriate estimation of methane generation is necessary to utilize the energy from it. There are various models available for predicting methane emissions from landfills. Kumar and Samadder (2017) summarized the most widely used models. But these models were used for estimating the methane emission in a specific year. Rates of waste degradation and methane generation over time are taken into account. Methane emission estimated by these models is suitable for planning and operation of a landfill and associated energy recovery facility, but not convenient for estimating the total methane emission when comparing the environmental sustainability of various MSW management strategies or evaluating the performance of landfill in its whole life span. For this reason, the model reported by Fong et al. (2015), which was built based on waste composition, could be used to estimate the total amount of methane emitted from landfills.

2.7. Predicting the Heating Value of MSW

Designing and operating an MSW-burn incinerator highly depend on the heating value of MSW which used to be expressed as higher heating value (HHV) or lower heating value (LHV) (Chang et al., 2007). Heating value of MSW can be measured either using a bomb calorimeter or predicted using empirical models based on ultimate analysis, proximate analysis or physical composition analysis (Liu et al., 1996; Chang et al., 2007; Lin et al., 2013; Lin et al., 2015; Boumanchar et al., 2018; Ibikunle et al., 2018; Drudi et al., 2019). Models built to predict the heating value of MSW are summarized in Table 2.3. Models based on ultimate analysis are the equations modified from Dulong's model (Shi et al., 2016; Boumanchar et al., 2018) in which the elemental components such as carbon, oxygen, hydrogen and sulfur as well as moisture content are used as input variables (Khan and Abu-Ghararah, 1991; Chang et al., 2007). Models based on proximate analysis use volatile combustible matter, moisture content, fixed carbon and ash in MSW (Liu et al., 1996; Akkaya and Demir, 2009; Özyuğuran and Yaman, 2017). Models based on physical composition estimate the percentage of physical components in MSW, such as paper, plastics and food, as input variables compositions (Liu et al., 1996; Chang et al., 2007; Drudi et al., 2019).

Table 2.3. LHV predictive models (source: Wang, 2020).

NO.	Reference	Equation	Unit
Models based on ultimate analysis			
Eq. 2.1	Khan and Abu-Ghararah (1991)	$LHV = 145.4C + 620(H - O/8) + 41S$	Btu/lb
Eq. 2.2	Liu et al. (1996)	$LHV = 81C + 342.5(H - O/8) + 22.5S - 6(9H + Wa)$	kcal/kg
Eq. 2.3	Liu et al. (1996)	$LHV = 19.96C + 44.30O - 671.82S - 19.92Wa + 1558.80$	kcal/kg
Eq. 2.4	Channiwala and Parikh (2002)	$HHV = 0.3491C + 1.1783H + 0.1005S - 0.1034O - 0.0151N - 0.0211A$	MJ/kg
Eq. 2.5	Kathiravale et al. (2003)	$HHV_d = 416.638C - 570.017H + 259.031O + 598.955N - 5829.078$	kJ/kg
Eq. 2.6	Chang et al. (2007)	$LHV_d = -7.91 + 170.50C + 320.30(H - O/8 - Cl/35.5) + 45.91S$	Btu/lb
Eq. 2.7	Khuriati et al. (2017)	$HHV = 5751.94 + 52.67C + 75.9H - 4.14N - 1044.03S - 97.68O$	kcal/kg
Eq. 2.8	Khuriati et al. (2017)	$HHV = -2762.68 + 114.63C + 310.55H$	kcal/kg
Eq. 2.9	Ibikunle et al. (2018)	$HHV = 1.3849 + 85.0807C - 28.9675H - 666.125N + 11.6296S - 97.68O$	MJ/kg
Models based on proximate analysis			
Eq. 2.10	Liu et al. (1996)	$LHV = 45V + 6Wa$	kcal/kg
Eq. 2.11	Liu et al. (1996)	$LHV = 44.75V + 5.85Wa + 21.2$	kcal/kg
Eq. 2.12	Kathiravale et al. (2003)	$HHV_d = 356.248V - 6998.497$	kJ/kg
Eq. 2.13	Kathiravale et al. (2003)	$HHV_d = 356.047V - 118.035FC - 5600.613$	kJ/kg
Eq. 2.14	Ibikunle et al. (2018)	$HHV = -7.19477 + 0.116768FC - 0.34728Wa + 0.151701V$	MJ/kg
Models based on physical composition analysis			
Eq. 2.15	Khan and Abu-Ghararah (1991)	$LHV = 23(Fo + 3.6Pa) + 160Pa$	Btu/lb
Eq. 2.16	Liu et al. (1996)	$LHV_d = 88.2\, Pl + 40.5(Ga + Pa) - 6Wa$	kcal/kg
Eq. 2.17	Liu et al. (1996)	$LHV_d = 28.16\, Pl + 7.90Pa + 4.87Ga - 37.28Wa + 2229.91$	kcal/kg
Eq. 2.18	Abu-Qudais and Abu-Qdais (2000)	$LHV_w = 267.0(Pl/Pa) + 2285.7$	kcal/kg
Eq. 2.19	Kathiravale et al. (2003)	$LHV_W = 112.157Ga + 183.386Pa + 288.737Pl + 5064.701$	kJ/kg
Eq. 2.20	Kathiravale et al. (2003)	$LHV_W = 81.209Ga + 285.035Pl + 8724.209$	kJ/kg

Continue of Table 2.3

NO.	Reference	Equation	Unit
Eq. 2.21	Kathiravale et al. (2003)	$LHV_W = 112.815Ga + 184.366Pa + 298.343Pl - 1.920Wa + 5130.380$	kJ/kg
Eq. 2.22	Chang et al. (2007)	$LHV_d = (38.52Pa + 92.09Pl + 49.24Te + 38.34Wo + 37.55Fo + 64.07Mi)[(100 - Wa)/Wa) - 6Wa$	kcal/kg
Eq. 2.23	Chang et al. (2007)	$LHV_d = (35.19Pa + 71.17Pl + 36.24Te + 48.06Wo + 42.21Fo + 44Mi)[(100 - Wa)/Wa) - 6Wa$	kcal/kg
Eq. 2.24	Chang et al. (2007)	$LHV_d = (39.04Pa + 101.47Pl + 38.47Fo)[(100 - Wa)/Wa) - 6Wa$	kcal/kg
Eq. 2.25	Lin et al. (2013)	$LHV_d = (47.3Pa + 58.6Pl + 38.6Te + 32.4Wo + 45.2Fo + 62.3Ru + 50.1Mi)[(100 - Wa)/Wa) - 6Wa$	kcal/kg
Eq. 2.26	Lin et al. (2013)	$LHV_w = 22.1Pa + 28.1Pl + 24.6Te + 12.7Wo + 6.0Fo + 57.4Ru + 17.2Mi$	kcal/kg
Eq. 2.27	Lin et al. (2015)	$LHV_W = 219Pl + 112Pa + 108Wo + 115Te$	kJ/kg
Eq. 2.28	Lin et al. (2015)	$LHV_W = 219Pl + 109(Pa + Wo + Te)$	kJ/kg
Eq. 2.29	Ozveren (2016)	$LHV_W = 20Fo + 83Pl + 187Pa + 105Wo + 170Te$	kJ/kg
Eq. 2.30	Nwankwo and Amah (2016)	$HHV = 17712.04Wo^{-0.0094}Fo^{-0.0063}Le^{0.041}Mi^{-0.019}Pa^{-0.044}Pl^{0.084}Te^{0.025}$	kJ/kg
Eq. 2.31	Nwankwo and Amah (2016)	$HHV = 22402 - 25.677Fo + 122.132Le - 56.697Mi - 104.471Pa + 49.728Pl + 4.442Te - 64.129Wa$	kJ/kg
Eq. 2.32	Oumarou et al. (2016)	$HHV_d = 1.0325 - 0.0011Wo + 0.2254Gr - 0.0046Pa - 0.0068L + 0.3184Fo - 0.0119Pl - 0.0053Me + 0.1099Gl$	MJ/kg
Eq. 2.33	Drudi et al. (2017)	$LHV_W = (13.69Or + 20.94Sa + 37.99Pl + 10.48Pa + 19.27Te)(1 - Wa) - (2.442 - Wa)$	MJ/kg
Eq. 2.34	Drudi et al. (2019)	$LHV_W = (15.42Or + 19.14Sa + 32.68Pl + 8.33Pa + 21.51Te)(1 - Wa) - (2.442 - Wa)$	MJ/kg

Note: *LHV* = Lower heating value; LHV_d = Lower heating value at dry basis; LHV_w = Lower heating value at wet basis; HHV = Higher heating value; HHV_d = Higher heating value at dry basis; *C* = Carbon, percentage by weight; *H* = Hydrogen, percentage by weight; *O* = Oxygen, percentage by weight; *S* = Sulfur, percentage by weight; *Cl* = Chlorine, percentage by weight; *A* = Ash content, percentage by weight; *FC* = Fixed carbon, percentage by weight; *V* = Volatile combustible matter, percentage by weight; *Wa* = Water (moisture content), percentage by weight at dry basis; *Pl* = Plastics, percentage by weight; *L* = Leaves, percentage by weight; *Le* = Leather, percentage by weight; *Pa* = Paper and cardboard, percentage by weight; *Wo* = Wood, percentage by weight; *Gr* = Grass, percentage by weight; *Ga* = Garbage, percentage by weight; *Gl* = Glass, percentage by weight; *Fo* = Food, percentage by weight; *Te* = Textile, percentage by weight; *Me* = Metal, percentage by weight; *Mi* = Miscellaneous components, percentage by weight; *Ru* = Rubber and leather, percentage by weight; *Or* = Organic waste, percentage by weight; *Sa* = Sanitary waste, percentage by weight. The percentage value is within the range of 0 – 100.

Multiple regression analysis is the commonly used method to build LHV predictive models. The majority of the models in Table 2.3 were built using multiple regression analysis. An emerging method artificial neural network (ANN) attracts researchers' attention and has been applied to estimate the heating value of MSW. Dong et al. (2003) applied a feed-forward neural network (FFNN) in predicting the LHV of MSW using 108 samples collected in Nanjing. The results indicated that FFNN was better in predicting LHV of MSW than models built using regression analysis. The application of ANN in Abuja also demonstrated the efficiency and accuracy of this tool in estimating the energy content of MSW (Ogwueleka and Ogwueleka, 2010).

2.8. Evaluation of MSW Management

To achieve the EU, national and local targets, or to establish a sustainable MSW management, it is essential to evaluate the performance or sustainability of an MSW management system. It can be evaluated by using the methodologies of MFA and LCA against a series of representative indicators (Zaccariello et al., 2015; Parkes et al., 2015; Masebinu et al., 2017; Coelho and Lange, 2018). Indicators can be useful in measuring and tracking the performance of waste management practices on a regular basis in a coherent and articulate manner (Wilson et al., 2012; Greene and Tonjes, 2014). Indicators have been utilized in studies to analyse and evaluate the environmental, economic and/or social performance and treatment efficiency of either an single processes (Rotter et al., 2004; Wen et al., 2015; Wen et al., 2009; Bertanza et al., 2018) or the entire waste management system (Desmond, 2006; Greene and Tonjes, 2014; Zaccariello et al., 2015; Castillo-Giménez et al., 2019; Ibáñez-Forés et al., 2019).

2.8.1. MFA

MFA analyses the flux of materials used and transformed as the flow goes through a defined space, a single process or a combination of processes within a certain period. The performing of MFA is based on the first law of the thermodynamics entailing conservation of matter and energy (Belevi, 2002; Rotter et al., 2004; Makarichi et al., 2018). It is expressed as: mass in = mass out + stocks (Brunner and Rechberger, 2016). Through material balance, the sources, flows, accumulations and changes of wastes, as well as their associated environmental loadings, become visible (Brunner and Rechberger, 2016). This develops a common discussion basis for stakeholders from different sectors (Allesch and Brunner, 2015). An MFA also identifies minor changes but might have long-term damage and the most promising processes and flows for improvements (Brunner and Rechberger, 2016). This is extremely important for planning and decision making.

MFA has been widely applied in evaluating the MSW management or supporting decision making (Turner et al., 2016; Masebinu et al., 2017; Zaccariello et al., 2015; Makarichi et al., 2018; Nakem et al., 2016). It has been used in the studies for performance evaluation, system description for further evaluation, comparison of MSW management systems, and scenario analysis by analysing the pathways of waste streams (Allesch and Brunner, 2015; Zaccariello et al., 2015; Turner et al., 2016; Makarichi et al., 2018). MFA identifies the sources of MSW generation, sources for minimizing waste, internal material flows, potential opportunities for reuse and recycling, and accounting for hidden flows and sinks that may be unexplainable in a more traditional method of MSW analysis (Masebinu et al., 2017). Taking the hidden flows and sinks into account, it provides an approach to thoroughly understand the elements and processes of a MSW management system, to identify opportunities for improvement (Owens et al., 2011; dos Muchangos et al., 2016; Zaccariello et al., 2015; Makarichi et al., 2018), and to select the most promising strategy to do so (Dahlén et al., 2009; dos Muchangos et al., 2016; Zaccariello et al., 2015).

MFA is a robust, transparent, and useful tool in measuring the performance of an MSW management system especially when efficient data is available and the reliability of data is high (Zaccariello et al., 2015; dos Muchangos et al., 2016; Makarichi et al., 2018). Complemented by indicators, MFA can be a useful tool to measure the achievements of an MSW management strategy, to distribute a waste stream to the best-suited management option and to plan and design new treatment facilities (Brunner and Rechberger, 2016). Indicators compliance with MFA are summarized in Table 2.4.

MFA alone cannot sufficiently and comprehensively assess or support an MSW management strategy in view of certain goals, such as protection of human health or mitigation of global impact (Allesch and Brunner, 2015). For such purposes, MFA has to be combined with other methods such as LCA, risk assessment or multi-criteria decision analysis (Turner et al., 2016; Makarichi et al., 2018). Even though, MFA is an essential first step and a necessary base for providing well-grounded inventory for every such task (Brunner and Rechberger, 2016; Allesch and Brunner, 2015).

Table 2.4. Indicators compliance with MFA to evaluate the MSW management system (source: Wang, 2020).

Indicator	Definition	Equation	Meaning	Reference
MSW generation Per capita	The MSW generation per capita per year (or per day)	$$G_{ca} = \frac{G_T}{P} \times 10^3$$ G_{ca}: MSW generation per capita (kg/a); G_T: The annual MSW generation in a local authority's spatial boundary (t/yr); P: the population in a local authority's spatial boundary.	It is an extremely important indicator of environmental pressure especially where comparisons with other cities and countries, or with historical status are necessary	Makarichi et al. (2018)
MSW collection rate	The quantity of waste collected by private operators and local authority and formal or informal recyclers.	$$R_c = \frac{Q_a + Q_p + Q_r}{G_T} \times 100$$ R_c: MSW collection rate (%); Q_a: quantity of waste collected by local authority (t); Q_p: quantity of waste collected by private operators; Q_r: quantity of waste collected by formal and informal recyclers.	It evaluates the effectiveness of waste collection. Any figure for R_c below 100% means there is an accumulation of materials somewhere in the system. In most cases, this represents waste buried at-source or dumped indiscriminately in the environment.	Makarichi et al. (2018)
MSW collection burden	The actual quantity of waste must be collected by local authority as a fraction of the total MSW generated assuming that waste diverted for recycling is collected by recycling companies themselves and the waste composted at source does not require collection	$$R_b = \frac{Q_a}{G_T} \times 100$$ R_b: MSW collection burden (%).	A lower waste collection burden represents a healthier MSW management system because fewer resources have to be expended in collecting residual waste and transporting it to the landfill.	Makarichi et al. (2018)
Separate delivery rate	The ratio between the amount of waste collected as separated streams and the total waste generated.	$$R_s = \frac{Q_s}{G_T} \times 100$$ R_s: separate delivery rate (%); Q_s: quantity of collected recyclables and green waste, either alone or co-mingled (t)	It evaluates the effectiveness of source separation. A higher R_s represents more waste is separated at source and less energy will be input for sorting.	Zaccariello et al. (2015)

Continue of Table 2.4

Indicator	Definition	Equation	Meaning	Reference
MSW recycling rate	The quantity of waste recycled expressed as a fraction of the primary waste generated	$$R_r = \frac{G_r}{G_T} \times 100$$ R_r: MSW recycling rate (%); G_r: quantity of recycled waste (t).	It reflects the effectiveness of a MSWM system to achieve goals related to waste recycling and reuse.	Makarichi et al. (2018)
Effective MSW recycling rate	The quantity of waste recycled after exclude the waste generated by recycling activities and include imported waste. It expressed as a fraction of the primary waste generated and imported.	$$R_{re} = \frac{G_r - G_{ra}}{G_T + G_I} \times 100$$ R_{re} : effective MSW recycling rate (%); G_{ra} : quantity of waste produced from recycling activities (t); G_I: quantity of waste imported across local authority's spatial boundary.	It reflects the effectiveness of a MSWM system to achieve goals related to waste recycling and reuse by taking into account waste produced by recycling activities.	Makarichi et al. (2018)
Recovery rate	The ratio between the amounts of waste used for recovery options (e.g. incineration with energy recovery, anaerobic digestion, and RDF production) and the total waste generated.	$$R_{en} = \frac{Q_{en}}{G_T} \times 100$$ R_{en}: recovery rate (%); Q_{en}: quantity of waste used for energy recovery (t).	It reflects the energy recovery ability of a MSWM system. However, high R_{en} does not necessarily mean that a MSWM is sustainable because energy recovery should theoretically happen after recycling have been done.	Zaccariello et al. (2015)
Landfill rate	The ratio between the amounts of waste disposed in landfill and the generated waste.	$$R_l = \frac{Q_l}{G_T} \times 100$$ R_l : landfill rate (%); Q_l : quantity of landfilled waste (t).	A higher R_l indicates a higher diversion of waste from landfill to other management options.	Zaccariello et al. (2015)
Landfill life span	The landfill lifespan (in years) is obtained by dividing the usable capacity by the landfill total stock in the assessed year.	$$Y = \frac{C_u}{Q_s}$$ Y: landfill life span (years); C_u: the usable capacity of a landfill (Mt); Q_s: the total stock in landfill in the assessed year (Mt/yr).	Landfill life span addresses the aspect of waste prevention and minimization as well as the level of waste diversion from landfills. The landfill stock value determines the rate of material accumulation in the landfill and hence its lifespan.	Makarichi et al. (2018)

Investigations of MSW management using MFA can be divided into two levels of goods and substances (Allesch and Brunner, 2015). On the level of goods, MSW (Zaccariello et al., 2015; Turner et al., 2016), plastic waste (Bogucka et al., 2008; Sevigné-Itoiz et al., 2015), food waste (Padeyanda et al., 2016; Ju et al., 2016), waste electrical and electronic equipment (WEEE) (Kumar and Shrihari, 2007; Duygan and Meylan, 2015), and household waste (Owens et al., 2011; Jensen et al., 2017) were analysed (Allesch and Brunner, 2015). On the level of substances, metals (Jung et al., 2006), carbon (C), lead (Pb), nitrogen (N), mercury (Hg), cadmium (Cd) and other compounds were analysed (Rotter et al., 2004; Chen et al., 2015; Allesch and Brunner, 2015; Liu et al., 2017b; Stanisavljevic and Brunner, 2014). MFA on the level of goods is often investigated in order to describe the structure of the MSW management sector with the goal of optimizing the treatment process or transport routes (Allesch and Brunner, 2015), but the qualitative aspects of MSW management are missed (Stanisavljevic and Brunner, 2014). MFA on the level of substances quantifies the overall flows of that substance through the sector for the identification of sources, pathways and sinks, as well as the links among them and the allocation of environmental burdens (Allesch and Brunner, 2015). For example, analysing the flow of carbon in an MSW management system quantifies the GWP caused by this system and identifies the GHG sources. Performing MFAs on both levels of goods and substances provides a complete picture of a given system for decision makers (Rotter et al., 2004; Allegrini et al., 2014; Stanisavljevic and Brunner, 2014; Allesch and Brunner, 2015).

The greatest challenge for performing an MFA either on the level of goods or substances is the availability and reliability of data. MFA data is cross disciplinary, so the format, quantity and quality of data vary from type to type originated from different sources, such as official statistics, field surveys, expert estimations, or scientific models (dos Muchangos et al., 2017). Considering the uncertainties and inconsistency of MFA data, it is important to estimate uncertainties of used data and obtained results, so as to inform stakeholders in an impartial way and make well-founded decisions (Allesch and Brunner, 2015; dos Muchangos et al., 2017). Unfortunately, data are not always available for all of a city's waste flows. When data is inadequate, scholars track the flow of specific goods or substances with reliable data, or focus on the flow of key resources (Liu et al., 2017b).

2.8.2. LCA

LCA is a tool to assess the environmental impacts throughout a product's life cycle. It is a cradle-to-grave analysis, from raw material acquisition, via production and use phases, to waste management (ISO, 2006). Based on the requirements and objective of a study, LCA can also be used for cradle-to-gate (from raw material extraction to a factory for production) or gate-to-gate (receipt of material by a factory to shipping the produced material to the gate) approaches (Yadav and Samadder, 2018).

LCA has been extensively applied to evaluate, identify, diagnose or compare the environmental burdens of MSW management technologies, practices, strategies and scenarios (Cleary, 2009; Zhao et al., 2009; Hong et al., 2010; Fernández-Nava et al., 2014; Yay, 2015; Parkes et al., 2015; Liu et al., 2017a; Milutinović et al., 2017; Coelho and Lange, 2018). It is able to identify resources used and emissions released to the environment associated with waste management options, then identify the opportunities for improvements (Cherubini et al., 2009). LCA 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, EfW reduces consumption of energy of fossil origins; recycling material replaces part of virgin materials; compost from biological treatment substitutes the production of chemical fertilizers; bottom ash from incineration plant may be used for road constructions (Franchetti and Kilaru, 2012; Jeswani et al., 2013; Turner et al., 2016).

Four phases are included in a general LCA study: Goal and Scope Definition, Life Cycle Inventory Analysis (LCI), Life Cycle Impact Assessment (LCIA), and Interpretation (Finnveden et al., 2009). In the first phases, reasons for performing the study are clarified; the intended application is stated out; functional unit and system boundary are defined. LCI phase counts all inputs and output over the systems' or products' whole life cycle related to the functional unit. The LCIA phase evaluates the magnitude and significance of the potential environmental impacts of the studied product or system. The potential impacts on the environment of each unit of products' or a system's life cycle will be expressed in a normalized way at the phase interpretation (Laurent et al., 2014a; Laurent et al., 2014b).

The functional unit ensures a functional equivalence between scenarios, thus makes scenarios comparable to each other (Zhou et al., 2018a). Functional units defined in literature can be divided into four categories (Elwan et al., 2015): 1) unitary functional unit (e.g., 1t of MSW), which is the most commonly used functional unit (Yadav and Samadder, 2018; Khandelwal et al., 2019); 2) generation based functional unit (e.g., total waste generation); 3) input-based functional unit; for instance, Malijonyte et al. (2016) used1GJ of fuel input of incineration plant as functional unit; and 4) output-based functional unit defined by the secondary material or energy output from the MSW management system; for example, Song et al. (2018) used the generation of 1kWh electricity from waste as the functional unit. Sometimes, More than 1 functional unit were used (Jeswani and Azapagic, 2016; Rajaeifar et al., 2017; Lima et al., 2019).

Defining the system boundary is an indispensable part of LCAs. System boundary has a major impact on the overall results (Yadav and Samadder, 2018). Clearly defined system boundary allows us to define and identify all related inputs into and outputs from the whole life of the target product or system (Zhou et al., 2018a). The system boundary must cover all the processes, inputs and outputs during the life cycle of the assessed system or product. In an MSW management system, the inputs include solid waste, fuel, electricity, capital goods and materials used for facility building, operation and maintenance. Outputs include all emissions and pollutants into air, water and soil. However, inputs and outputs in the processes of building a facility and its maintenance are seldom included in the system boundary of MSW management, probably because they are identical for all scenarios studied.

The system boundary is always affected by the goal, scope and selected scenarios of the studies (Yadav and Samadder, 2018). Treatment and disposal processes are commonly included in the life cycle of MSW management, but particular life phases, for example, collection and transport are excluded in some studies when these are identical for all scenarios studied (Dong et al., 2014; Zhou et al., 2018a). Waste prevention activities are included in the system boundary in a few studies (Nessi et al., 2013; Matsuda et al., 2018), but the normalized framework to analyse the waste prevention activities from a life cycle approach is unavailable (Cobo et al., 2018).

There are more than 50 LCA software available to aid the performing of LCA (Yadav and Samadder, 2018), but using an LCA software is not mandatory. Yadav and Samadder (2018) and Khandelwal et al. (2019) reported that more than half and one-third of the reviewed studies did not use software to perform LCA. Some commonly used LCA softwares are SimaPro, Gabi, WASTED (Waste Analysis Software Tool for Environmental Decisions), IWM-1, 2 (Integrated Waste Management-1, 2), and EASEWASTE (Environmental Assessment of Solid Waste Systems and Technologies). Numerous suppliers also provide databases at various levels (e.g. global, regional, and national) and departments (e.g. agriculture, industry, transportation) along with LCA software. The users of SimaPro can access the Ecoinvent database, this makes it the most commonly used LCA software (Yadav and Samadder, 2018; Khandelwal et al., 2019). Although, the LCA results calculated by different software show high variation and not negligible, even lead to contradictory conclusions in some cases (Winkler and Bilitewski, 2007). Besides, the suitability of LCA software for any study depends on many factors and conditions, such as the cost of the software, availability of the database, language of the software, objective of the study, etc (Yadav and Samadder, 2018). Therefore, LCA software should be carefully selected.

At the phases of LCIA and interpretation, impacts of the target product, process or system are assessed and reported. The commonly reported impact categories in LCA results are GWP, acidification potential, eutrophication potential, human toxicity potentials, photochemical oxidation potential, ozone layer depletion potential, energy consumption, and abiotic depletion potential (Yadav and Samadder, 2018). Among them, GWP is the often reported one because there is standardized framework and method to calculate GWP. There are also a number of impacts assessment methods are available for calculating impacts, such as CML proposed by Centre of Environmental Science of Leiden University, Ecopoints developed by the Swiss Ministry of the Environment, EDIP (Environmental Design of Industrial Products), EPD (Environmental Products Declarations) published by Swedish Environmental Management Council, the method IPCC 2007, 2013 developed by the International

Panel on Climate Change, AWARE (Available Water Remaining) recommend by the working group under the umbrella of UNEP-SETAC Life Cycle Initiative, etc. Most methods could calculate multiple impacts categories. Some methods only focus on a single environmental issue; for example, the method IPCC 2007, 2013 only focus on GWP.

LCA results are normally influenced by multiple factors at the phases of Goal and Scope Definition, LCI and LCIA such as the definition of system boundary, waste composition, technology choice, the methodologies or software adopted for calculation, energy and virgin material substitution (Yadav and Samadder, 2018; Zhou et al., 2018a; Khandelwal et al., 2019). Therefore, sensitivity analysis is a crucial part of the interpretation of LCA results to inform the robustness of the LCA results and the potential for improvement. Sensitivity analysis identifies whether any of the assumptions have considerable influences on the LCA results and if so which assumption has the highest influence (Khandelwal et al., 2019). Ideally, sensitivity analysis should be conduct for every parameter in every LCA study, but in practice, only limited selected number of parameters were assesses due to lack of data (Khandelwal et al., 2019), and not all LCA studies include sensitivity analysis. Most sensitivity analyses were conducted by varying the efficiencies of treatment and sorting, waste composition, recycling rate, and electricity mix (Khandelwal et al., 2019).

Chapter 3 Methodologies and Materials

3.1. Study Area

Nottingham is one of the UK’s eight Core Cities, located in the central UK (52° 57' N and 1° 09' W) (Figure 3.1). It covers an area of 7,537.7 hectares with an estimated population of 329,200 in 2017 (Nottingham Insight, 2018). Since the enforcement of the EU Landfill Directive, strategies, measurements and new technologies were adopted in Nottingham to divert waste from landfills, as well as preventing waste generation. KCS was introduced to Nottingham in 2002 for separating paper at source. Thereafter, material categories collected and outreach of KCS were increased year by year. Currently, alternative weekly KCS is offered by Nottingham City Council for collecting household residual waste and dry recyclables. Recyclable materials collected through KCS include paper, cardboard, cans, mixed plastics, and mixed glass. Garden waste is collected every two weeks on the same day as the collection of recyclable materials. Booking advance is required for bulky waste collection. One Civic Amenity (CA) site (also known as Household Waste Recycling Centre) and dozens of bring sites (also known as Mini Recycling Centres) located across the city for collecting multiple categories of recyclables. Orange recycling bags are provided to the homes that cannot use bins, such as communal dwellings and flats.

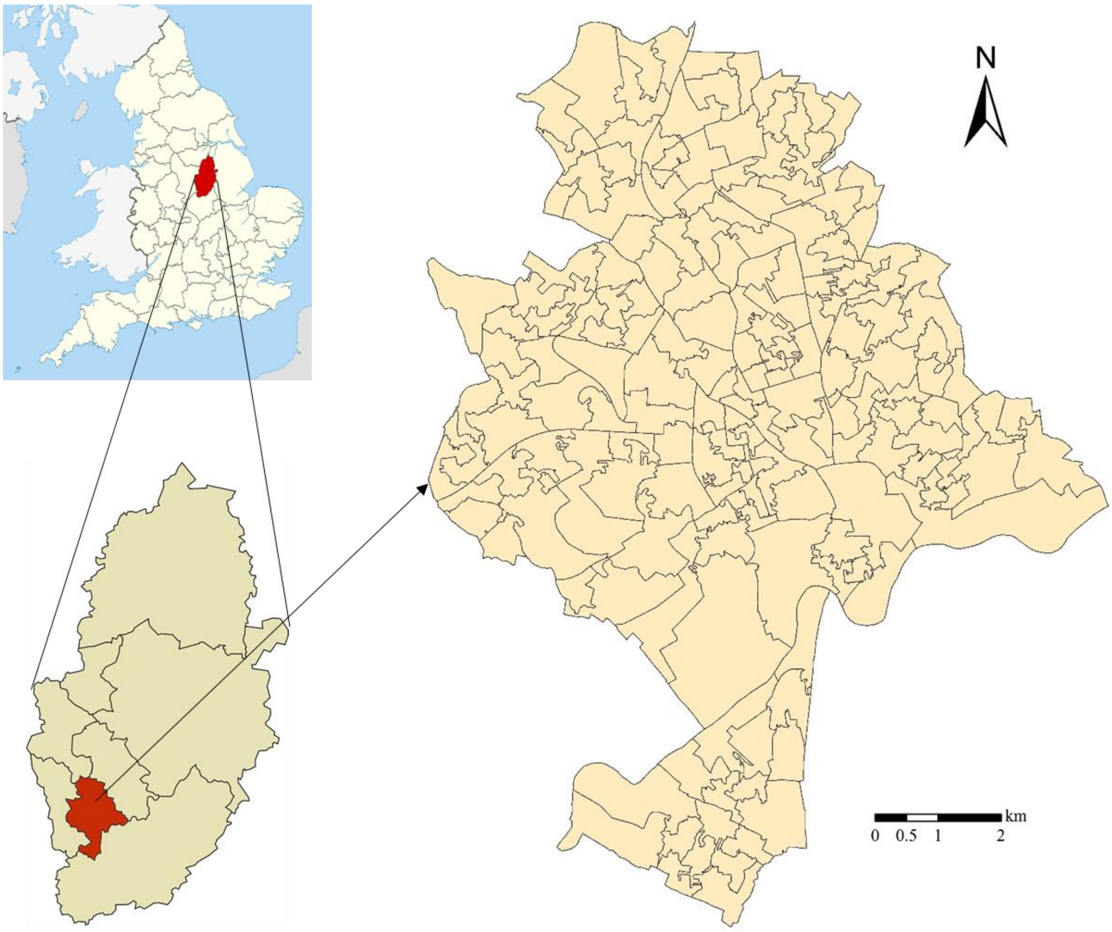


Figure 3.1. The location of Nottingham in Nottinghamshire and the UK, and Lower Layer Super Output Areas (LSOA) within Nottingham (source: Wang, 2020).

Nottingham is the pioneer regarding EfW and waste minimization in England. Eastcroft EfW cogenerates combined heat and power (CHP) from waste. Recovered power was supplied to National Grid and the heat was supplied for heating city centre buildings via a district heating scheme. Per capita MSW production in Nottingham was much lower than the average per capita MSW generation in England (412 kg) and EU (487 kg) (Eurostat, 2017; DEFRA, 2018a). Besides, Nottingham City Council has introduced ambitious MSWM targets for 2025: 1) reducing the household waste generation to 390 kg per person per year; 2) recycling 55% of household waste; and for 2030: 1) reducing the residual household waste generation to less than 200 kg per person per year; 2) achieving “zero waste to landfill” (NCC, 2010).

3.2. System Boundary

3.2.1. Conceptualized MSW

In Nottingham, MSW is defined as "household waste and any other wastes collected by a waste collection authority, or its agents, or managed by the waste disposal authority". Though MSW is defined as covering every type of solid waste apart from hazardous waste and healthcare waste, the collection of industrial and commercial waste is different and separate from the collection of household waste, so that is equivalent to household waste. For the purposes of this study then, we take conceptualized MSW as household waste (i.e. excluding hazardous, healthcare, industrial and commercial wastes), for which we have been able to obtain relatively complete statistics and comparing with the household waste cantered targets set in the EU Directives and national plans.

3.2.2. System boundary

The spatial boundary is the administrative boundary of Nottingham City Council. That means all MSW generated within the administrative boundary is counted. The temporal boundary was the statistical year from April to March of the next year; 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 processes analysed included waste generation, collection, transfer, transport, treatment and disposal. Waste treatment facilities were identified from WasteDataFlow Database (www.wastedataflow.org).

3.3. Data Collection

Quarterly data on MSW management from April 2006 to March 2017 (earliest and latest data available at the time for writing) in Nottingham was collected from WasteDataFlow Database. To fill the data gap between the year when EU Landfill Directive started in Nottingham (2000) and 2006, around fifty related documents recorded during the period 2000-2016, including meeting records and government plans, were obtained from local government websites. These documents were critically reviewed by comparing the data from different sources to confirm the reliability of these documents, for further understanding the transition of local MSW management after the EU Landfill Directive came into force. National statistical data was also collected to complement and/or verify the analysis in this study. Detailed data and data sources are depicted in Table 3.1.

Table 3.1. Data and data sources in this study (source: Wang, 2020).

Data	2001/02	2006/07	2016/17
Composition of household waste	N/A	DEFRA (2009)	NCC (2013a)
Composition of incinerated waste	N/A	Waste Research Limited (2008)	Waste Research Limited (2014)
Amount of Waste generation/collected	NCC (2005)	WasteDataFlow Database	WasteDataFlow Database
Waste treatment	NCC (2005)	WasteDataFlow Database	WasteDataFlow Database
Waste streams	N/A	WasteDataFlow Database	WasteDataFlow Database
Emission from energy mix	DEFRA and DECC (2002)	DEFRA and DECC (2007)	DEFRA and DBEIS (2017)
Policies and actions	Government documents from official websites of Nottingham City Council		
Economy and social demography	Nottingham Insight and WasteDataFlow Database		

DEFRA: Department for Environment, Food & Rural Affairs. NCC: Nottingham City Council. DECC: Department of Energy & Climate Change. DBEIS: Department for Business, Energy & Industrial Strategy.

MSW Composition in England in 2001 (Burnley, 2001), MSW Composition in England in 2006 published by Department for Environment, Food & Rural Affairs (2009) and MSW Composition in Nottingham in 2013 (Table 3.2) recorded in an unpublished government report (NCC, 2013) were adopted for our analysis in the year 2001/02, 2006/07 and 2016/17 because the data of MSW composition in these two years for Nottingham was unavailable. Composition of incinerated waste in 2008 and 2014 recorded by Waste Research Limited was adopted for our analysis in the year 2006/07 and 2016/17. There is no sufficient data to support a comprehensive MFA of MSW management in 2001/02.

Table 3.2. The composition of MSW (sources: Burnley, 2001; DEFRA, 2009; NCC, 2013).

Composition category	2001	2006	2013
Paper & card	32.0%	22.7%	14.4%
Food	21.0%*	17.8%	21.3%
Garden waste		15.8%	14.9%
Plastics	11.0%	10.0%	8.6%
Glass	9.0%	6.6%	5.5%
Metals	8.0%	4.3%	3.7%
Wood	-	3.7%	2.7%
Textiles	2.0%	2.8%	5.8%
WEEE	-	2.2%	2.8%
Other	17.0	14.0%	20.3%

WEEE: Waste electrical and electronic equipment. *: The value is the sum of percentage of food and garden waste. -: There was no record for this waste category.

3.4. Scenarios

In total, four MSW management scenarios including three historical scenarios (S1 – S3) and a future scenario (S4) have been developed (Figure 3.2), assessed and compared in this study to assess the transition of MSW management and to facilitate the future improvement for meeting the targets set in waste management regulations. The statistical year in the UK is the period from April to the following March; for example, April 2016 – March 2017, so 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.

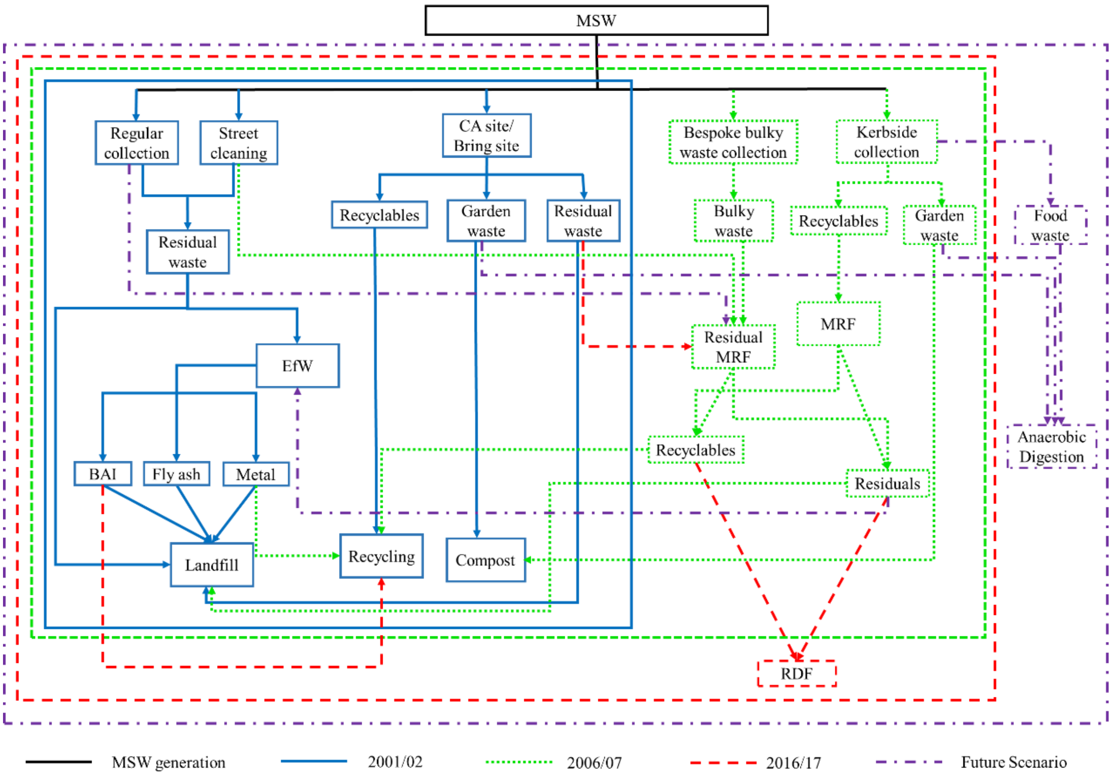


Figure 3.2. Schematic illustration of MSW management in all scenarios assessed in this study (source: Wang, 2020). Newly introduced processes and changed waste flows are identified by different colours. BAI represents bottom ash from the incineration plant.

S1 The historical state of MSW management in 2001/02. This was the year when the EU Landfill Directive put into enforcement in Nottingham and the earliest year recorded the amount of waste generated and disposed of. In 2001/02, 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. Landfill was the main waste disposal method, followed by incineration with energy recovery (NCC, 2005). Recyclable materials were collected at CA) and bring sites (also known as Mini Recycling Centres) (NCC, 2005). 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. Pre-treatment before incineration/landfill and methane collection systems at the landfill were unavailable. Bottom ash from incineration (BAI) was landfilled.

S2 The historical state of MSW management in 2006/07. This was the year before the enforcement of the Waste Framework Directive and the earliest year documented waste flows. In S2, waste management initiatives, such as kerbside collection, bespoke bulky waste collection and MRF, had been introduced to separate recyclable materials at source and prepare materials for recycling, but not fully implemented. A transfer station was still used, but now to store and transfer waste to MRF. Incineration with energy recovery became the dominate method for the disposal of MSW, followed by landfilling. Metal from bottom ash was recycled. Separately collected garden waste was treated via open windrow composting.

S3 The historical state of MSW management in 2016/17 and represents the most recent full year for which data was available for our analysis. KCS was further strengthened to serve all households in Nottingham. Only residual waste from MRF and fly ash from the incinerator were landfilled. Production of RDF had been introduced. Bottom ash from the incineration plant was recycled for aggregates.

S4 An alternative scenario based on the same quantity and quality of waste in S3 with improved source segregation and alternative waste treatment. Food waste is separately collected. Textile is added into the categories of waste collected through

KCS. Anaerobic digestion replaces open windrow composting for treating food and garden waste. Biogas from anaerobic digestion is utilized for power and heat generation. Regularly collected residual waste used to be incinerated is pre-treated in residual MRF for material recycling and RDF production before incineration.

3.5. Methodologies

MFA (addressed in section 2.8.1 in Chapter 2) was used to identify the waste flows in MSW management scenarios in Nottingham. Appropriately selected indicators based on the waste management hierarchy was used to assess the performance of MSW management based on the results of MFA. LCA (addressed in section 2.8.2 in Chapter 2) was used to assess the environmental impacts of MSW management based on the inventory of MFA. To estimate the energy recovery potential from incineration, models were built based on the methods of multiple regression analysis and ANN to predict the LHV of MSW. The details of the methodologies for model building (Chapter 5) and performance assessment (Chapter 6 and 7) will be introduced in the following chapters. The detailed technical flowchart of methodologies used in this study is illustrated in Figure 3.3

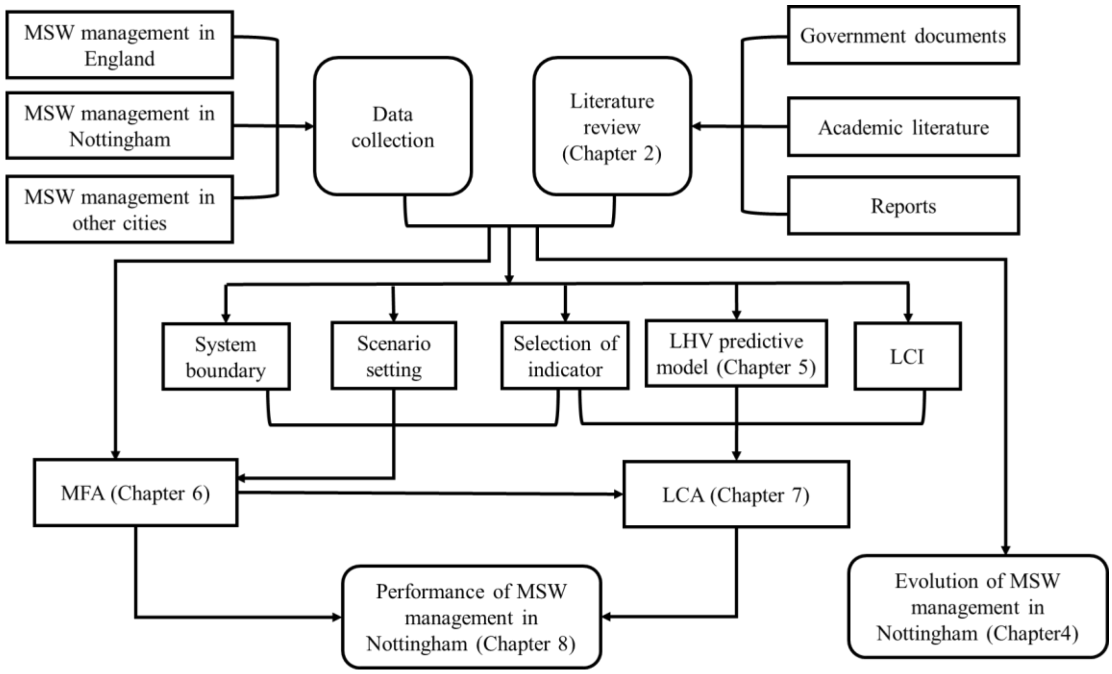


Figure 3.3. Technical flowchart of this study (source: Wang, 2020).

Chapter 4 Waste Management Policies and Progress

This chapter systematically reviewed the major directives, regulations, policies, strategies and actions concerning MSW management, as well as the progress achieved so far, at the levels of European, England and Nottingham. Figure 4.1 illustrates the timeline of successively published national and local waste management strategies responding to the EU waste directives.

This chapter contains three sections. The first section reviews the EU waste directives and the progress achieved at the EU level. The second section reviews the waste management strategies and regulation in England, focusing on the four waste management strategies which are highlighted in bold in Figure 4.1, and the achievements have been made in England. The last section discusses the strategies, policies and actions undertaken by Nottingham City Council. It also covers the backgrounds, waste situation and evolution of the waste management system in Nottingham since the millennium.

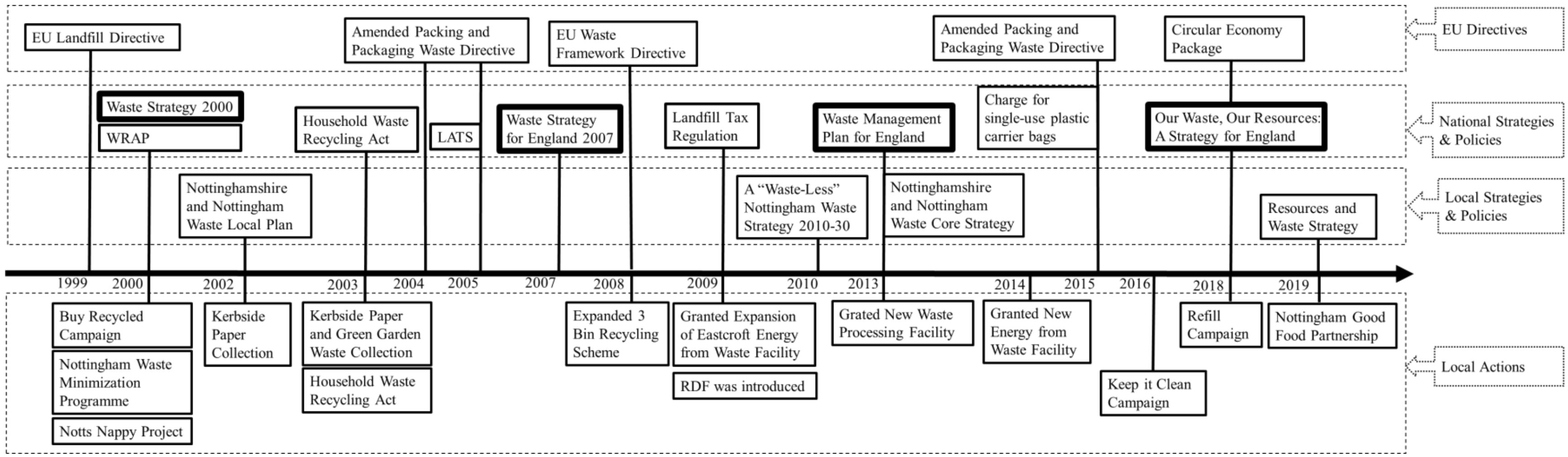


Figure 4.1. Systematic overview of the EU, national and local strategies, policies and actions for MSW management (source: Wang, 2020). The bold textbooks indicated the main waste strategies published in England.

4.1. EU Waste Directives and Progress

Since the enforcement of the EU Landfill Directive (EU Directive 99/31/EC), a series of waste directives have been successively published to reduce the quantity of waste sent to landfills, as well as to guide the establishment of sustainable waste management in EU member states (Figure 4.1, Table 4.1). Targets including waste generation reduction, landfilled waste reduction, materials recycling and recovery were set in these waste directives (Table 4.1).

EU Landfill Directive was introduced in 1999 to reduce the quantity of BMW sent to landfill and set targets of reducing the amount of landfilled BMW to 35% of 1995 levels by 2016 (EC, 1999). This directive focused on preventing and/or reducing the negative effects on the environment and human health from landfills. Mended Packing and Packaging Waste Directive 2004/12/EC and 2005/20/EC set targets for recycling 60% by weight for paper and board and 15% by weight for wood by the end of 2008 (EC, 2004; EC, 2005). After that, the EU Waste Framework Directive (EU Directive 2008/98/EC) introduced in 2008 established the “waste management hierarchy” as guiding principles for practicing sustainable waste management (EC, 2008). Sustainable MSW management objectives and targets were outlined in it and EU member countries were legally obligated to establish and enforce regional policy instruments to meet these targets. The latest published 2018 Circular Economy Package points out the crucial importance of turning waste into resources to close the loop of product lifecycles, thus benefiting both the environment and economy, and targets of recycling 60% and 65% of MSW by 2025 and 2030 and reducing landfill to maximum of 10% are set for EU member states (EC, 2019).

Table 4.1. Summary of EU directives and progresses have been made at EU level (source: Wang, 2020).

Publication Year	Directives & Strategies	Targets	Progresses
1999	EU Landfill Directive (EC, 1999)	**Reduction** in the amount of landfilled BMW to: 75% of that in 1995 within five years; 50% of that in 1995 within eight years; 35% of that in 1995 within fifteen years.	**(1) Reduction**: Per capita MSW generation declined from 521kg in 2000 to 487 kg in 2017 (Eurostat, 2019); (**2) Recycling**: improved from 28% of MSW in 2004 to 37% of MSW in 2012 (EEA, 2016), improved from 52 kg per capita in 1995 to 144 kg per capita in 2017 (Eurostat, 2019); **(3) Incineration**: improved from 67 kg per capita in 1995 to 133 kg per capita in 2017 (Eurostat, 2019); (**4) Composting**: improved from 30 kg per capita in 1995 to 81 kg per capita in 2017 (Eurostat, 2019); **(5) Landfilled waste reduction**: from 205 billion tonnes in 2004 to 157 billion tonnes in 2010 (EEA, 2016); from 302 kg per capita in 1995 to 133 kg per capita in 2017 (Eurostat, 2019).
2004, 2005	Packing and Packaging Waste Directive 2004/12/EC and 2005/20/EC (EC, 2004; EC, 2005)	**(1) Prevention**: implementing preventive measures. **(2) Recycling**: the materials contained in packaging waste by 31 December 2008: 60% by weight for glass; 60% by weight for paper and board; 50% by weight for metals; 22.5% by weight for plastics; 15% by weight for wood. **(3) Recovering**: no less than 60% by weight of packaging waste by 31 December 2008.	
2008	EU Waste Framework Directive (EC, 2008)	**(1) Prevention**: Establish waste prevention programs no later than 12 December 2013; **(2) Reusing and recycling**: 50% of household waste by 2020; **(3) Recycling and material recovery**: 70% of waste by 2020.	
2015	Packing and Packaging Waste Directive 2015/720 (EC, 2015)	**(1) Reduction**: the number of lightweight carrier bags per person to Less than 90 by 31 December 2019 and less than 40 by 31 December 2025; **(2) And/or no free** lightweight carrier bags are provided at the point of sale of goods or products.	
2018	Circular Economy Package (EC, 2019)	**(1) Recycling**: 60% and 65% of municipal waste by 2025 and 2030; **(2) Landfill reduction**: to maximum of 10% of municipal waste by 2030.	

Since 1999, the EU directives have been transposed into national legislations in EU member states as part of European waste management strategy development, to encourage separate collection and waste pre-treatment, as well as upgrading disposal methods (Pan and Voulvoulis, 2007; Vehlow et al., 2007; Lasaridi, 2009; Costa et al., 2010; Stanic-Maruna and Fellner, 2012; Brennan et al., 2016; Apostol and Mihai, 2011). All these EU waste directives promote the establishment of circular integrated MSW management which has the ability to harness the resources from waste in the form of materials and energy to close the urban metabolism circle for facilitating urban sustainability (Liang and Zhang, 2012; Cobo et al., 2018).

Waste prevention programmes covering food/organic waste, paper, packaging waste, waste electrical and electronic equipment (WEEE), and bulky waste, have been implemented in many European countries, such as Austria, Finland, Germany, Latvia, Poland, Spain, etc. (Table 4.2) (EEA, 2014b). As a result, waste generation in most of the European countries was reduced (Table 4.3), especially after the enactment of the Waste Framework Directive in which EU member states were required to implement measures and programmes for facilitating waste prevention (EC, 2008; Castillo-Giménez et al., 2019).

Table 4.2. Waste types covered in waste prevention programmes in European countries (source: EEA, 2014a).

Waste categories	AT	BE	FI	DE	HU	IE	IT	LV	LT	LU	MT	NL	NO	PL	PT	SK	ES	SE	UK
Food/organic	■	■		■	■	■	■	■	■	■	■	■	■		■	■	■	■	■
Household/municipal waste	■	■	■	■	■	■		■	■	■	■	■	■	■		■	■		■
Paper		■		■		■	■	■	■		■	■		■	■	■	■		■
Packaging waste		■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■		■
WEEE/batteries	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■		■	■	■
Bulky waste	■	■		■		■			■		■	■		■	■		■		■
Other	■	■			■	■			■		■	■	■	■	■	■	■	■	■

Note: The full names of European countries are listed in Appendix A.

Table 4.3. MSW generation per capita (kg) from 2000 to 2017 (source: Eurostat, 2019).

Country	2000	2005	2011	2017	Change 2000-2017
EU	521	515	497	487	-34
BE	471	482	456	409	-62
BG	612	588	508	416	-196
CZ	335	289	320	344	9
DK	664	736	781	781	117
DE	642	565	626	633	-9
EE	453	433	301	390	-63
IE	599	731	616	-	-
GR	412	442	503	-	-
ES	653	588	485	462	-191
FR	513	530	534	513	0
HR	262	336	384	416	154
IT	509	546	529	489	-20
CY	628	688	672	637	9
LV	271	320	350	438	167
LT	365	387	442	455	90
LU	654	672	666	607	-47
HU	446	461	382	385	-61
MT	533	623	589	604	71
NL	598	599	568	513	-85
AT	580	575	573	570	-10
PL	320	319	319	315	-5
PT	457	452	490	487	30
RO	355	383	259	272	-83
SI	513	494	415	471	-42
SK	254	273	311	378	124
FI	502	478	505	510	8
SE	428	477	449	452	24
UK	577	581	491	412	-165
IS	462	516	495	-	-
NO	613	426	485	748	135
TR	465	458	416	425	-40
CH	656	661	689	706	50

Note: The full names of European countries are listed in Appendix A.

In addition to that, waste management in European has been changed due to the introduction of new technologies and techniques for waste collection, treatment and disposal. Kerbside collection has been widely implemented in European countries for separating recyclable materials at source (Larsen et al., 2010; Dahlén et al., 2007; Gallardo et al., 2010). Waste streams in European countries were gradually shifted from landfill to other more sustainable management options such as recycling and EfW. At the early stage of transformation (2004), the majority of waste was landfilled while recycling only took a small proportion of collected waste in the European countries (Figure 4.2). In 2016, less than 40% of the generated 247 million tonnes of MSW in Europe was landfilled and recycling became the main waste management option in many European countries (Figure 4.2) (Eurostat, 2018). As a result, GHG emissions from the waste management sector reduced by 26% during 2007 – 2016 (Eurostat, 2018).

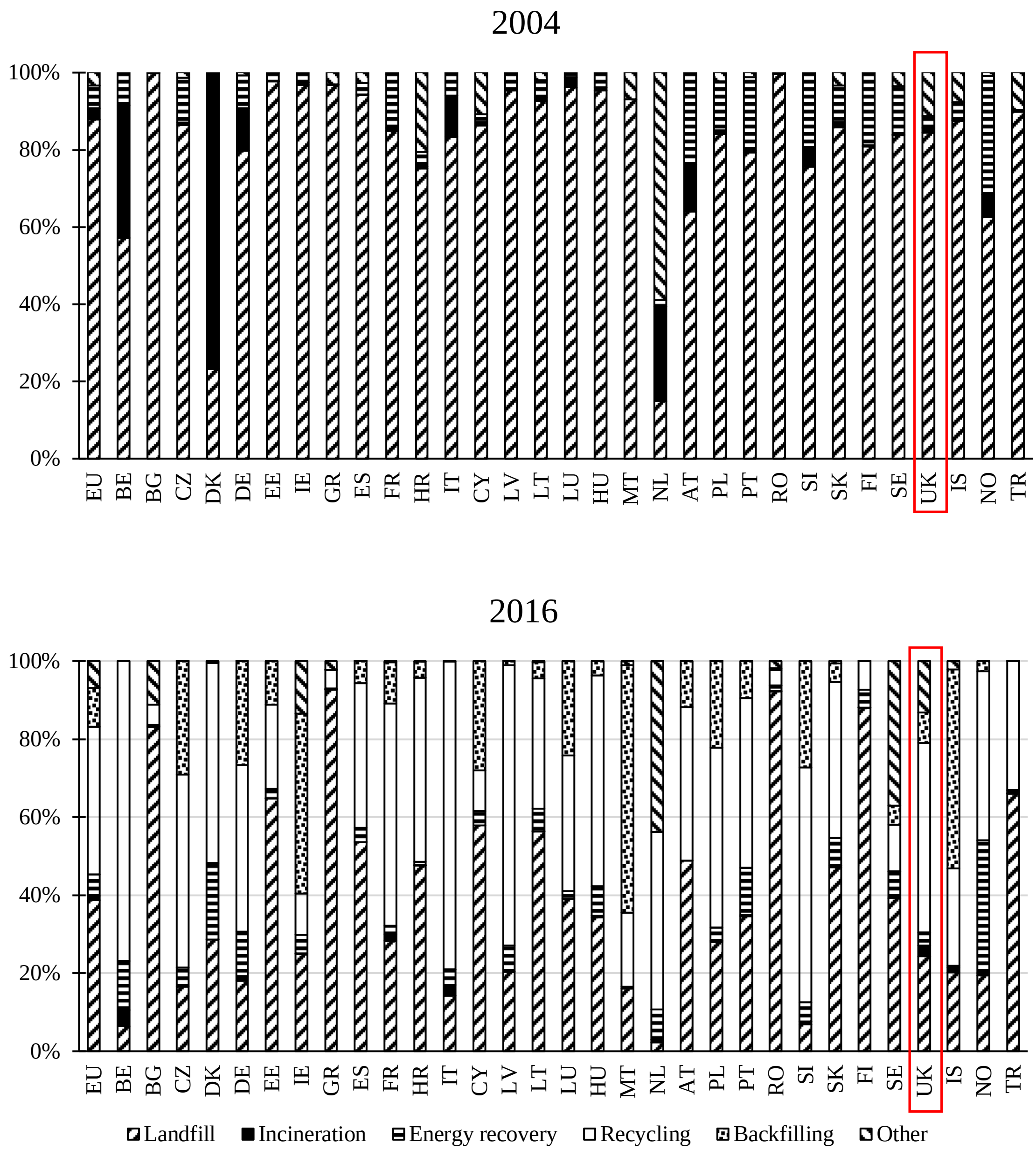


Figure 4.2. MSW management in Europe during 2004 – 2016 (source: Eurostat, 2018) (Earliest and latest data available at the time for writing). MSW management in the UK is highlighted by red box.

4.2. Implementation of EU Directives and Progress Achieved in England

Four main waste management strategies, highlighted in Figure 4.1, were successively published in England for implementing the requirements of the EU directives. Detailed management targets were also listed in these strategies (Table 4.4). Waste management programmes and regulations were also launched to facilitate the achievement of national targets. For example, the Waste and Resource Action Programme (WRAP) was set up in 2000 to promote sustainable waste management, by launching a series of campaigns and measures to educate and support public recycling and reusing waste, as well as changing consumption behaviour (WRAP, 2018a; WRAP, 2018b). WRAP also cooperates with various communities, industries and governments to make production and consumption more sustainable (WRAP, 2018a; WRAP, 2018b). Landfill Allowance Trading Scheme (LATS) was introduced in 2005 to progressively reduce the amount of BMW that could be landfilled (Fisher, 2006). As a result, the landfilled BMW was reduced by 7% annually during 2005/06–2011/12, though LATS was suspended after 2012/13 because of its coexistence with the Landfill Tax, which applies similar enforcement (Calaf-Forn et al., 2014).

Table 4.4. Summary of waste management strategies and progress achieved in England (source: Wang, 2020).

Publication Year	Directives & Strategies	Targets	Progress at each stage
2000	Waste Strategy for England and Wales 2000 (Burnley, 2001)	**(1) Recycling**: 25% of household waste by 2005; 30% of household waste by 2010; 33% of household waste by 2015; **(2) Recovery**: 40% of MSW by 2005; 45% of MSW by 2010; 67% of MSW by 2015.	**Stage 1: 2000-2007** **(1) Reduction** of household waste: 450kg/person in 2000 to 370 450kg/person in 2005; **(2) Recycling and composting** of waste: increased from around 8% in 1996/97 to 27% in 2005/06; **(3) Landfilled waste reduction**: by 9% between 2000/01 and 2004/05 (DEFRA, 2007).
2007	Waste Strategy for England 2007 (DEFRA, 2007)	**(1) Reduction**: 29% of not reused, recycled or composted household waste by 2010; 45% of not reused, recycled or composted household waste by 2020; **(2) Recycling and composting**: 40% of household waste by 2005; 45% of household waste by 2015; 50% of household waste by 2020; **(3) Recovery**: 53% of MSW by 2010; 67% of MSW by 2015; 75% of MSW by 2020.	**Stage 2: 2007 - 2013** **(1) Reduction** of household waste: by 2% per year since between 2007-2013; **(2) Recycling and composting** of household waste: increased from 36% in 2007/08 to 43% in 2012; **(3) Landfilled waste reduction**: by 60% in 2000 - 2012; **(4) Refuse derived fuel** was introduced in 2009 and increased to 887,465 tonnes in 2012 (DEFRA, 2013b).
2013	Waste management plan for England 2013 (DEFRA, 2013b)	**Reusing and recycling**: at least 50% of household waste by 2020.	**Stage 3: 2013 - 2018** **(1) Recycling** of household waste: increased to 44.9% in 2016; **(2) Landfilled waste reduction**: by 51% in 2012-2016 (DEFRA, 2018a).
2018	Our Waste, Our Resources: A Strategy for England (DEFRA, 2018b)	**(1) Recycling**: 75% of packaging waste by 2030; 65% of MSW by 2035 **(2) landfill reduction**: to maximum of 10% of municipal waste by 2035	

MSW management in England has historically been characterized by the dominate use of landfills. After 2000, a variety of waste treatments were gradually introduced to improve MSW management (Ryu et al., 2007; DEFRA, 2013b). These included MBT, production of RDF, incineration with energy recovery, compost, anaerobic digestion, gasification, and pyrolysis. In this way, the targets and strategies have facilitated the practices of MSW management based on the waste management hierarchy moving from the least favourable option to preferable options for waste disposal (Uyarra and Gee, 2013). As results, the national recycling and composting rate of MSW have been steadily improved, while the quantity and proportion of waste sent to landfill have been gradually reduced (Table 4.4, Figure 4.3).

The national regulations also drove the changes in waste collection and classification. The Household Waste Recycling Act 2003 required local authorities to collect at least two types of recyclables together or individually separated from the rest of the household by the end of 2010; this separate collection of recyclables, through the KCS, was progressively provided to every household (DEFRA, 2005). This resulted in an improvement in waste recycling and a reduction of landfill waste. Especially, separating green garden waste at source and treating it using biological methods reduced the landfilled BMW. As results, the recycling and composting share of MSW in England increased from around 10% in 2001 to around 44.9% in 2016, which was very close to the 2015 target (DEFRA, 2018a), while the landfill share of MSW reduced from 84% in 1996/7 to 44% in 2015 (Ryu et al., 2007; EEA, 2016), and the landfilled BMW in 2016 reduced to 21% of that in 1995 (DEFRA, 2018a).

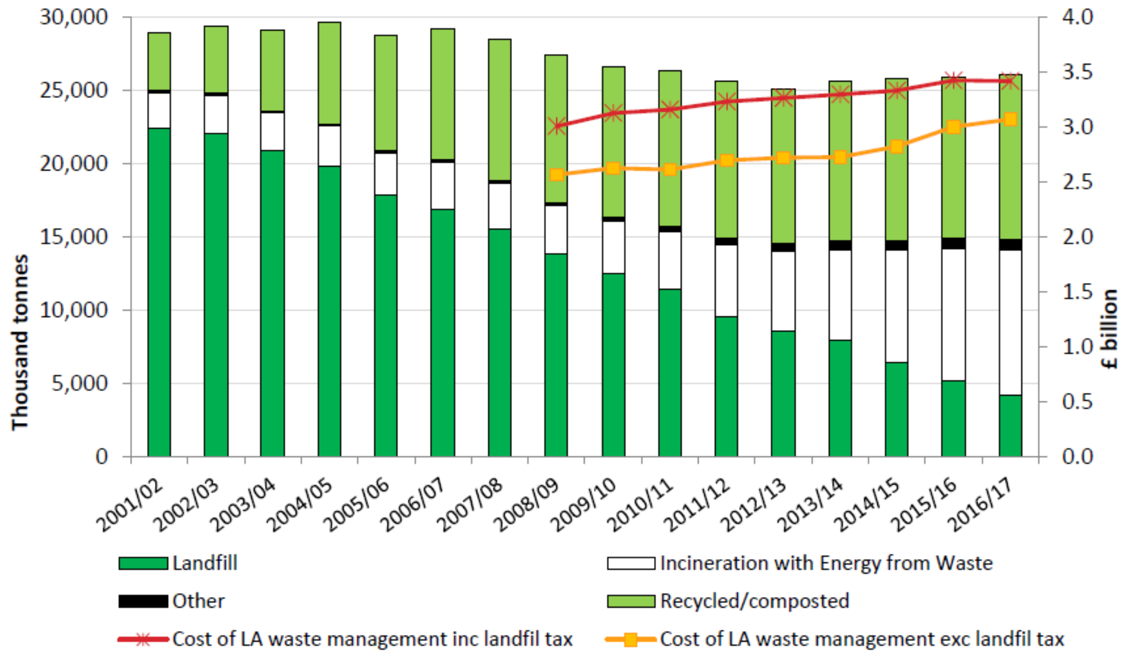


Figure 4.3. Change of MSW management in England from 2001/02 to 2016/17 (source: DEFRA, 2018a).

4.3. Efforts and Progress Achieved in Nottingham

4.3.1. Waste management strategies and measurements in Nottingham

Since the millennium, a series of waste management strategies, measurements and campaigns have been launched by Nottingham City Council to prevent unnecessary waste generation and to divert waste from landfill to material recycling and energy recovery in response to the EU and national policies (Figure 4.1) (NCC, 2006; NCC, 2009; NCC, 2010; Nottingham Insight, 2019). An Integrated MSW Management Strategy based on the waste management hierarchy was proposed by Nottingham City Council and Nottinghamshire County Council, upon the launch of the Waste Strategy for England 2000 (Nottinghamshire County Council Environment, 2002). Even before this, sustainable MSW management strategies were proposed in local plans and a variety of public related engagements and education were carried out to promote waste prevention. For example, the Notts Nappy Project was launched in 2000 to promote washable nappies and nappy laundry service to reduce disposable nappy consumption (NCC, 2000). The Buy Recycled Campaign was launched to encourage shoppers to buy paper and plastic products made with recycled materials (NCC, 2000). However, the projects were mostly voluntary; there is no legal basis for enforcing these changes to consumption behaviour.

Waste prevention was especially emphasized and reduction targets were set in local waste management strategies (Table 4.5) (NCC, 2010). Nottingham City Council cooperated with WRAP to promote waste prevention through undertaking various measures and campaigns to influence the consumer s' behaviours, providing guidance and facilities to facilitate reuse. For example, the "Love Food, Hate Waste" campaign send message and materials to influence purchasing and cooling behaviours for reducing the unnecessary purchasing of food and food waste requiring disposal (NCC, 2010), and the Refill campaign was launched to reduce the use of plastic bottles by promoting free tap water (NCC, 2018a). The Nottingham city council promotes "Freecycle" and "Exchange" networks to encourage the reuse of unwanted goods from businesses and households (NCC, 2010).

Table 4.5. Summary of waste management strategies and measurements in Nottingham (source: Wang, 2020).

Publication Year	Directives & Strategies	Targets
2002	Nottinghamshire and Nottingham Waste Local Plan (Nottinghamshire County Council Environment, 2002)	**(1) Recycling**: 25% of household waste by 2005; 30% of household waste by 2010; 33% of household waste by 2015; **(2) Recovery**: 40% of MSW by 2005; 45% of MSW by 2010; 67% of MSW by 2015.
2010	A "Waste-Less" Nottingham Waste Strategy 2010-2030 (NCC, 2010)	**(1) Reduction** household waste: 400 kg/person of household waste by 2020; 390 kg/person of household waste by 2025; 222 kg/person of residual household waste by 2020; <200 kg/person of residual household waste by 2030; **(2) Recycling**: 50% of household waste by 2020; 55% of household waste by 2025; **(3) Landfill waste reduction**: Zero waste to landfill by 2030.
2013	Nottinghamshire and Nottingham Waste Core Strategy (NCC, 2013b)	**(1) Recycling and** composting 70% of all waste by 2025; **(2) Landfill waste reduction**: Disposal rate reduce to no more than 10% by 2025.
2019	Resources and Waste Strategy (Nottingham Insight, 2019)	**(1) Prevention**: Eliminating avoidable plastic waste by 2042; **(2) Recycling**: all plastic packaging waste to be recyclable, reusable or compostable by 2025; **(3) Landfill waste reduction**: Eliminating food waste to landfill by 2030

In addition to these initiatives and outreach waste reduction programmes, schemes were introduced to supplement the waste management hierarchy in Nottingham. KCS was introduced in 2002, then both the number of collection locations and the types of recyclables to be collected by KCS have expanded annually (NCC, 2006; NCC, 2009). For the waste that may not be readily recycled, alternative solutions for waste treatment other than landfilling have been developed. The Eastcroft EfW Facility built in the early 1970s was retrofitted and upgraded in 1998 to generate energy from waste in the form of combined heat and power. Its capacity is currently 170,000 tonnes/year, and this is proposed to increase to 300,000 tonnes/year in the near future (FCC Environment, 2015). RDF was introduced in 2009 to improve the energy recovery potential. These investments in waste treatment infrastructure did not only reduce the amount of waste to be landfilled to fulfil the national and EU targets, but also provide new resources for energy generation. However, the recycling and composting rate of MSW in Nottingham is relatively lower than that in other core cities, and lower than the national, but higher than the EU average value in the latest years (DEFRA, 2018a; Castillo-Giménez et al., 2019).

4.3.2. Evolution of MSW management in Nottingham

During the last two decades, Nottingham underwent a transformation from a relatively simple landfill & EfW model to a complex, multi-technology MSW management (Figure 4.4). Before the millennium, source-separated collection was unavailable, MSW was regularly collected, then sent to either landfill for disposal or incineration plant for energy recovery. The landfill was the dominated waste disposal method. Pre-treatment before incineration and landfill was unavailable. Only minimum recyclable materials such as glass, paper and cardboard were recycled at bring sites at most shopping centres and other public locations and CA site.

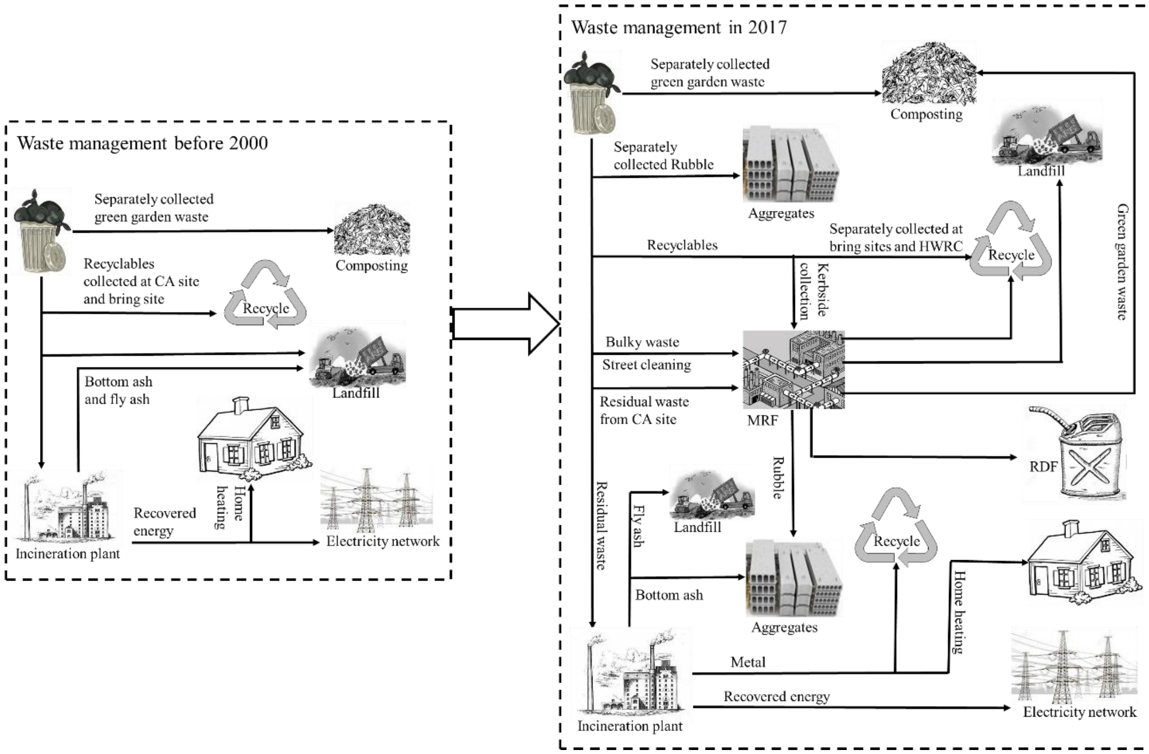


Figure 4.4. Overview of MSW management system in Nottingham in 2000 and 2017 (source: Wang, 2020).

After introducing multiple technologies, techniques and facilities, MSW management in Nottingham became more complex than before, and more pathways of waste flow were created (Figure 4.4). The introduction of KCS diverts waste from landfills to recycling and composting. The introduction of MRF further improved the recycling rate by pre-treating the landfilled waste. The recovery rate was also improved by producing RDF. As a result, the MSW generation and landfilled waste decreased while waste sent to recycling and composting increased during the period of 2001/02 – 2016/17 (Figure 4.5). It is worth noting that the per capita MSW generation in Nottingham (361 kg) was much lower than that in the majority of British cities (DEFRA, 2018c), and lower than the average value in England (412 kg) and EU (487 kg) in 2016 (Figure 4.4) (Eurostat, 2017; DEFRA, 2018a). It is possible that in the long term these initiatives may have contributed to waste reduction.

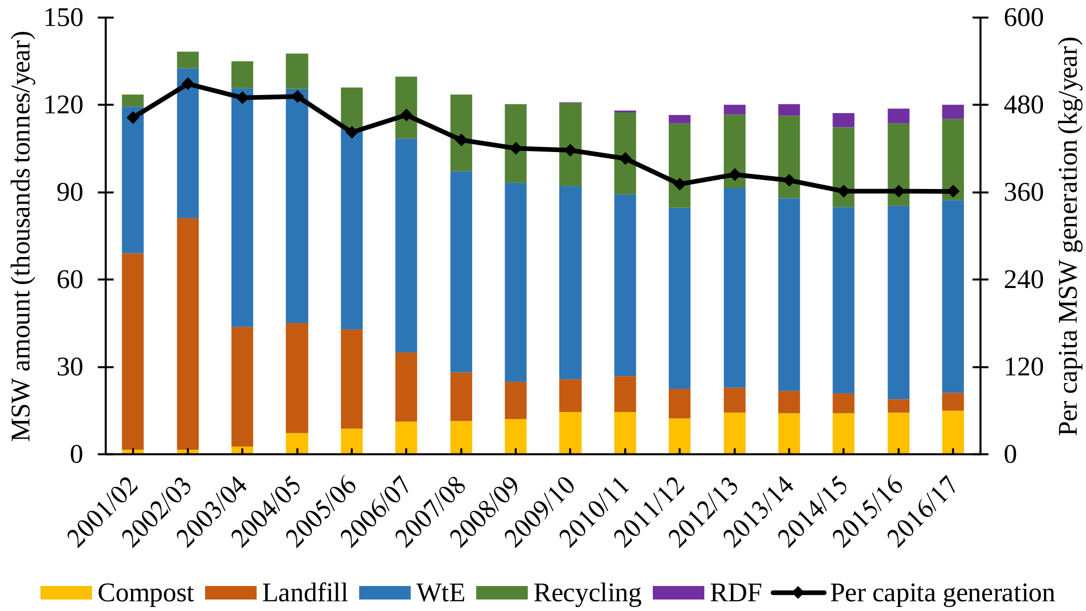


Figure 4.5. MSW generation and treatment from 2001/02 to 2016/17 in Nottingham (source: Wang, 2020).

Chapter 5 Generalised Models to Predict the LHV of MSW

5.1. Introduction

EfW is an essential part of integrated MSW management. Especially, incineration with energy recovery is increasingly popular worldwide (Brunner and Rechberger, 2015). EfW incineration is not only beneficial for environmental protection and conserve resources in an affordable way (Ryu et al., 2007; Cheng and Hu, 2010; Brunner and Rechberger, 2015), but also having an attractive net profit margin and return on investment (Corvellec et al., 2013; Zhao et al., 2016a; Kim and Jeong, 2017).

5.1.1. Predicting LHV for EfW approach

Designing and operating an MSW incinerator requires an understanding of the heating value of MSW (Chang et al., 2007). LHV is the more relevant parameter than HHV to evaluate the feasibility of using a particular set of MSW as a fuel. This is because the latent heat of the moisture in the waste is unable to be harnessed to produce neither electricity nor heat when the water in MSW vaporizes during combustion (Pavlas et al., 2011) In this case, it is more realistic to use LHV over HHV in demonstrating the energy that can be harvested in MSW in the form of electricity and/or heat in an incinerator (Ogwueleka and Ogwueleka, 2010; Drudi et al., 2019). Therefore, properly predict the LHV of MSW is vital for designing and operating EfW facilities.

Estimating LHV using a bomb calorimeter or using models built based on proximate analysis or ultimate analysis is costly and time-consuming, the performing of these experiments also requires the staff having a higher level of skills in chemical analysis (Shu et al., 2006; Chang et al., 2007). Alternatively, LHV models that based on the physical composition of MSW are less costly and less skill intensive, requiring only a relatively straightforward sorting and weighting of the waste to produce estimates of LHVs at an acceptable level of accuracy (Drudi et al., 2019; Liu et al., 1996; Chang et al., 2007; Lin et al., 2015; Shu et al., 2006). To minimize the cost for evaluating the applicability of EfW, models based on physical components are more convenient for predicting the LHV of MSW under a variety of scenarios.

5.1.2. Using wet-based physical compositions to predict the LHV of MSW

Based on the definition of LHV, the moisture content in MSW is particularly influential (Komilis et al., 2014). To predict LHV, using a mathematical relationship including the moisture content of MSW together with the dry basis of the waste composition is theoretically correct. However, in practice, waste is usually sorted and reported on a wet-basis. Replacing the weight estimation of MSW compositions at dry-basis with the ones at wet-basis in the relationship double counted the impacts of moisture. For example, the LHV of MSW with a higher proportion of food waste is more likely to be lower partially because of its higher moisture content (Komilis et al., 2014; Zhou et al., 2014). Besides, developing models based on the wet-based composition also consumes less time and money during the data collection phase of analyses.

Table 5.1 summarizes the existing LHV predictive models based on the wet-based physical composition of MSW from published literature. Wet-based physical compositions of MSW have been used to predict LHVs but the models vary greatly depending on the level of detail in physical compositions as explanatory variables. In general, plastics and paper are included in the models but not necessarily other types of waste. Khan and Abu-Ghararah (1991) developed a predictive model (Eq. 5.1 in Table 5.1) based on the physical composition of MSW collected globally (86 cities in 35 countries) using linear regression analysis. With the composition of food, plastics (including rubber), paper (including cardboard) as explanatory variables, the predicted LHVs were consistent with the predictions based on the modified Dulong equation. However, the consistency with the experimental values was not evaluated. Other models were estimated based on data collected from one or a small number of specific locations. They perform well when data used for validation was obtained from the same location as the training data, but as one would expect, they perform less well otherwise (Lin et al., 2015). For example, Lin et al. (2015) developed predictive models (Eq. 6 and Eq. 7 in Table 5.1) based on the waste composition in Xiamen, China. The prediction performance of the model was good when being verified using the data from the same location, but not good when using data from western and northern cities in China. Furthermore, collinearity among independent variables has seldom been assessed in the evaluation of prior regression models.

Therefore, developing an easy and rapid predictive model for estimating the LHV of MSW which could be applied regionally or globally can be beneficial for designing and operating an incinerator and associated waste management strategies. This may be especially important in light of the trend towards the rapid urbanized of developing countries, where the characteristics of waste change during the 15 to 20 years' life span of an incinerator (DEFRA, 2013a).

Table 5.1. Summary of empirical models for predicting LHV of MSW based on wet-based physical composition (source: Wang, 2020).

Country	Equation	NO.	Source
35 countries	LHV = 53.50 (F + 3.6Pa) + 372.16 Pl	Eq.5.1	Khan and Abu-Ghararah (1991)
Jordan	LHV = 621.04(Pl/Pa) + 5316.54	Eq.5.2	Abu-Qudais and Abu-Qdais (2000)
Malaysia	LHV = 112.157F + 183.386Pa + 288.737Pl + 5064.701	Eq.5 3	Kathiravale et al. (2003)
Malaysia	LHV = 81.209F + 285.035Pl + 8724.209	Eq.5 4	Kathiravale et al. (2003)
Taiwan	LHV = 92.53Pa + 117.65Pl + 102.99 T + 53.17 W + 25.12 F + 240.32R + 72.01Mi	Eq.5.5	Lin et al. (2013)
China	LHV = 219 Pl + 112 Pa + 108 W + 115 T	Eq.5.6	Lin et al. (2015)
China	LHV = 219 Pl + 109 (Pa + W + T)	Eq.5.7	Lin et al. (2015)

Note: LHV, lower heating value (Unit: kJ/kg); F, percentage of food waste; Pa, percentage of paper; Pl, percentage of plastics; R, percentage of rubble and leather; T, percentage of textile; W, percentage of wood; Mi: miscellaneous component.

5.1.3. Methods for developing LHV predictive model

Multiple regression analysis is the commonly used method in building predictive models for estimating the heating value of MSW (Liu et al., 1996; Nwankwo and Amah, 2016; Ibikunle et al., 2018; Drudi et al., 2017), but it has limitations in estimating dependent variables (in this case LHV) when the resolution of independent variables (in this case, the composition of waste) is low, and regression models are sensitive to the precision of the input data (Dong et al., 2003; Ogwueleka and Ogwueleka, 2010). This issue tends to be amplified when LHV models are to be based on the physical compositions of MSW which are highly related to the environmental, geographical, social and economic factors and associated spatiotemporal variations (Das et al., 2019). Indeed, this limitation usually restricts the applicability of regression models within the spatiotemporal boundary of the original datasets.

Equations built using regression analysis in previous studies are usually linear (Khan and Abu-Ghararah, 1991; Lin et al., 2013; Chang et al., 2007), but this is not always true. Abu-Qudais and Abu-Qdais (2000) have developed a relationship between energy content and the plastic to the paper ratio of MSW, which indicates that this relationship is not necessarily linear. Nwankwo and Amah (2016) have developed a non-linear relationship between energy content and the physical composition (wood, food, leather, paper, plastics, textile and miscellaneous component) of MSW. Cross-plots between the individual physical composition and LHV also indicate that the linearity of relationships between some variables (e.g., textile and wood) and LHV are not clear (Figure 5.1).

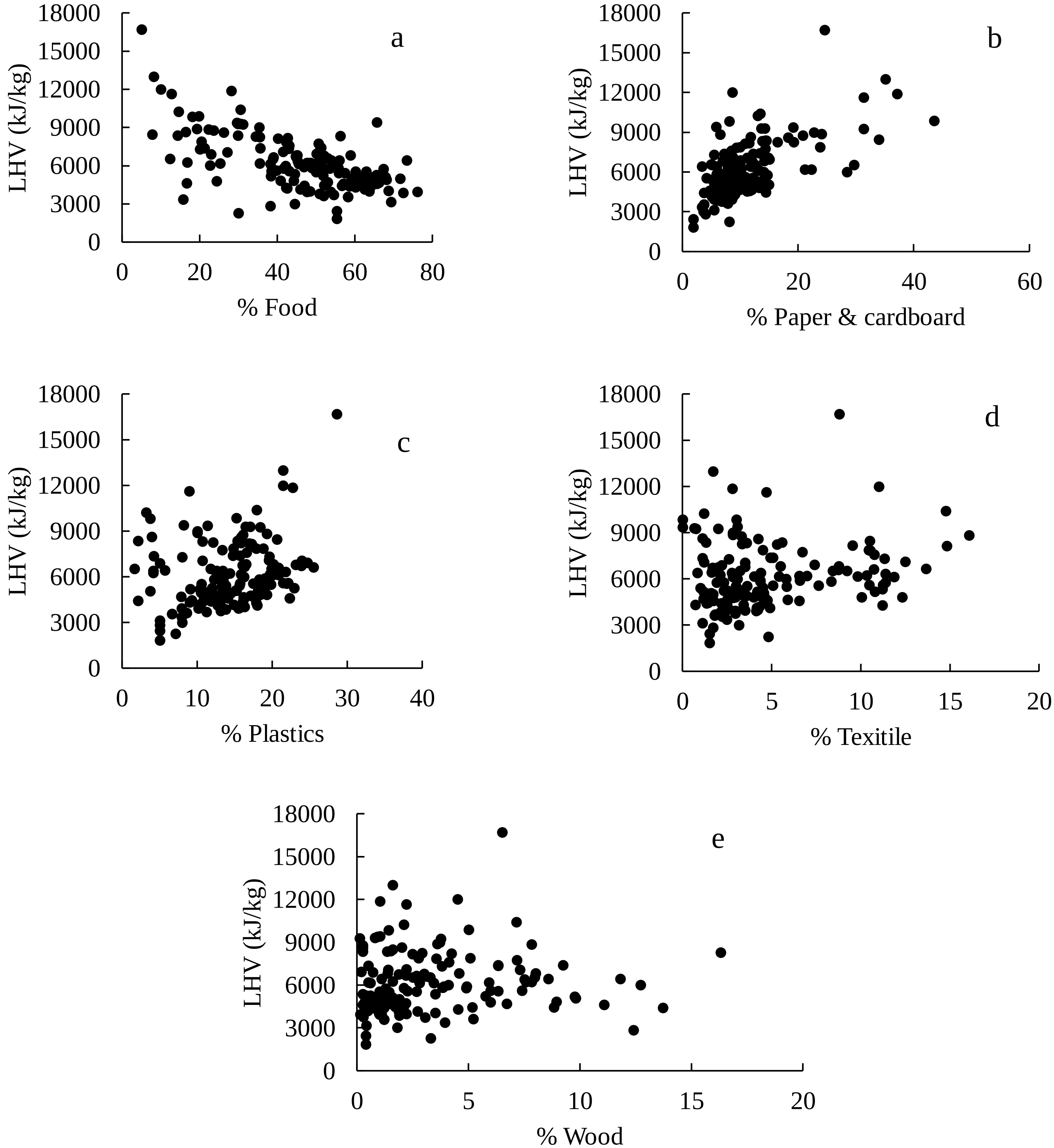


Figure 5.1. Cross-plots of LHV verses individual waste composition (source: Wang, 2020).

Besides, regression analysis utilizes limited variables and the range of functions they can model is limited (Eftekhar et al., 2005; Kumar, 2005). But MSW is complex with large numbers of distinct materials, and its complexity is increasing with the consumption of new products and technologies. Furthermore, as a standard statistical technique, regression analysis often ignores the variables not making statistically significant contributions. The insignificant contributions of these variables are because of their small shares in MSW samples rather than their energy content. For example, wood and textile were included in Eq. 5.5 – Eq. 5.7 but not in Eq. 5.1 – Eq. 5.4. Since we are trying to build an internationally applicable model, we cannot ignore those materials that may locally significant in some cases.

Alternatively, artificial neural networks (ANN) have the inherent ability to learn and utilise some prior hidden information in the training data. They are also able to straightforwardly capture non-linear relationships between dependent and independent variables (Dong et al., 2003; Ogwueleka and Ogwueleka, 2010; Khuriati et al., 2015), as they avoid the need to identify an appropriate data-fitting function (Patel et al., 2007). ANN models also allow for the inclusion of multiple inputs (or variables) and model adjustment of the models when new datasets are input (Eftekhar et al., 2005). Furthermore, the performance of ANN models is improved when the sample size and the number of groups or the number of variables increases (Kumar, 2005). This means that ANNs have the potential to take all MSW compositions into account to estimate the LHV.

ANN is an artificial intelligence technique that quantitatively analyses information and builds models by learning and training from the input data in a way that mimics the neuron functions of the brain (Dong et al., 2003; Ogwueleka and Ogwueleka, 2010). It is widely applied to problems relating to predicting, forecasting, clustering, and pattern classification (Samarasinghe, 2016). Predicting the LHV of MSW is one of its plausible applications (Dong et al., 2003; Ogwueleka and Ogwueleka, 2010). Dong et al. (2003) applied a feed-forward neural network in predicting the LHV of MSW using 108 datasets collected in Nanjing. When being verified using the experimental LHV of these samples, the ANN model appeared to perform better in predicting LHV of MSW than that of the regression analysis. The application of ANN in predicting the LHV of MSW in Abuja also demonstrated good accuracy (Ogwueleka and Ogwueleka, 2010).

A model built based on ANN methodology consists of an input layer of neurons representing the behaviours of the explanatory variables, one or several hidden layers of neurons and a final layer of output predicting the values of the dependent variable (Figure 5.2). Increasing the number of hidden layers and/or neurons in each hidden layer increases the complexity of an ANN model; this can potentially gain the reliability of the results that simulate non-linear functions (Gevrey et al., 2003). However, it is not always good to design the model with as many hidden layers as possible. The larger size of input datasets is required for model training as the complexity of the model grows. For the LHV of MSW, the amount of the available datasets is usually limited (Bishop, 2007). When the dataset is not big enough, complex ANN over fits easily (Bishop, 2007). In addition, a complicated model slows down the process to obtain an optimal solution (Bishop, 2007).

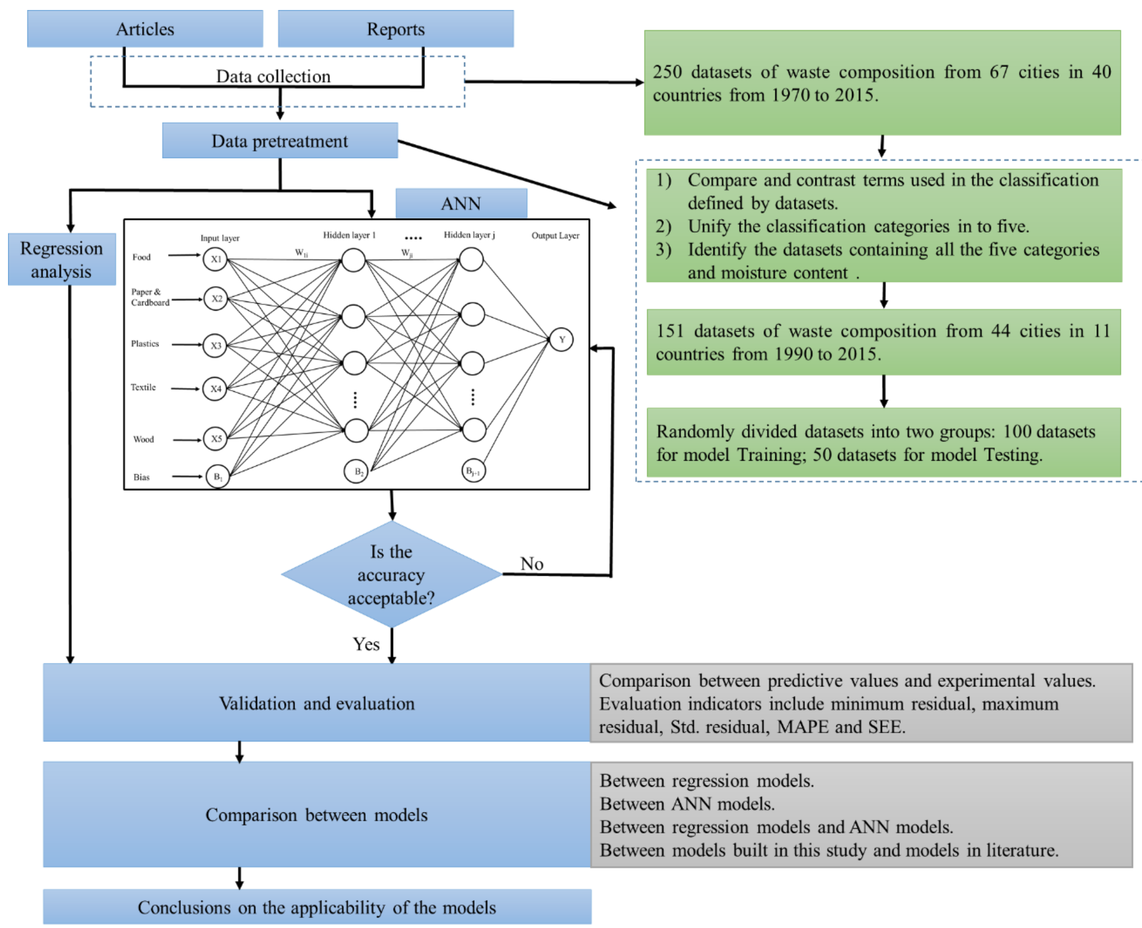


Figure 5.2. Flowchart for building and validating the LHV predictive model (source: Wang, 2020). MAPE, mean absolute percentage error; SEE, the standard error of the estimate.

The deep learning process of ANN almost works as a "black-box" and it is believed that the model generated based on a larger quantity of data can accommodate to the real situation better. However, the quantity of datasets in the previous studies related to LHV prediction is as small as dozens of data points and fixed groups of MSW materials have been modelled (Ogwueleka and Ogwueleka, 2010; Khuriati et al., 2015). Besides, data used for training and validating the ANN model in previous studies were collected from the same place and these ANN models have not been verified using global or regional data.

These models may only be used locally. To generate an ANN with wider applicability and suitability in predicting LHV of MSW produced in any places of the globe may be needed.

Therefore, this Chapter (1) builds LHV predictive models of MSW, utilizing wet-based physical composition, and employing both regression analysis and ANN, (2) evaluates how model building methods compare in their accuracy in predicting the LHV of MSW, (3) selects the most reliable model for application to city, national and global scales, and (4) provides a reference for the choice and application of the most applicable model building techniques. The model which has the best performance will be used to aid the performance assessment of MSW management by estimating the energy recovery potential from incineration.

5.2. Materials and Methodology

5.2.1. Data collection and processing

MSW compositions and their associated LHVs were collected from a systemic literature review. Literature or statistical data were obtained from articles and reports published in English and Chinese since 1990. Keywords such as "lower heating value + municipal solid waste", "characterization + municipal solid waste" and "energy + municipal solid waste" were used for searching articles and reports through Google, Google Scholar and Scopus. We were able to collect a total number of 250 datasets documenting average waste compositions and LHVs from 67 cities in 40 countries from 1970 to 2015.

Not all of these 250 datasets contained all of the five categories ('food', 'paper & cardboard', 'plastics', 'textile' and 'wood') of waste we intended to account for, because of the diverse waste classification systems used by the sources we identified, with some researchers using categories that are specific to their purposes e.g. Alam and Kulkarni (Alam and Kulkarni, 2016) and Lin et al. (Lin et al., 2015). In some research, variety types of waste were grouped into the category of miscellaneous or others because they occupied a very small proportion or contributed little to LHV (Zhou et al., 2014). For the purpose of this analysis, we retain all datasets containing all the five categories, but the percentage of the five categories do not necessarily add up to 100%. Additionally, the categories of 'putrescible', 'organics', 'vegetable and fruit' in some research are considered as subcategories of food waste; the values of these compositions in the same dataset were added together to represent food waste. In one case garden waste was combined with the wood category, so that it became part of the statistic wood (Liu et al., 2013).

Applying the criteria described above, the 250 datasets identified from the literature were filtered. As a result, 151 datasets from 44 cities in 11 countries from 1990 to 2015 were identified as inputs for the ensuing model building work. Most of these datasets (around 90%) covered cities in developing countries such as China, India, Philippines and Thailand, with the remaining datasets relating to cities in developed countries such as Finland, Italy, Spain, the USA and the UK.

5.2.2. Multiple regression analysis

Regression analysis was performed to establish a linear model for LHV prediction, which is a similar style to the equations in Table 5.1. The average wet weight percentage of the five MSW categories were used as explanatory variables to estimate the overall LHV of the mixture of MSW. The collinearity between variables in models was diagnosed using condition indexes and variance inflation factors (VIFs). A condition index exceeding 15 indicates potential collinearity, while indices exceeding 30 suggest serious collinearity (James et al., 2013) and a VIF exceeding 10 indicates unacceptable collinearity (James et al., 2013).

5.2.2.1. Model development

The full model (Eq. 5.8), containing all the five variables, was established first. Moisture content was eliminated from the model to prevent double counting of its influence on LHV:

$$LHV = \alpha_1 F + \alpha_2 Pa + \alpha_3 Pl + \alpha_4 T + \alpha_5 W + C \qquad \text{Eq. 5.8}$$

The definitions of the variables in the equations are consistent with those in Table 5.1. The variable C was added to represent bias.

Some researches indicated that most of the LHV of MSW was contributed by the paper, food (or organics in some cases) and plastics in the mixture, with this accounting for over 70% of the MSW on a dry basis (Chang et al., 2007; Drudi et al., 2019; Khuriati et al., 2015). The contribution of the remaining ingredients is statistically insignificant and may be neglected in the model. On this basis, the full model (Eq. 5.8) can be simplified, as follows (Eq. 5.9):

$$LHV = \alpha_1 F + \alpha_2 Pa + \alpha_3 Pl + C \qquad \text{Eq. 5.9}$$

5.2.3. ANN

The ANN model consists of an input layer of neurons representing the behaviours of the explanatory variables, one or more hidden layers of neurons and a final layer of output predicting the value of the dependent variable (Figure 5.2). Neurons within the same layer have no connection or interaction with each other. They only sequentially connect with the neurons in the next layer (Figure 5.2). Each connection is associated with a weight by an activation function and the output value is calculated by multiplying the weight and input. The weights and bias values are not pre-determined. During the training phase, the ANN learns and adjusts the weights and biases to optimize the model based on the input of training data. The output y_i of neuron i is expressed as in Eq. 10 (Shu et al., 2006; Bunsan et al., 2013):

$$y_i = f(\sum_{i=1}^{n}(w_{ji})(x_i) + b_j) \quad \text{Eq. 5.10}$$

Where y_i is the output of neuron i; x_i is the input of neuron i; w_{ji} is the connecting weight between neuron i of the input layer and the neuron j of the output layer; b_j is the bias; and f is the activation function.

Initial training parameters are generally set based on experience, then adjusted to optimize the solution when all training data are input. In previous attempts to predict LHV using ANN, only one hidden layer was defined, with the number of nodes in the hidden layer varying from 3 to 35. The epoch (or the number of iterations) varies from 200 to 4000, and the learning rate varies from 0.01 to 0.15 (Dong et al., 2003; Shu et al., 2006; Ogwueleka and Ogwueleka, 2010; Khuriati et al., 2015). According to Dong et al., changes in the sum-squared error of the ANN are minimal beyond 800 epochs (Dong et al., 2003). Based on the literature, the ANN modelling in this study was set as: 3 – 35 nodes in one hidden layer, a maximum of 1000 epochs, and a learning rate of between 0.005 and 0.1, with an interval of 0.005.

5.2.4. Validation and evaluation

5.2.4.1. The contribution of compositions estimated by ANN

ANN modelling works like a black-box in comparison to conventional regression analysis. It is difficult to directly identify all the weights that were employed in the calculation of the LHV. To identify the contribution of each explanatory variable to LHV in the optimized ANN model we obtained, a simulation was conducted by inputting one variable in the range of 0 – 100 with an interval of 0.1 in the ANN model, while setting the value of all other variables to 0. The results of the simulation are considered as the contribution of each explanatory variable to LHV at various levels of percentages in MSW composition.

5.2.4.2. Validation and evaluation

100 out of 151 pre-treated datasets were randomly selected for model training in both regression analysis and ANN. The remaining 51 datasets were used for validation and evaluation. Models were evaluated and compared using statistical indicators include the minimum residual, maximum residual, the standard deviation of the residual (Std. residual), the mean absolute percentage error (MAPE) and the standard error of the estimate (SEE). The MAPE and SEE are expressed as follows (Lin et al., 2013; Lin et al., 2015):

$$MAPE = \frac{1}{n}\sum_{i=1}^{n}\frac{|x_{pi} - x_{ei}|}{x_{ei}} \times 100 \qquad \text{Eq. 5.11}$$

$$SEE = \sqrt{\frac{\sum_{i=1}^{n}(x_{pi} - x_{ei})^2}{n-1}} \qquad \text{Eq. 5.12}$$

where x_{pi} represents predicted LHV, x_{ei} represents experimentally measured LHV, and n represents the number of datasets. The evaluation of MAPE was divided into four levels:

excellent (MAPE < 10), good (MAPE = 10 to 20), acceptable (MAPE = 20 to 50), and unacceptable (MAPE > 50) (Lin et al., 2013). In practice, the lower the SEE the better the model is deemed to perform.

5.3. Results

5.3.1. Characterization of MSW

A statistical analysis reveals that the characteristics of MSW collected from the literature vary among cities and change over time (Table 5.2 and 5.3). On average, food waste covers the greatest proportion in MSW (average 46.0%, ranging between 5.0% and 76.0%), followed by paper (ranging between 1.8% and 43.5%) and then plastics (ranging between 11.6% and 28.6%). In the majority of the cases, these three types of waste together account for over 60% of the MSW. Textile and wood constitute much smaller percentages. Moisture content among the MSW data collected ranges from 8.3% to 70.5%. The highly variable compositions may have also contributed to one order of magnitude of difference in their corresponding LHVs (1,663 kJ/kg to 16,700 kJ/kg). The results also reveal that cities with higher per capita incomes tend to have a higher percentage of paper and plastics.

Table 5.2. Summary of MSW compositional data and LHV (source: Wang, 2020).

	N	Mean	Median	Min.	Max.	Std. Error of Mean	Std. Deviation
Food (%)	151	46.0	48.5	5.0	76.0	1.4	16.7
Paper (%)	151	11.1	9.1	1.8	43.5	0.6	7.1
Plastics (%)	151	14.2	14.8	1.6	28.6	0.5	5.5
Textile (%)	151	4.8	3.5	0.0	16.1	0.3	3.6
Wood (%)	151	3.3	2.2	0.1	16.3	0.3	3.1
Moisture (%)	151	48.1	51.3	8.3	70.5	1.1	13.5
LHV (kJ/kg)	151	6,157.4	5,777.0	1,849.0	16,700.0	178.7	2,195.3

Table 5.3. Comparison of composition and LHV of MSW in three representative cities at the various development stage (source: Wang, 2020).

	Beijing [a]	Nottingham		Roorkee [b]
	2013	2008	2014	2015
Food (%)	63.96	26.15	29.88	17.03
Paper (%)	5.61	18.19	14.21	10.88
Plastics (%)	12.05	15.79	16.4	3.48
Textile (%)	2.44	4.23	0.65	1.9
Wood (%)	2.35	0.25	0.8	1.54
Moisture (%)	61.39	33.74	40.52	13.425
LHV (kJ/kg)	5,926	8,600	9,290	8,016.75
Population (person)	20,693,000 [c]	286,400	308,735	2,52,784
Disposable Income per head ($)	6,783.01 [c]	-	16,973.47 [d]	1,570

Note: a, Liu et al. (2013); b, Alam and Kulkarni (2016); c, National Bureau of Statistics of China (2013); d, NCC (2018b).

5.3.2. Regression analysis

Collinearity diagnostics indicate that collinearity is not observed in the explanatory variables of the regression models (Table 5.4). As described in section 5.2.2, the values of condition index smaller than 15 suggest that collinearity is not observed in the regression models. Collinearity should be concerned if the value of VIF exceeds 10. VIFs of variables in both Eq. 5.8 and Eq. 5.9 are smaller than 10. Therefore, no collinearity exists among variables in the regression models.

Table 5.4. Results of collinearity diagnostics (source: Wang, 2020).

Input variables	Condition index		VIF	
	Eq. 5.8	Eq. 5.9	Eq. 5.8	Eq. 5.9
Constant (C)	1	1		
Food (F)	3.31	3.57	1.65	1.28
Plastics (Pl)	4.14	6.05	1.77	1.04
Paper (Pa)	4.39	10.69	1.51	1.3
Textile (T)	9.29	-	2.06	-
Wood (W)	14.42	-	1.2	-

The model containing all the five variables (food, paper, plastics, textile and wood) was developed first as the full model (Eq. 5.13). The coefficient of multiple correlation (R) is 0.73 within a 95% confidence interval. The regression model is statistically significant (F = 21.79, P < 0.001).

$$LHV = -68.06\,F + 91.44Pa + 52.65\,Pl + 30.73\,T + 34.91\,W + 7{,}342.79 \quad \text{Eq. 5.13}$$

The coefficients of textile, wood and plastic are not statistically significant in this model (Table 5.5). Considering that plastics are one of the three major compositions in most MSW and the sum of wood and textiles accounts less than 10% in most cases (Table 5.2), the full model was simplified to the model without textile and wood (Eq. 5.14). The correlation coefficient R of Eq. 5.14 is 0.73, and it is statistically significant (F = 36.51, P < 0.001). The coefficient of plastic became statistically significant (p < 0.05) after the variables were reduced.

$$LHV = -72.42F + 83.20Pa + 67.90Pl + 7{,}669.08 \quad \text{Eq. 5.14}$$

Table 5.5. Linear regression coefficient results for the model contains five variables (source: Wang, 2020).

Variables	Unstandardized Coefficients		Standardized Coefficients	t-test ratio	Sig.	95.0% Confidence Interval for B	
	B	Std. Error	β			Lower	Upper
Constant (C)	7,342.8	808.5		9.08	0.000	5,737.5	8,948.0
Food (F)	-68.1	11.5	-0.53	-5.91	0.000	-90.9	-45.2
Plastics (Pl)	91.4	29.2	0.27	3.13	0.002	33.5	149.4
Paper (Pa)	52.7	36.4	0.14	1.45	0.151	-19.5	124.8
Textile (T)	30.7	58.1	0.05	0.53	0.598	-84.6	146.1
Wood (W)	34.9	67.8	0.04	0.51	0.608	-99.7	169.5

Food waste is negatively correlated with LHV ($P < 0.01$) for the coefficient in both Eq. 13 and Eq. 5.14). This negative contribution to LHV is discussed later. The reduction of the number of dependent variables did not reduce the coefficient of multiple correlations; keeping the three explanatory variables has not reduced the power of the model to explain the variability of the dependent variable (Table 5.6).

Table 5.6. Linear regression coefficient results for the model contains three variables (source: Wang, 2020).

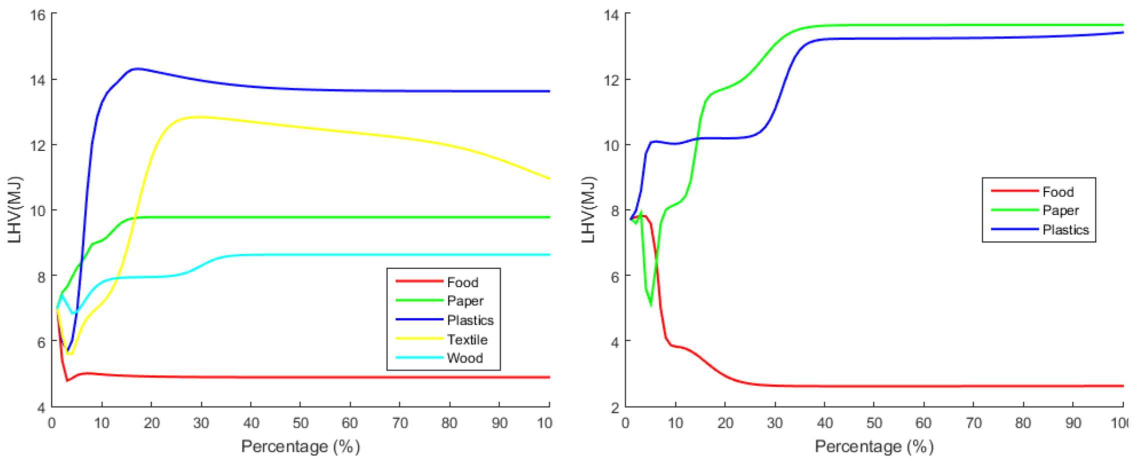
Variables	Unstandardized Coefficients		Standardized Coefficients	t-test ratio	Sig.	95.0% Confidence Interval for B	
	B	Std. Error	β			Lower	Upper
Constant (C)	7,669.1	716.1		10.71	0.000	6,247.7	9,090.5
Food (F)	-72.4	10.1	-0.57	-7.18	0.000	-92.4	-52.4
Plastics (Pl)	83.2	26.9	0.25	3.10	0.003	29.9	136.5
Paper (Pa)	67.9	27.8	0.17	2.45	0.016	12.8	123.0

5.3.3. The development of ANN models

Model ANN1 was generated using the same explanatory variables as regression model Eq. 5.13; its nodes in the hidden layer, learning rate and epoch were 31, 0.08 and 40, respectively. Similarly, model ANN2 was generated using the variables of Eq. 5.14. Without the two variables of wood and textile, fewer nodes (25) in the hidden layer and a lower learning rate (0.005) but more epochs (650) were required in ANN2 to model LHV: fewer epochs and a larger learning rate are required as the number of ANN input variables increases, as more information is provided for each sample point.

The contribution of each variable to the modelled LHV was estimated as illustrated in Figure 5.3. When the proportions of all waste compositions were set as 0, the models still return estimations of LHV at around 7 MJ/kg; we consider this to be the systemic bias of the model. The bias in ANN models is similar to the constants in the regression models (Table 5.5. and 5.6). This bias is partly derived from the nature of the input data: the waste categories we used as input variables did not account for 100% of MSW.

The estimated LHV is lower for MSW with a higher proportion of food waste, quite consistent with the results obtained from the linear regression analysis. Models generated using both methods indicate that food waste negatively contributes to LHV because of its high moisture content. Moisture content is significantly and positively correlated with the percentage of food waste ($r = 0.88$, $p < 0.001$). The negative contribution results from the increased latent heat required during combustion to evaporate the water. Plastics have the greatest contribution to LHV, as it has the highest energy content amongst all waste streams (Lin et al., 2015); however, its contribution to LHV is estimated to be smaller than paper when its proportion is greater than 15% in ANN2 (Figure 5.3b), or smaller than paper and wood when its fraction is smaller than 5% in ANN1 (Figure 5.3a). A similar situation arises in the regression models, as the paper has larger standardized coefficients than plastic. The possible reasons for these results are discussed in Section 5.4, considering chemical composition and the basis of the models.



(a) ANN1 (b) ANN2

Figure 5.3. Comparison of the contribution significance of each variable to LHV in ANN models (source: Wang, 2020). For the convenience of simulating, the unit of LHV was changed from kJ/kg to MJ/kg.

5.3.4. Evaluation and comparison of models

Model performance was evaluated using statistical evaluators (Table 5.7). The MAPE and SEE indices suggest that the ANN models perform better and could give more accurate results when being compared with the regression models. However, Std. deviation and range of the residuals (Min-Max) of the regression models are smaller than those of the corresponding ANN model. This indicates that the regression models may produce predictions that are closer to the measured LHVs. However, their performance may have been affected by non-linear cases. While ANN models may not give as precise predictions as the regression models, they are more robust in their ability to handle non-linearity.

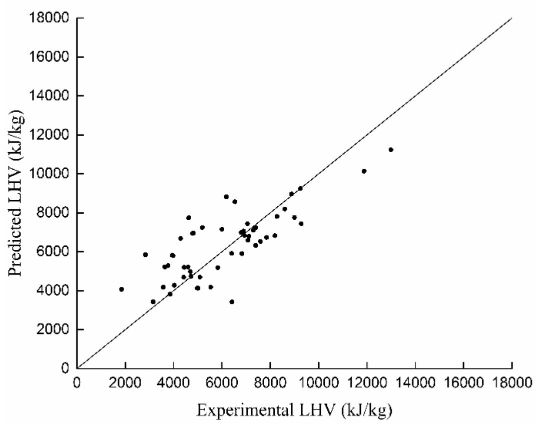
Table 5.7. The evaluation of predictive models (source: Wang, 2020).

Model	Residual(kJ/kg)			MAPE (%)	SEE (kJ/kg)
	Min.	Max.	Std. Deviation		
Eq. 5.13	-3,003.60	3,117.46	759.64	22.18	1,414.69
Eq. 5.14	-3,401.10	3,352.51	803.07	21.94	1,410.36
Eq. 5.3	3,308.02	14,527.80	2,670.02	197.39	10,544.60
Eq. 5.7	-4,393.90	2,264.43	1,684.63	24.09	1,918.04
Eq. 5.3*	-16.46	13.57	8.88	-	-
Eq. 5.7*	-69.42	67.79	-	18.16	1,111.44
ANN1	-2,183.37	4,261.35	1246.94	18.38	1246.94
ANN2	-2,960.11	4,171.90	1301.92	15.92	1,301.92

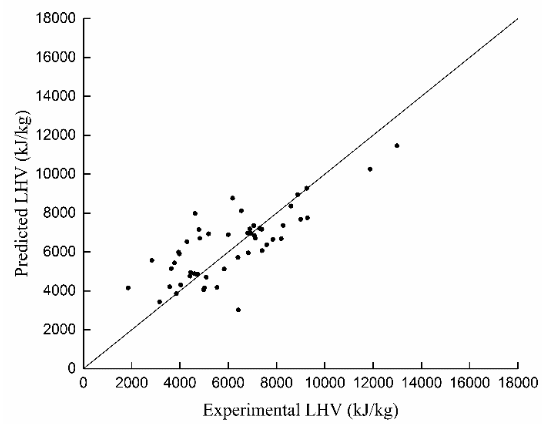
Note: *: The values of indicators are from the original research as shown in Table 1. -: Data deficient.

Of the two ANN models, ANN1 performs better than ANN2 based on the range and Std. deviation of the residual and SEE, but not MAPE (Table 5.7). As more complete information in terms of waste composition was input to ANN1, the behaviour of the model is expected to more closely match reality. On the contrary, the regression model which contains fewer explanatory variables performs better than the full model (Table 5.7). The restricted linear relationship between explanatory variables and the dependent variable reduced the likelihood of overfitting.

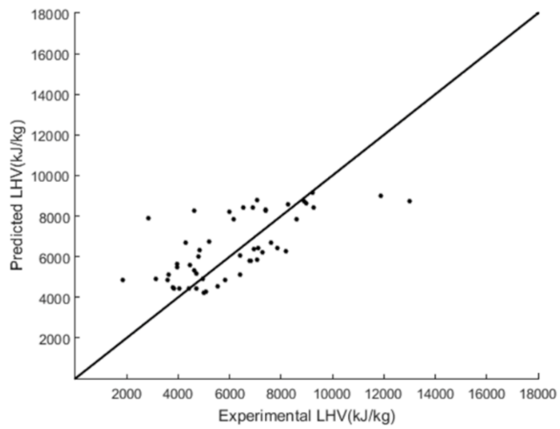
The performances of models built in this study were also compared with models in the literature containing similar explanatory variables, but built on data collected in specific geographic regions. These models did not perform as satisfactorily as documented in previous studies and performed less well than the models built in this study when the data we collected globally was used as input (Table 5.7).



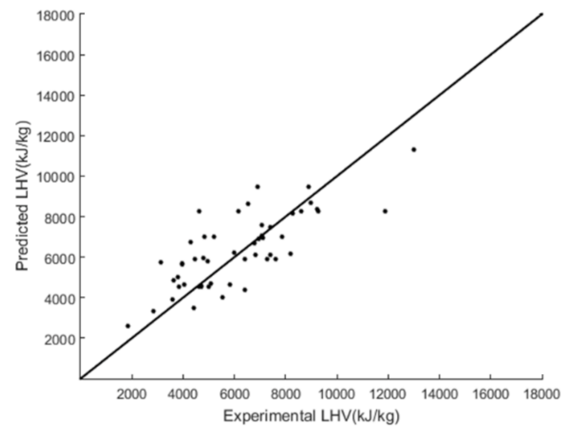
Eq. 5.13



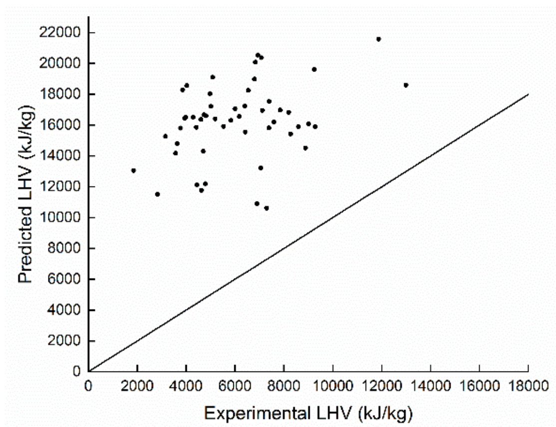
Eq. 5.14



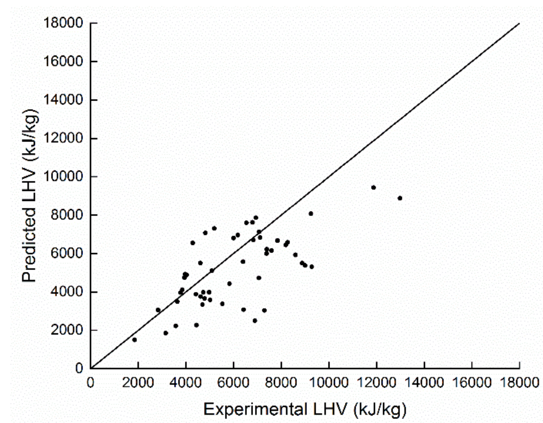
ANN1



ANN2



Eq. 5.3



Eq. 5.7

Figure 5.4. Comparison between predicted and experimental LHV (source: Wang, 2020). Lines in figures represent the best fit line where predicted LHVs equals experimental LHVs.

To better illustrate the performance of the models, predicted and measured LHV are compared in Figure 5.4. LHVs estimated using regression models are qualitatively closer to the best fit line than those estimated using ANN models. Models from the literature appear to be considerably less accurate than the models generated in this study; Eq. 5.3 in particular. Our models appear to be more universally applicable and they may have higher tolerance to less precise dataset.

5.4. Discussion

The primary aim of this study is to develop a generally applicable model to predict LHV based on the wet composition of MSW, using data that is relatively easy to acquire. A secondary aim is to compare two distinct methods to generate such models: linear regression and ANN. With those aims, our discussion is focused on three points: data quality and availability, the inherited diverse composition of MSW, and the implementation of models to support MSW management.

5.4.1. Data quality and availability

The quality of collected data clearly affects the performance of data-driven models, like linear regression and ANN. As mentioned earlier (Section 5.2.1), most data is based on the average percentage of MSW composition and LHV in a city or country. Though these point estimations could represent well the general MSW characteristics in that city or country, they cannot reflect the variability of this MSW composition. Moreover, is these point values are not representative of the broader sample mean, the resultant models could be seriously in error in aggregate terms. In statistical terms, this could produce the effect named ecological fallacy. This may influence the prediction accuracy of the models built based on the point of estimation. The ecological fallacy may even create a false model if individual data points were used.

The best way to verify the model is to verify the results using individual point data to further examine the relationship between waste composition and LHV. However, such data is difficult to obtain on a global scale. The best we can do is to separate data into training and validation datasets, and then to evaluate the models, performance using relevant statistical indices that measure the deviation between prediction and observation, such as MAPE, SEE and residuals (Table 5.7). Using such indices, we have shown that our models can estimate LHVs reasonably well, so that the relationship between the MSW composition and LHV can be verified. Furthermore, the results of LHV estimation using two totally different methodologies have both confirmed the positive contribution of plastic and paper waste and the negative contribution of food waste. Thus, we may be confident that the models have some discriminatory power, in terms of the physicochemical properties of the constituent parts of MSW.

In addition to the issues of using average values of MSW compositions and LHVs from cities, the data we collected are from various sources, so the quality of data is unavoidably inconsistent. The diverse characters in the same categories of MSW from different data sources or geographical regions (e.g. the characteristics of food waste) may have adversely affected the performance of our estimated models. However, collecting high quality primary MSW data, using consistent measurement and recording methodologies is like to be prohibitively time and labour consuming; and sometimes access to these data is simply not permitted. Under these circumstances (the absence of international standardisation), the use of publicly accessible secondary data is seen as a reasonable and necessary compromise to developing a model of wide geographical applicability.

The geographical distribution of the cities where these types of secondary data are available is uneven. In our case, most of our datasets were obtained from developing countries in Asia. We found that our models performed better than the models built based on the data only collected from local areas or small regions where the composition of MSW is less variable. This suggested that previous regression models built based on local and regional data only may not be generalized and applied with confidence elsewhere. Indeed, models generated from limited numbers of sites may be spatially skewed, and unrepresentative in terms of the MSW composition, reducing their utility both beyond and within their spatial bounds of applicability. Furthermore, the size of these samples is usually small, increasing the likelihood of overfitting the models and thus reducing their utility, even within their spatial bounds of applicability. Thus, the predicted LHV using our models for developed countries need to be interpreted with extra caution. The models may be further refined when additional data can be collected from developing countries/cities from regions such as Africa and Latin America.

Standardising the categories of MSW and the units of LHV measurement from a variety of sources and may also have introduced uncertainties. As mentioned in Section 5.2.1, food waste is classified as a standalone category in many cases but merged with other green waste in others (Alam and Kulkarni, 2016).

All the quality issues relating to input data affect the performance of the models generated. Based on the results of our performance evaluation, ANN methods seem to be good at accommodating a variety of data quality and return a model that can produce acceptable overall predictions. ANN also makes better use of more categories of data that may be considered insignificant in regression models (textile and wood) to further improve model performance.

5.4.2. Possible reasons for a higher contribution of paper

We established the models based on the waste categories commonly used in openly available statistics that most relevant to energy recovery during the incineration process. In this way, the models may be conveniently used to estimate the LHV of MSW at an initial stage in designing incinerators in any place in the world. However, our data does include the MSW dataset whose contents under the same or similar categories are diverse and sometimes inconsistent. An obvious example of some biodegradable waste has been noted previously (Section 5.2.1 and Section 5.4.1). The uncertainty of the diversified compositions within the categories may also have contributed to the inconsistency between our models and the commonly known heating values estimated based on the combustion theory. Our models mostly showed reduced contribution of plastic to the heating value of MSW compared paper; except for one model generated based on ANN (ANN1). We only identified limited studies showing similar patterns in their models. For example, based on the MSW samples and their LHVs collected by Lin et al. (2015) from the eastern and middle regions of China from 1996 to 2012, Ozveren (2016) derived a model depicting lower than expected contribution of plastics to LHV.

One possible explanation for this seemingly counterintuitive result could be that the presence of the types of plastics that produce relatively low LHV in the MSW dataset we collected are higher than generally expected. Among the commonly used types of plastics, the heating values of PVC can behalf of the other plastic ingredients such as LDPE, HDPE, PP and PS (Zevenhoven et al., 1997). The lower end of heating value among the range of the plastics materials (17.8MJ/Kg – 47.5 MJ/Kg) can be lower than the higher end of the heating values produced by paper (10.4MJ/Kg – 27.3MJ/Kg) or wood (14.6MJ/Kg – 28.6MJ/Kg) (Shi et al., 2016). Ragaert et al. (2017) indicated that PVC, as the major ingredient in some soft-packages, is usually not the target plastic ingredient for mechanical sorting for recycling/recovery and may be sent for incineration. Thus, it is quite possible that with well-organized recycling activities in a city, the proportion of plastic materials with a high energy content in the MSW to be incinerated may be reduced. Furthermore, the non-plastic ingredients may be attached on the surface of the plastic waste. Taking Nottingham city as an example, the main plastic materials remaining in the residual waste sent to incineration for energy recovery are packaging waste, plastic films, refuse sacks and carrier bags (Waste Research Limited, 2014); these types of plastics waste are usually contaminated and moisturized easily by other wet waste with low energy contents, such as food waste. In addition, moisturized plastic products are more difficult to dry than paper products. As a result, the proportion of the plastic materials with a high energy content in the MSW to be incinerated may be reduced. This may have been reflected in our models.

5.4.3. Application of ANN in LHV estimation

Given the varieties of MSW recycling and incineration methods as well as the types of waste generated based on the lifestyles and economic development among municipalities, the LHV contributed under each category may not be as unified as we originally considered according to the theory of combustion that describes a linear relationship between chemical elements and energy contents (Ryu et al., 2007; Giusti, 2009). From this aspect, the use of an ANN methodology that can accommodate non-linear relationships in establishing predictive models may be advantageous. This may be explained by the MAPE and SEE values of ANN models that showed an overall better performance amongst the models we built.

Implementing ANN to build models in predicting LHV is becoming increasingly popular (Khuriati et al., 2015). Indeed, it has advantages in learning from new data and taking potential non-linear relationship and ingredient variabilities into account, and handling multiple variables and processing big data, but these advantages of ANN have not been harnessed in previous and current research. Thus, the suitability of ANN in building LHV predictive models needs to be further assessed and discussed by increasing the number of input variables and the size of training datasets.

In addition, constructing a good network for a particular application is a non-trivial task. It involves choosing an appropriate architecture (the number of layers, the number of nodes in each layer, and the connections among nodes), selecting the transfer functions of the middle and output units, designing a training algorithm, choosing initial weights, and specifying the stopping rule (Alon et al., 2001). To a non-modeller, ANN appears to work as a black box. It is difficult to verify or validate the model based on knowledge of the chemical mechanism involved in incineration. Thus, we suggest applying ANN models only to predict the LHV for MSW composition, of which the percentage in each waste category is within the range covered by the dataset used to generate the model. Beyond (or rather below) these ranges, the explanatory power of the model decreases dramatically. As Figure 5.3 illustrated, The LHV changes little with the increased percentage of the waste composition after it exceeds a certain value, for instance, 35% for wood in ANN1 and 30% for food waste in ANN2. When the influence of variables on predicted LHV of MSW in

ANN models was analysed, the range of the proportion of waste compositions that the model can be applied needs to be specified. We randomly selected 100 from the 151 datasets for modelling while leave the remaining 51 datasets for validation to ensure the performances are evaluated based on the collection of data that are reviewed and organized under the same criteria. The waste composition ranges between training and validation datasets overlap. As such, the MAPE values indicate that ANN models in this study performed well, even slightly better than the regression models. However, the time invested to build ANN models may be several times longer than to build a linear regression model. Whether this level of improvement is worth the time may need to be further evaluated based on the benefits of improved prediction of the energy recovery efficiency in designing and operating the incinerator.

Another issue concerned in applying ANN in building the LHV predictive model is the size of samples required for model building. There is no answer to the question that how much training data is needed for machine learning to achieve a specific performance goal (Brownlee, 2017; Mitsa, 2019). The general concept is “the more the better” and too little training data results in poor approximation (Brownlee, 2019). Du et al. (2018) characterized the statistical efficiency of the conventional neural network with simple architectures. The results indicated that the testing error decreased with the increase of sample size from 500 to 2000. The acceptable performance of ANN models in this study may be because the number of datasets in this study is relatively small and only five variables included in models when compared to the implication of ANN in other fields. With the increase in the records of the types and LHV of MSW in the future, the application of ANN in building the LHV predictive model might be promising because ANN has the ability to adapting unlimited variables and its predicting performance increases with the growth input of training data.

5.4.4. Selection of model building techniques

Researchers often treat regression analysis and ANN as competing techniques for model building. However, these two methods may mutually assist each other, resulting in better decision making. Models built using these two distinct techniques can be validated and assessed against one another with the best performing model selected to support case-specific decision. In this study, both regression and ANN models have consistent performance. There is no single model that overwhelmingly out performs another. Either Eq. 5.14 or ANN1 can be selected to predict the LHV of MSW for researchers without or with ANN model building experience.

Whilst both techniques have their strengths and limitations (as mentioned in section 5.1.3 and section 5.4.3), neither can be guaranteed to always perform better than the other (Wang and Elhag, 2007). It is thus valuable to employ both techniques, selecting that which performs better, based on the specificities of the available training data. However, when big datasets and/or multiple independent variables are available, ANNs (ANN1 is recommended) are a better choice; otherwise, regression analysis (Eq. 5.14 is recommended) can be applied to save time. Considering the complex nature of MSW and likely improvements in MSW management in the future, ANNs have greater potential of application than regression analysis in this field.

5.4.5. Implication of the models in MSW management and policy making

The negative or low contribution of food waste to the overall LHV in both regression analysis and ANN illustrates that food waste is not suitable for incineration. Because of the lower carbon content and higher oxygen content in food waste (or organic waste in some cases), in comparison to materials like plastics, combustion is an ineffective means of disposal or energy recovery from food waste (Khuriati et al., 2015). The positive correlation between the proportion of food waste and moisture content of MSW in our datasets also confirms the unsuitability of incinerating moisture-rich food waste for energy recovery. Apart from the unrecoverable energy as latent heat, substances with high moisture content require high energy input to increase the unit temperature and this makes self-sustained combustion more difficult, more energy intensive and also readily produces incomplete combustion (Van Caneghem et al., 2014; Komilis et al., 2014). This latter increases the risk of producing pollutants such as dioxins and carbon monoxide (McKay, 2002; Tsai and Chou, 2006). Thus, reducing the proportion of food waste in the MSW for incineration may improve combustion efficiency and reduced pollutants by not only increasing LHV but also by enhancing the combustion processes. Hence, based on the results of our model, we recommend separating food waste or any other waste with a high moisture content at the point of collection, and applying alternative disposal methods for this type of waste, such as composting and anaerobic digestion (Papargyropoulou et al., 2014); otherwise, the pre-treatment to dehydrated MSW before incineration may be needed.

At the same time, efforts need to be invested in planning and decision making to assist the separate collection and treatment of food waste. We recommend that authorities set targets on the separation rate and biological treatment rate of food waste, as with the recycling rate that is set in many waste management regulations. Economic incentives and investments in technical facilities may facilitate the achievement of these target.

Plastic waste has a higher LHV than paper waste does, and incineration is the most suitable choice to treat plastic waste. However, the inherently diverse composition of plastic waste means that its LHV is highly variable. This might make the LHV of incinerated plastics waste unstable and make self-sustained combustion difficult. Therefore, we recommend that policies and actions should be undertaken to refine the classification of plastic waste, seeking alternative treatment options for plastics that have relatively lower LHV, such as those that can be easily moisturized, like plastic films and packaging waste. As mentioned in section 5.4.1, the inconsistency of waste classification is an important factor influencing the building and application of an international LHV predictive model. To this end, further effort would be welcome in the standardization of MSW classification.

5.5. Conclusion

In this chapter, we employ multiple linear regression and artificial neural network (ANN) techniques to estimate models to predict LHV, using 151 globally distributed datasets, describing the wet physical composition of MSW and measured LHV. The results show that models generated using the two methods exhibited acceptable and compatible levels of performance in predicting LHV. However, the ANN models proved to be more robust in their handling of datasets of diverse quality. The models built in this study can be applied with confidence at a regional level, especially in developing countries, to estimate the LHV of MSW. But where local data is available, we would advise building a model from this data. If feasible, we advise employing both regression and ANN techniques, comparing their relative performance and selecting that which perform best.

Our results also indicated that waste with higher moisture content, such as food waste, reduced the overall LHV of MSW, and thus reduced energy recovery efficiency for incineration. Separating this type of waste from waste to be incinerated at the collection point and applying an alternative method to treat it is recommended for more efficient and environmentally friendly MSW management. Furthermore, there would be value in refining separation and treatment practices for plastics, to divert plastics of low calorific value to other more beneficial treatment strategies.

Chapter 6 MFA of MSW Management Scenarios in Nottingham

6.1. Introduction

The evolution of waste management driven by the EU directives, and the performance of a waste management system can be measured by tracking and comparing historical and current status, as well as comparing to the targets (Zaccariello et al., 2015). Such comparisons can be made by using the methodologies of MFA with a series of representative indicators (Zaccariello et al., 2015; Masebinu et al., 2017). The results of MFA can also provide grounded inventory data for further assessment. Waste management hierarchy is the basis for building sustainable MSW management and correspondingly influence the choice of suitable indicators to evaluate the performance of the MSW management system. For example, recycling rate, recovery rate and landfill rate are frequently used as indicators to measure the performance of a waste management system (Zaccariello et al., 2015; Pomberger et al., 2017; Haupt et al., 2017a).

This chapter analyses and compares the material flows in three historical MSW management scenarios and a future MSW management scenario (details of scenario setting were addressed in Section 3.4). Then the performance of each scenario was assessed via indicators which appropriately selected based on the waste management hierarchy.

6.2. Materials and Methods

6.2.1 Selection of performance indicators

As listed in Table 6.1, five indicators based on the waste management hierarchy and targets set in waste management regulations were selected to evaluate the performance of MSW management in Nottingham. Waste prevention ranks the highest on the waste management hierarchy and is regarded as the most desirable option to divert waste from landfill (Gertsakis and Lewis, 2003); besides, reduction targets are set in local waste management plans. The effectiveness of waste prevention policies could be measured by calculating the waste generation per capita (GPC) (Desmond, 2006; Makarichi et al., 2018). Recycling is at the second top on the waste management hierarchy and recycling targets are often defined in waste regulations and management strategies (EC, 1999; DEFRA, 2007). Recycling rate (RCR) reflects the collective efficiency during sorting and selection steps to prepare the recyclable materials for reprocessing (Zaccariello et al., 2015). Source-separated collection, measured by separate delivery rate (SDR), is a critical component of effective MSW management and identified as the effective mean in landfilled waste minimization and resource utilization; it may increase the quantity and quality of well sorted waste (Zhuang et al., 2008; Rigamonti et al., 2009), so as to improve RCR (Tai et al., 2011; Ghani et al., 2013). Besides, recovering energy from waste which can be measured by recovery rate (RECR), is another important function of MSW management (Othman et al., 2013). The last option for waste management is landfill, which can be measured by landfill rate (LCR).

Table 6.1. List of indicators selected (source: Zaccariello et al., 2015; Haupt et al., 2017a; Makarichi et al., 2018)

Description	Acronym	Definition	Application
Waste generation per capita	GPC	The MSW generated by each resident in a specific place (in this case is Nottingham) in a statistical year.	GPC is the quotient of the total MSW generation divided by the total population in an area. When the collection coverage is 100%, the amount of waste generated equals the amount of waste collected.
Recycling rate	RCR	The ratio between the amount of waste prepared for recycling or the waste sent to producing secondary material and the total amount of waste generated.	It counts all material prepared for recycling from all sources including materials separated at source, at material recovery plant, and waste treatment and disposal plant, i.e. metal recovery from bottom ash at incineration plant.
Separate delivery rate	SDR	The ratio between the amount of waste collected as separated streams and the total amount of waste generated.	It counts all separately collected recyclables and green waste, either alone or co-mingled. This indicator only takes the separately collected waste streams into account, without considering the quantity or percentage of waste actually addressed to recycling and recovery.
Recovery rate	RECR	The ratio between the amounts of waste used for recovery options and the total amount of waste generated.	It counts waste sent to all types of treatment where energy is recovered, such as incineration with energy recovery and biogas production. Composting is usually not counted because no energy has been recovered, but landfill should be counted when landfill gas is recovered.
Landfill rate	LCR	The ratio between the amount of waste disposed in landfill and the total amount of waste generated.	It counts all waste sent to landfill including the rejected and residual waste from waste treatment facilities, such as the rejected waste from composting plant, bottom ash and fly ash from incineration plant.

Note: The sum of RCR, RECR and LCR is normally equal to or greater than 100% because the waste formulating bottom ash and fly ash counted twice by RECR and LCR. In calculation, the total amount of waste generated equals the total amount of waste collected when the collection coverage is 100%.

Generally, smaller values on GPC and LCR or higher values on RCR, SDR and RECR indicate a better performance of an MSW management system. To make the research results comparable to the targets which are usually set as the recycling and composting rates in waste management regulations, RCR has been adjusted to combine the share of recycled and composted waste. Waste sent to residual MRF is separately collected street waste, bulky waste and residual waste from CA site, but they are not included in the calculation of SDR because the waste from these sources are mixed waste with heterogeneous materials and the recycling potential of them is low.

6.3. Results and Discussions

Figure 6.1 and 6.2 illustrate the material flows in S2 and S3 (defined in section 3.4 in Chapter 3). KCS collected most of MSW in Nottingham, and through which MSW entered the following management practices. Followed by garden waste, residual waste was the dominant waste stream and majority of it was incinerated for energy recovery for both S2 and S3. While landfill was the major final destination of MSW in S2, recycling became the dominate one in S3.

The major improvements in S3 identified are the increase of SDR and the reduction of waste sent to landfill. Other notable improvements include the reduction of waste generation (from 129,814 tonnes to 115,170 tonnes) and the amount of incinerated waste (from 73,333 tonnes to 66,287 tonnes). Thus, the reduction of landfilled waste is achieved by measures in all levels of waste management hierarchy. The results of MFA are presented in detail in the following subsections to demonstrate in what way the values of those indicators are changed under the driving of waste management regulations.

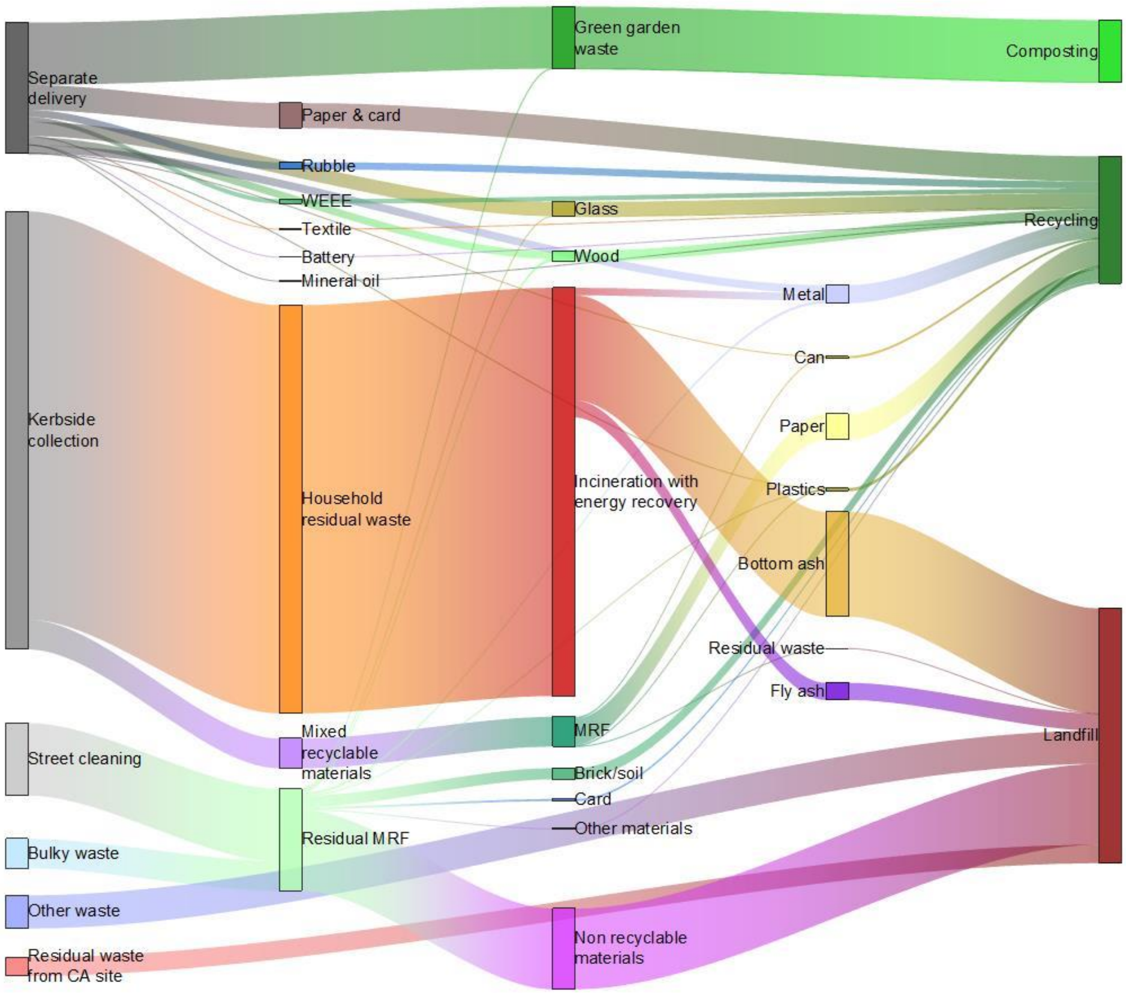


Figure 6.1. Diagram of waste flows in S2 (source: Wang, 2020).

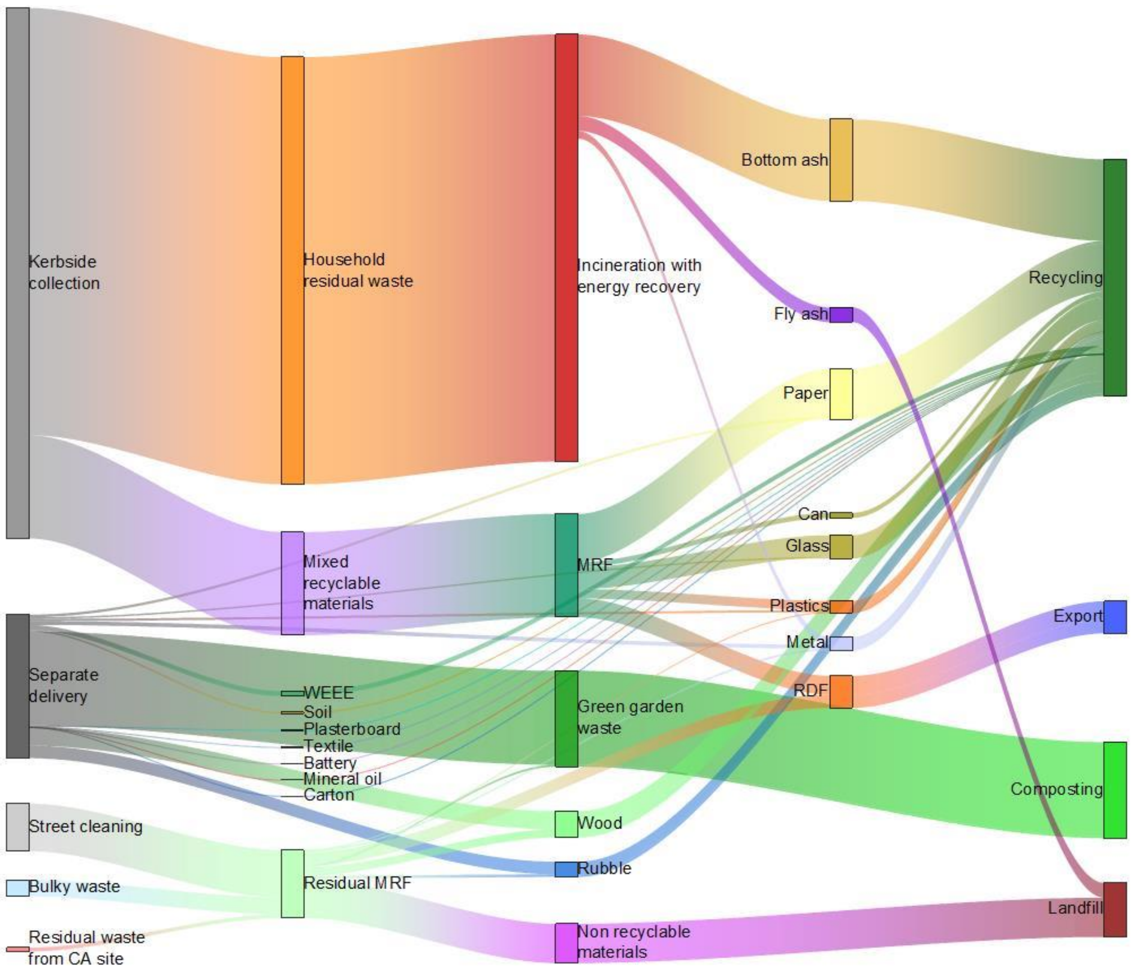


Figure 6.2. Diagram of waste flows in S3 (source: Wang, 2020).

6.3.1. Waste prevention

GPC increased slightly from 463 kg in 2001/02 to 466 kg in 2006/07, then decreased to 361 kg in 2016/17 (Figure 6.3), which was significantly lower than the national level (412 kg) (DEFRA, 2018a). This contributed to the total MSW reduction from 123,615 tonnes to 115,170 tonnes although population increased by 19.4% during the study period (Table 6.2, Figure 6.3 and 6.4). Towards the end of the period studied (after 2011/12), GPC stayed under the target 390 kg that was to be met by local government by 2025 (Figure 6.3).

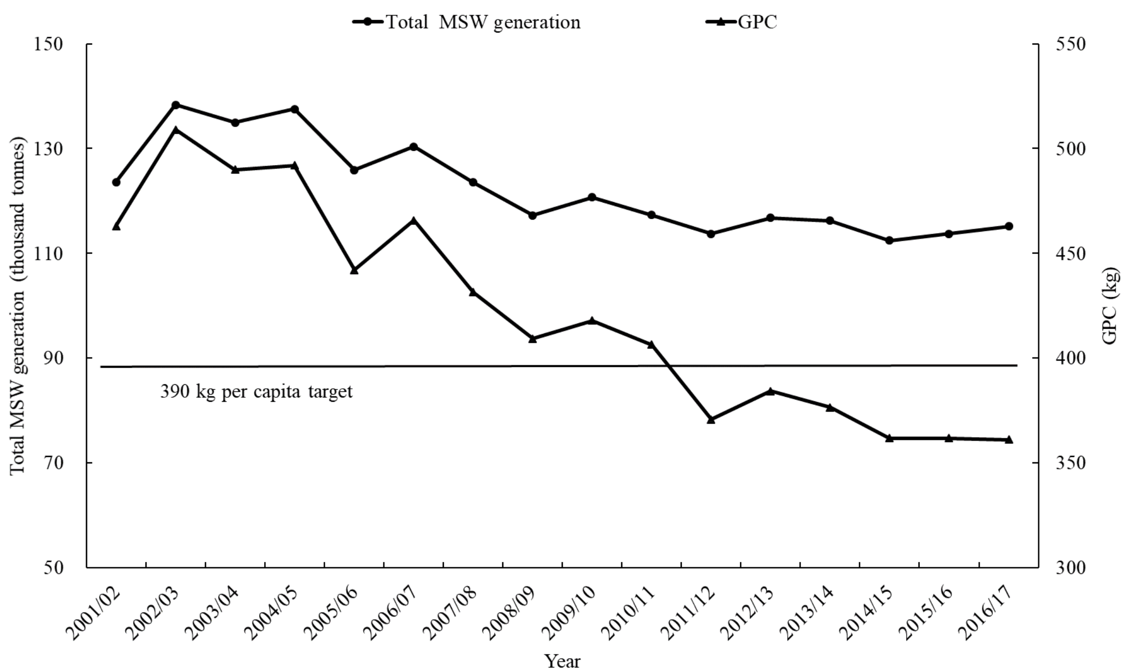


Figure 6.3. MSW generation during 2001/02 – 2016/17 in Nottingham (source: Wang, 2020).

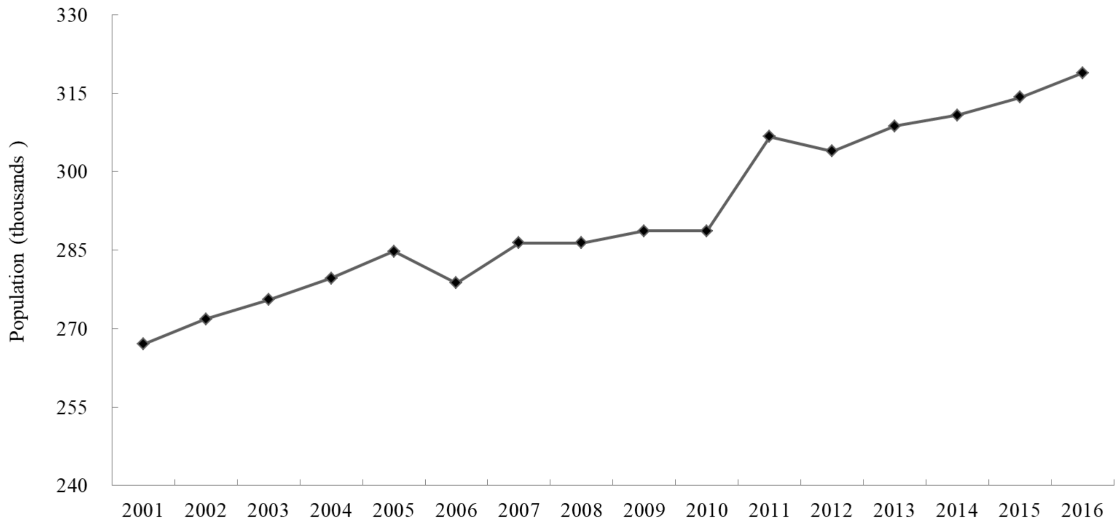


Figure 6.4. Population change in Nottingham from 2001 to 2016 (source: Wang, 2020).

The improvement of public awareness on waste prevention played an important role in waste reduction. Both national and local waste prevention programmes, such as WRAP, and public education initiatives raised public awareness to reuse products before their disposal. As a result, the waste generation in Nottingham significantly reduced under most waste categories and as a whole (Table 6.2 and Figure 6.3). The recent policy charging for single-use carrier bags, which was introduced in October 2015, reduced the generation of plastic waste as can be seen in Table 6.2. By contrast, a notable increase in textile waste was observed during the study period, which might be attributed to the development of fast fashion industry in recent years (Morgan and Birtwistle, 2009; Wicher, 2016; Perry, 2018).

Table 6.2. Results of the performance assessment of MSW management system for total MSW and selected classes of wastes (source: Wang, 2020).

Scenario	Waste category / Indicators	Metal	Garden waste	Plastics	Paper & Card	Textile	Glass	Wood	MSW
S1	Generated amount (t)	9,889	N/A	13,598	39,557	2472	11,125	N/A	123,615
	Percentage (%)	8.0	N/A	11.0	32.0	2.0	9.0	N/A	100.0
	GPC (kg/y)	37.0	N/A	50.9	148.2	9.3	41.7	N/A	463.0
	RCR (%)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	3.4 (4.6)
	RECR (%)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	40.7
	LCR (%)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	54.7
S2	Generated amount (t)	5,582	20,523	12,968	29,454	3,674	8,620	4,842	129,814
	Percentage (%)	4.3	15.8	10.0	22.7	2.8	6.6	3.7	100.0
	GPC (kg/y)	20.0	73.6	46.5	105.7	13.2	30.9	17.4	465.8
	Recycled amount (t)	3,599	11,171	496	9,571	193	2,672	1,935	22,831 (34,002)
	RCR (%)	64.5	54.4	3.8	32.5	5.3	31.0	40.0	17.6 (26.2)
	Recovered amount (t)	0	477	11,814	15,261	2,413	0	191	73,333
	RECR (%)	0	2.3	91.1	51.8	65.7	0	3.9	56.5
	Disposed amount (t)	1,983	8,875	658	4,622	1,068	5,948	2,716	45,786
	LCR (%)	35.5	43.2	5.1	15.7	29.1	69.0	56.1	35.3
S3	Generated amount (t)	4,312	16,212	10,708	16,582	7,161	6,115	4,294	115,170
	Percentage (%)	3.7	14.1	9.3	14.4	6.2	5.3	3.7	100.0
	GPC (kg/y)	13.5	50.8	33.6	52.0	22.5	19.2	13.5	361.2
	Recycled amount (t)	2,681	14,899	1,880	7,881	95	3,625	4,110	36,760 (51,659)
	RCR (%)	62.2	91.9	17.6	47.5	1.3	59.3	95.7	31. 9(44.9)
	Recovered amount (t)	0	1122	8623	7808	6940	0	92	71,267
	RECR (%)	0	6.9	80.5	47.1	96.9	0	2.2	61.9
	Disposed amount (t)	1,631	191	205	893	127	2,490	92	8,422
	LCR (%)	37.8	1.2	1.9	5.4	1.8	40.7	2.1	7.3

Continue of Table 6.2

Scenario	Waste category / Indicators	Metal	Garden waste	Plastics	Paper & Card	Textile	Glass	Wood	MSW
S4	Generated amount (t)	4,312	41,070*	10,708	16,582	7,161	6,115	4,294	115,170
	Percentage (%)	3.7	35.7*	9.3	14.4	6.2	5.3	3.7	100.0
	GPC (kg/y)	13.5	128.8*	33.6	52.0	22.5	19.2	13.5	361.2
	Recycled amount (t)	3,149	35,079*	3,900	11,768	1,050	4,967	4,110	38,847 (73,327)
	RCR (%)	73.0	85.4*	36.4	71.0	14.7	81.2	95.7	33.7 (63.7)
	Recovered amount (t)	0	13,007*	6,808	4,814	6,111	0	184	51,594
	RECR (%)	0	31.7*	63.6	29.0	85.3	0	4.3	44.8
	Disposed amount (t)	1,163	0*	0	0	0	1,148	0	4,093
	LCR (%)	27.0	0*	0	0	0	18.8	0	3.6

Note: values in brackets () represent the quantity and percentage of recycled waste plus the composted green garden waste. *: The sum of food waste and garden waste in S4. GPC: waste generation per capita, RCR: Recycling rate, RECR: Recovery rate, LCR: landfill rate.

Social and economic developments are other possible factors affecting waste generation and reduction in a number of ways. GPC is generally regarded as positively correlated with the income, population and population density (Dahlén et al., 2009, Das et al., 2019). The average earnings without taking inflation into account increased during the study period; however, the 'real' earnings adjusted for inflation have declined in every year since 2009 and are at levels last seen in the early 2000s (NCC, 2015). The decrease of 'real' earnings seems potentially reduced the GPC, but positive correlation between the number and percentage of workless households and the GPC was observed (Figure 6.3 and 6.5). Besides, the GPC declined steadily during the study period and was remarkably lower in 2016/17 than that in 2001/02 and 2006/07. The GPC is not always correlated with income because decoupling of income and waste generation might occur (Namlis and Komilis, 2019). Some researchers also reported that the correlation between income and GPC sometimes is weak in developed countries (Dahlén et al., 2009; Passarini et al., 2011; Namlis and Komilis, 2019), even in developing countries (Miezah et al., 2015). The population and population density increased from 278,700 and 37 persons/ha in 2006 to 318,901 and 42 persons/ha in 2014, but they had not resulted in the increase on waste generation. The average family size increased from 2.2 persons/household to 2.4 persons/household from 2006 to 2016. It is believed that bigger family size might lead to smaller GPC (Miezah et al., 2015). The social and economic factors influence waste generation from different directions. Overall, the GPC showed a decreasing trend during the study period.

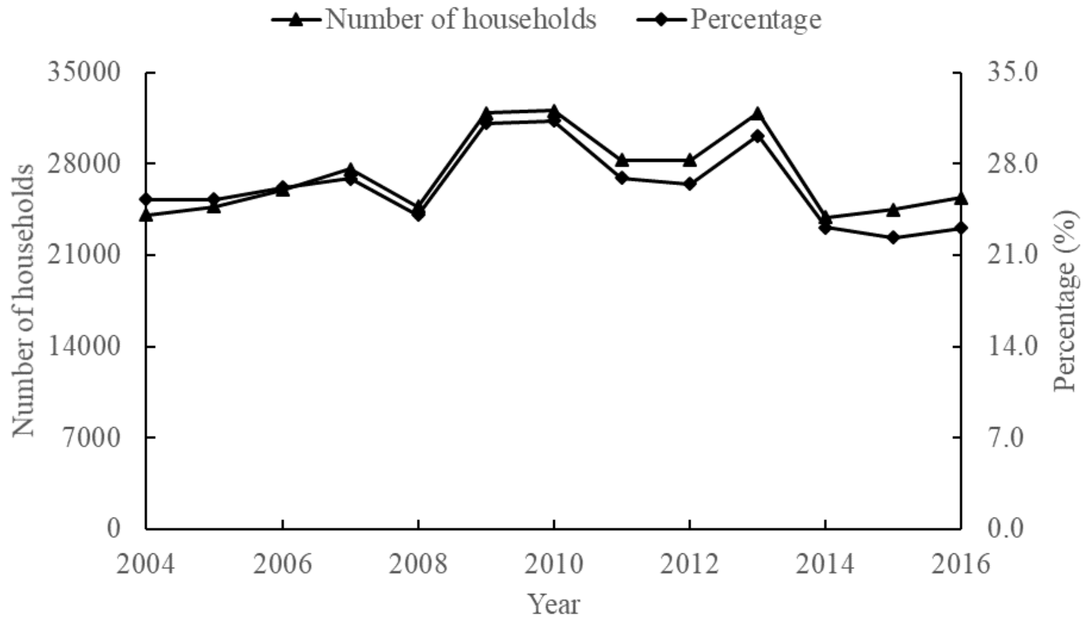


Figure 6.5. The change of workless households from 2004 to 2016 (source: NCC, 2017a).

6.3.2. Separate delivery

SDR in Nottingham increased from 22.2% in 2006/07 to 33.3% in 2016/17 due to the introduction and expansion of KCS, and resulted in the improved recycling share, and a high interception of garden waste (90.0%) (Figure 6.1 and 6.2). Kerbside collection has been demonstrated to be the most efficient and sustainable separate collection scheme (Tucker et al., 1998; Larsen et al., 2010). It was introduced to Nottingham in 2002 for separating paper at source. Thereafter, the categories of material collected in the scheme and spatial extent of the scheme were increased year by year. The expansion was so significant that in 2008, the local authority started to offer three types of wheeled bin for waste containment to households for free for separating recyclable materials and garden waste at sources (Figure 4.1 and Figure 6.6). From 2006/07 to 2016/17, the percentage of households served by KCS increased from 4.7% to 100%, and the proportion of households received separate garden waste collection increased from 32.7% to 74.4%. Other types of containment, such as orange survival bags, communal bins, refuse bins and plastic sacks were offered in areas not covered by KCS but the number of bring sites where recyclable materials used to be collected reduced from 88 to 17. It is also noted that the quantity of street waste and other waste received by residual MRF site all reduced. The improvement of source-separated collection in the past decades was directly related to the implementation of KCS in Nottingham.

(a) Recyclables (b) Garden waste (c) Residual waste

Figure 6.6. Bins used for separating and storing waste in KCS (source: NCC, 2018c).

The SDR of textiles was very low and reduced from 5.2% to 1.3% during 2006/07 – 2016/17. Textile is not included in the waste categories collected by KCS. Recyclable textile was usually collected at bring sites and CA sites. The reduction of the number of bring sites may have reduced accessibility to facilities for textile recycling without replacement, as the average distance between households and bring sites increased. Further, usually the second-hand textile products that are reusable with minimal fixation can be accepted in charity shops, rather than being brought to the recycle centres; clothes that cannot be worn any longer may be put in a residual bin and sent to the incineration plant intuitively by the owners, while in fact, these disposed unwearable cloth could have been used as wiping and polishing cloth, or reprocessed into textile products such as nonwovens and mats (Wang, 2010). Recycled polymers could be used as matrices in glass fibre reinforced composites or to make producers in a moulding process (Wang, 2010). Recycling textile can contribute to reduce the environmental burden compared to using virgin materials (Woolridge et al., 2006). However, for the time being, the increased textile waste has been used more for the energy recovery (RECR 96.90% for S3, Table 6.2).

6.3.3. Recycling and composting

RCR in Nottingham has significantly increased from 3.4% in 2001/02 to 17.6% in 2006/07, then to 31.9 % in 2016/17. The values are higher when including the composted waste (Table 6.2), but another over 5% of waste needs to be recycled or composted to reach the national and local targets of recycling and composting 50% of household waste by 2020. The recycling and composting rate in 2016/17 in Nottingham, taking recycled bottom ash into account, was equal to the national level of 44.9% which excludes the recycled bottom ash (DEFRA, 2017). It is possible to meet the target if separate source collection is further improved. On the other hand, based on the relatively low GPC (Section 6.3.1), we cannot exclude the possibility that public awareness of prevention and reuse before recycling contributed to the declined proportion of recyclable materials in MSW. The positive effort in prevention is also reflected in the declined amount of glass, paper and cardboard with increased RCRs.

The improvement of public awareness on waste recycling and the improved technologies and techniques on waste collection, sorting and treatment driven by the waste management regulations are the factors contributing to the improvement of RCR. The combination of the KCS and public education on waste recycling leaded the improvement of waste separation at source, especially for garden waste, thus the improvement of RCR. Recycling materials from residual waste through residual MRF and bottom ash utilization further improved the RCR. However, the improved RCR often sacrifices the quality of secondary materials due to the accumulation of hazardous substances (Kral et al., 2013), and the accumulation of hazardous substances is more likely to happen when materials are recycled from residual waste or bottom ash. Apart from improving the public awareness on waste recycling and classification to reduce the contamination of recyclables, more attention should also be paid on improving the quality of secondary products rather than meeting the quantitative targets.

RCRs of all waste categories, except textile, were maintained if not improved (based on the RCR values in S2 and S3, Table 6.2), although still a large fraction of metal and glass were addressed to landfill or recycled as aggregates with bottom ash. To further reduce the landfill volume, plans and actions relating to recycling textile, glass and metal may be needed in future waste management. Unrecyclable plastic materials such as plastic film, packaging waste and single-use carrier bags account for a big proportion in plastic waste, making the RCR of plastics low (3.8% in S2 and increased to 17. 6% in S3). Most of them were treated for energy recovery in both historical states of MSW management. Since plastic waste normally has a high energy content, recovering energy from it is deemed to be an appropriate way of disposing it.

Garden waste accounted for around 15% of MSW in Nottingham. It shares the highest SDR among all waste categories in both S2 and S3. Most garden waste was separately collected at source and sent to farm for fertilisation after being composted. The adoption of composting did reduce the quantity of BMW sent to landfill, but the GHG emission factor of composting is four to five times higher than anaerobic digestion (Fong et al., 2015). Capturing methane from composters or adopting advanced technology to treat garden waste is recommended for reducing the global impact of waste management.

Processing efficiency of separately collected mixed recyclables in MRF reduced from 99.6% in 2006/07 to 81.8% in 2016/17 as the KCS expanded (Table 6.3). This most likely is the results of the misclassification at sources, which lead to a high contamination of 14.2% in comingled recyclables. This misclassification might be due to the comparatively low level of outreach or education of households that were new to the extended KCS. This, in combination with the introduction of additional types of recyclable materials and collection bins, might have confused citizens regarding the ways of classifying and recycling the materials. Thus, an increased portion of unrecyclable materials was mixed with the comingled recyclable collections (BBC, 2017a), and around 17% of the materials placed into the residual waste bin were actually recyclable (Table 6.4). Educational campaigns combined with economic incentives or punishment to improve waste classification are recommended, to improve the quality of recyclable wastes and thus RCR. On the other hand, in S3, the increased misclassified unrecyclable wastes were sent for producing RDF as a means for energy recovery, instead of being sent to landfill. The development of new technology somewhat made up for the lack of sufficient outreach in this way.

Table 6.3. Recycling and rejection rate in two types of MRF during 2006/07 – 2016/17 (source: Wang, 2020).

	Residual MRF					MRF			
	Input (t)	Recycling (%)	RDF (%)	Landfill (%)	Compost (%)	Input (t)	Recycling (%)	RDF (%)	Landfill (%)
2006/07	18312	20.8	0	79.2	0	5350	99.6	0	0.4
2007/08	20099	18.9	0	80.5	0	10748	96.1	0	3.9
2008/09	16007	19.0	0	72.9	0.5	13060	92.9	0	7.1
2009/10	15076	25.1	1.6	67.0	8.2	13610	91.2	0	8.9
2010/11	14597	19.6	4.4	70.7	7.3	18206	89.3	0	10.7
2011/12	13565	18.7	6.3	74.9	6.0	17982	88.7	11.3	0
2012/13	11335	18.3	14.5	59.4	7.8	16297	79.2	9.5	11.4
2013/14	11557	22.6	19.9	52.8	4.7	19049	82.5	8.8	8.8
2014/15	10974	17.9	31.0	46.1	5.1	17780	84.1	7.9	7.9
2015/16	9322	18.6	24.2	49.7	4.3	17150	83.0	16.2	0
2016/17	10564	16.9	19.5	57.8	2.9	16036	81.8	18.2	0

Table 6.4. Percentage of recyclable materials in residual waste which could be separated at source (source: NCC, 2013a)

Recyclable materials	Percentage (%)
Paper	3.5
Card & cardboard	2.6
Plastics	3.2
Glass	2.9
Metal	2.0
Textile	1.9
Garden waste	0.9
Total	16.9

6.3.4. Energy from waste

The implementation of EfW incineration and RDF leads a high RECR in Nottingham, 56.5% and 61.9% in both historical situations (Table 6.2). Residual waste was incinerated in Eastcroft EfW for recovery energy. This has contributed remarkably to reducing the volume of waste to be landfilled and improving the performance of the MSW management system in Nottingham. The facility produces nearly 20 mega Watts of thermal energy displacing non-renewable methods for generating electricity and serving around 4,600 homes for heating (FCC Environment, 2015). This contributed to the 3% of the energy consumed in Nottingham in 2006, making it the most energy self-sufficient city in the UK at that time (NEP, 2010). The production of RDF is considered a good way to enhance energy recovery. The proportion of waste separated to produce RDF was increased to 4% in 2016/17.

However, it is undeniable that over 50% of MSW in Nottingham was directly incinerated without sorting in 2016/17. Food waste made the greatest proportion of the incinerated residual waste (33.4%) for energy recovery. However, food waste is not suitable for incineration because its high moisture content reduces the calorific value of the waste mixture (Zhang et al., 2010; Bai et al., 2012) and increases the chances of incomplete combustion that produces pollutants such as dioxins and carbon monoxide (McKay, 2002; Tsai and Chou, 2006). Food waste may be better used for making fertilizers after composted, which also produces biogas for energy production (World Energy Council, 2016). Therefore, more effort should be made to separate food waste from residual waste to improve the energy recovery efficiency. By doing so, the food waste is also dealt with using a more favourable (composting or anaerobic digestion) methods based the waste management hierarchy.

6.3.5. Landfill

The improvement of recycling and recovery, also prevention, potentially lead to a remarkable reduction of LCR in Nottingham from 54.7% in 2001/02 to 35.3% in 2006/07 and further to as low as 7.3% in 2016/17 (Table 6.2). In the S3, only the residual waste from residual MRF that cannot be recycled or processed to RDF was landfilled. It is believed that with continued improvement of separated source collection to prevent cross contamination, the LCR can be further reduced to approach the zero landfill target set by the Nottingham Waste Strategy 2010-2030.

6.3.6. MAF and evaluation of the future scenario (S4)

Future scenario (S4) is set based on the gaps identified from the MFA results of S2 and S3. In S4, food waste was separately collected, textile is separated at source, composting technology is upgraded from open windrow composting to anaerobic digestion, residual waste is sorted at residual MRF before sent to Eastcroft EfW. By doing these, the material flows in S4 will be vastly changed. Although the volume of each changed waste stream cannot be accurately calculated because the application of new waste management strategies affected by multiple quantifiable and unquantifiable factors related to the dimensions of policy, economy, society, culture and environment, the change direction could be predicted. Following the residual waste, organic waste including food waste and garden waste will be the second biggest waste stream. More materials will be recycled and get back to the circle of urban metabolism, and more energy will be recovered. Besides, materials recovered from residual waste might be less contaminated since food waste is separated at source.

To give an example of the waste flows in future scenario, some assumptions have been made as follows. 90% of food waste and reusable textiles are assumed to be separated at source considering the SDR of some waste streams, for instance garden waste, could reach 90%. The composting of garden waste is replaced by controlled anaerobic digestion to produce biogas in addition to fertilizer. The biogas is assumed to be produce with a yield of 20% by weight, of which, 63% is methane (Zaccariello et al., 2015; Turner et al., 2016). The collection of biogas for energy generation may reduce the GHG like methane being directly released into the atmosphere as it would be during the composting process. Residual waste is admitted to MRF first to recycle materials as much as possible. In this process, 80% of recyclable materials in residual waste is assumed to be recycled by considering that the processing efficiency of mixed recyclables in MRF is over 80%. After separating these recyclable materials, 80% of unrecyclable but combustible materials with a high LHV, namely plastics, textiles, paper and card, and 20% of combustible materials with a lower LHV, namely garden waste, food waste and combustible miscellaneous are processed to produce RDF. Then the remaining combustible residual waste is incinerated for volume reduction and energy recovery. Non-combustible waste is sent to landfill. Bottom ash from the incinerator is recycled for aggregates or road construction. The waste flows in S4 is illustrated in Figure 6.7.

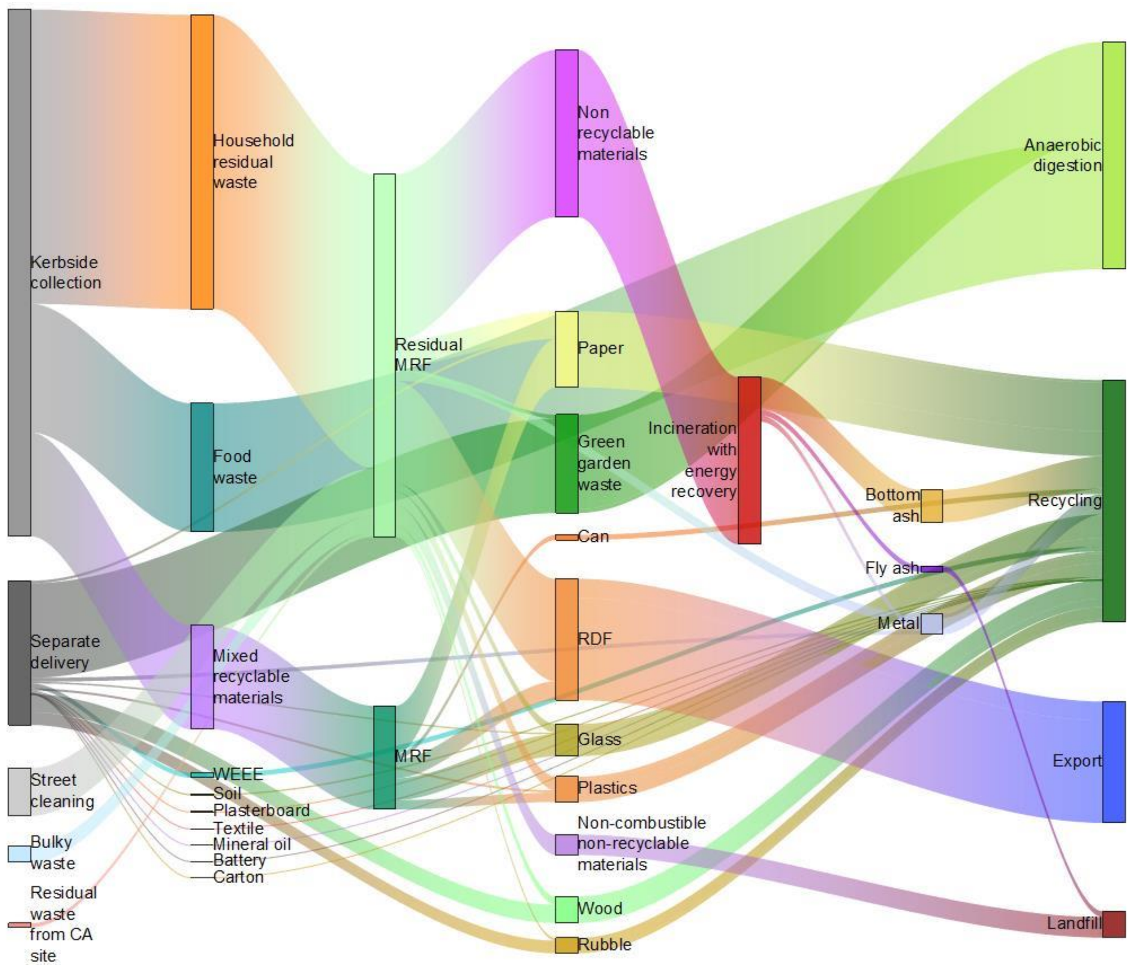


Figure 6.7. Diagram of waste flows in S4 (source: Wang, 2020).

By taking these actions, the SDR of the MSW management system can be improved to 51.4%. In this way, the total recycling and composting rate can reach 63.7% and the LCR will be reduced to as low as 3.6% (Table 6.2). The RECR is reduced to 44.8%, 13.4% of which is derived from the organic waste treated in anaerobic digestion. As the reduction of

RECR indicates only the reduction of the amount of waste treated for energy recovery, the decreased volume may not be viewed as negative because the quality of waste treated in energy recovery process (heating value) is expected to be improved due to the production of RDF and biogas. The good results in terms of the recycling and composting rate obtained by moving from S3 to S4 demonstrate a waste management with better performance can be achieved by improving separating at source as well as bettering sorting process.

The MFA results of S4 demonstrate that combination of separating food waste and textile at source and upgrading waste treatment technologies can divert waste from landfills, and thus improves the recycling and composting rate and the performance of the MSW management system. It is also believed that applying anyone of newly introduced practices in S4 could help to improve the performance of MSW management, but separate collection makes the biggest contribution because more materials enter into the recycling and recovery process through it and less investment of energy and labour are input for sorting. However, multiple actions need to be undertaken to meet the high recycling and composting target.

6.3.7. Opportunities and challenges for future improvements

Waste prevention is the key to decouple the correlation between economic growth and waste generation. Absolute decoupling between waste growth and economic growth has not been demonstrated in Europe so far (Zorpas et al., 2014), but the reduction on the number and percentage of workless households did not result in a growth of GPC in Nottingham. Waste prevention actions such as food waste prevention and establishment of the reuse or exchange networks underpin the waste reduction in Nottingham and should be promoted in future MSW management.

Enhancing source separation seem to play an important role in improving the performance of MSW management in Nottingham, and the public participation will be the most important factor influences the MSW management. On the one hand, most citizens in Nottingham have been well educated for waste minimization, separation and recycling, and kerbside collection system have been well established and implemented. Households are actively involved in the separation and collection process. This is facilitating the separate collection of food waste and textile. On the other hand, the incorporation of the separate collection of food waste changes the current waste management habits of households. The willingness of public to change will be a decisive factor determining the success of this strategy. The study conducted by Bernad-Beltrán et al. (2014) in Spain demonstrated a high willingness to separate food waste if supportive facilities, for instance, bins are provided by local authority. Besides, adding more waste categories in the KCS list causes confusion easily and increases the difficulty and inconvenience of householders to separate waste at source. This might hinder the public engagements in waste management, and potentially increase the contamination of separated recyclables, hence reduce the efficiency of sorting and processing and the quality of recycled materials. Therefore, public education and facilities supporting source separation should be strengthened.

Economic development provides opportunities, as well as challenges on MSW management. Local authorities in numerous countries seek partnerships with private enterprises to cut the increasing cost and enhancing the efficiency of MSW management (Massoud and El-Fadel, 2002). By-products from MSW management bring profits to waste management entities, but the limited market for these products and the poor source separation of waste might have constrained the entry of private entities into the waste management sector (Banerjee and Sarkhel, 2019). At the meantime, increased separated streams requires more investment on technologies, facilities and workers to treat or process them. This will increase the financial burden on local government, as well as entities. Therefore, the improved MSW management should be associated with the expansion and management of the market for secondary products from waste management sector and cost reduction measures such as ensuring the low transaction costs through improving the transparency and effectiveness of market signals (Banerjee and Sarkhel, 2019).

To introduce MRF for the pre-treatment of the waste that was sent to incineration could potentially increase the RCR by recovering recyclables from residuals waste. However, the quantity and quality of recycled materials will be reduced because recyclable materials are contaminated easily by mixed waste. Alternatively, production of RDF might be possible to improve the RECR of the MSW management system.

6.3.8. Uncertainties and limitations

National average value of the household waste composition in 2006 and local waste composition in 2013 were acquired to present the waste composition in Nottingham in 2006/07 and 2016/17 respectively due to the data unavailability. It is acknowledged that using this data could introduce uncertainties of the MFA results. The variation on waste composition might change the values of indicators assessing the management on specific waste streams, for instance, paper and plastics, but it does not change the results of the evaluation of the MSW management system as a whole.

The indicators selected in this study well assessed the performance of the MSW management following the rule of the waste management hierarchy and the targets in waste regulation. However, they have limitations to assess the sustainability of MSW management system. An MSW management system with higher RCR is not necessarily more sustainable than the one with lower RCR because the actually recycled secondary material is also related to the efficiency of reprocessing and the replacement of primary materials (Haupt et al., 2017a). Besides, the quality of recycled materials is not guaranteed with the improved RCR. Kral et al. (2013) pointed out that high recycling rates often contradict high product qualities. A comprehensive assessment on the sustainability of MSW management should always be complemented with a life cycle analysis, and more attention should be paid on the quality of secondary products. Even though, the improvement indeed reflects a level of resources utilization efficiency that has positive consequences of environmental conditions. The improvement of waste collection and recycling system that leads to the reduction of landfilled waste is a reflection of the effectiveness of the EU directives on the improvement of the MSW management. Furthermore, the MFA results provides well-grounded inventory for LCA.

Because of data deficiency, analysis of material flows in each management process and the material flows at substance level have not been performed at this stage. This might reduce the comprehensiveness of this study. The missing parts might affect the assessment on the qualitative aspects of MSW management. For example, analysing the material flows at MRF could help to identify the quality of recycled materials and the quantity of materials actually enter the reprocessing process (Haupt et al., 2017a). Performing MFA on substance level identifies and quantifies the sources, pathways and sinks, as well as the links among them and the allocation of environmental burdens, of a concerned substance which might have specific impacts on environment or human health, such as nitrogen, phosphorus and heavy metal.

Waste prevention is emphasized in this study and collection is the most important and the first step of MSW management, but the factors, especially the spatial distribution of these factors, influencing waste generation has not been explored at this stage. This is extremely important for decision making on waste prevention. Spatial analysis of MSW generation and collection could support a better decision making on waste management.

Besides, only one future scenario integrating all identified possible solutions has been set and assessed. This might not highlight the contribution of the most promising measurement. Separating possible solutions in different scenarios and assessing their performances could improve the persuasiveness and the quality of this study. And it is more realistic to undertake one measurement at a time.

6.3.9. Future endeavours

In the future work, more data will be collected to complete the MFAs both at good and substance levels. The spatial distribution of MSW generation and collection will be explored. The factors influencing MSW generation and management will be investigated. With more detailed data on MSW management, more scenarios will be set to identify the potential improvements and the most promising measurements for the MSW management in Nottingham.

6.4. Conclusions

The comparison between historical states of MSW management in Nottingham suggests that the policies and regulations implemented to respond to EU Directives have considerably reduced the waste generation and improved the recycling and energy recovery from waste for the city, but the loopholes in treating the textile waste and food waste were identified. The EU Landfill Directive only focus on the reduction of the landfilled materials. Fulfilling the target does not mean the waste management system performs very well. The implementation of Waste Framework Directive which established the “waste management hierarchy” improved on the EU Landfill Directive by focussing on the performance of the whole system. Nottingham City Council may now consider that a more sophisticated strategy goes beyond the objective of fulfilling the target of the EU Landfill Directive. The system can be further improved by better allocating wastes in the upper layers of the waste management hierarchy and in the layers where the wastes may maximise its potential to be converted into resources (energy and materials).

Waste separation at source is the key to improve the efficiency of waste treatment methods. Hence, at all layers of the waste management hierarchy, effective public education and supportive facilities on waste classification are recommended to accompany the expansion of KCS and the future separation of food waste, so as to reduce the misclassification of the recyclable and recoverable materials. Besides, economic instruments should follow up to manage the secondary products from waste management sector. Waste generation could also be further reduced by decoupling the correlation between economic development and waste generation through waste prevention actions.

Chapter 7 LCA of MSW Management in Nottingham

7.1. Introduction

LCA has been extensively applied to evaluate the environmental burdens associated with MSW management (Fernández-Nava et al., 2014; Yay, 2015; Milutinović et al., 2017; Coelho and Lange, 2018). However, 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 transformation of MSW management 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 process for the study site, or whether there has actual progress towards a more environmentally friendly waste management strategy (Poulsen and Hansen, 2009). Besides, Waste prevention is regarded as the most important technique to be applied to MSW management (Tseng et al., 2018), but seldom assessed as an integrated part in an MSW management system in previous LCA studies. Therefore, this chapter evaluates the GWPs from MSW management in the three historical scenarios and the future scenario (Section 3.4) using LCA based on the MFA results in Chapter 6.

7.2. Methodology

7.2.1. Goal and scope

The goal of the LCA in this chapter was to quantify and compare the GWP of three historical MSW management scenarios at three development stages in Nottingham, and a future scenario in response to the EU directives. The functional unit is defined as the treatment of one tonne 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.

7.2.2. System boundary

The overall system addressed in the present chapter is illustrated in Figure 7.1. It contains all waste management processes including collection, transfer, 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 (DEFRA and DECC, 2002, DEFRA and DECC, 2007, DEFRA and DBEIS, 2017).
- Waste collection.

- Transport to/between treatment facilities.
- Direct emissions from waste; for example, CO_2 from waste incineration.
- Avoided GHG emissions by materials recycling and energy recovery.
- Environmental burdens from the operation of the CA and bring sites were excluded due to data deficiency.
- Bottom ash from incinerator was not considered as a source of GHGs.

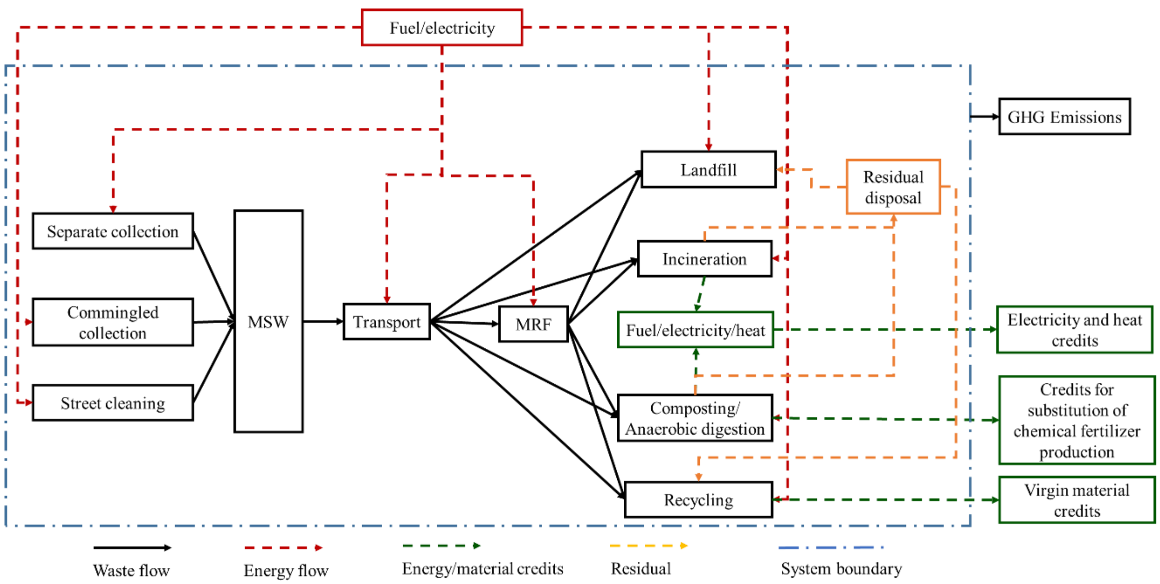


Figure 7.1. The overall scheme of MSW management system analysed in LCA (source: Wang, 2020).

7.2.3. Life cycle inventories

7.2.3.1. Collection and transfer

GHG emissions from waste collection processes were modelled based on the type of vehicle used for waste collection and the distance travelled. GHG emission to air for each type of vehicle to collect and transport one-tonne waste 1 km was obtained from Ecoinvent v3 database and are summarized in Table 7.1. To operate the transfer station, 4 kWh/t electricity and 0.84 kg/t diesel were assumed to be consumed (Turner et al., 2016).

Table 7.1. GHG emission factors applied for each vehicle type (source: Ecoinvent v3 database).

Vehicle type	kg CO_2eq./tkm
Road, lorry	0.135
Road, lorry 3.5-7.5 metric tonne	0.555
Road, lorry 16-32 metric tonne	0.177
Rail freight	0.0431
Ocean, transoceanic freight ship	0.0112

Distance travelled in kerbside collection is modelled based on the distance travelled within the Lower Layer Super Output Areas (LSOA), and the distance between LSOA and treatment facilities. LSOA is a geographical hierarchy designed to improve the reporting of small area statistics in England and Wales. Postcode is assigned to each LSOA. The Minimum population in LSOA is 1000 and the mean population is 1500. The Nottingham City Council operated an alternative weekly collection service to collect household residual waste and dry recyclables. The collection day is assigned based on the postcode. Therefore, the distance travelled within a round kerbside collection or house-to-house collection could be assumed as the distance from an LSOA to a waste treatment facility plus the distance traveled within the LSOA.

The distances travelled in collection and transportation were calculated using Google Earth. The distance travelled in the collection process within an LSOA was the length of vehicle accessible street. We assumed that the collection process is a round trip so that one street will be travelled twice by the collection vehicle. The average length of streets within an LSOA is 3.5 km (Table 7.2). Therefore, the distance traveled within an LSOA is 7 km. The distance between LSOA and treatment facilities was measured as the shortest route from the center of the LSOA to the treatment facilities (Figure 7.2). Then the average distance between LSOAs and waste treatment facilities was calculated as the transport distance between LSOA and waste treatment facilities.

Table 7.2. The average distance (km) travelled within LSOA, between LSOA and waste treatment plant (source: Wang, 2020).

	LSOA	Transfer station	Incineration plant	MRF	Composting plant
LSOA	3.5	5.7	5.7	8.5	18.5

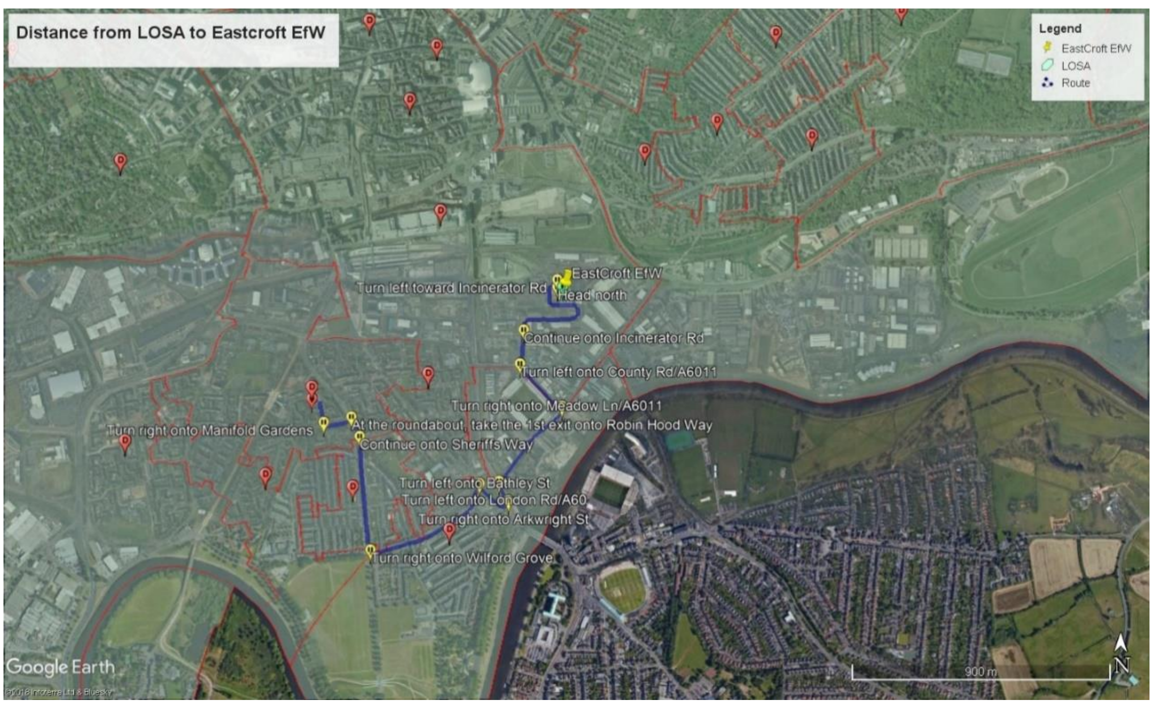


Figure 7.2. Measuring the distance between LSOA and EastCroft EfW using Google Earth (source: Wang, 2020).

The investigations conducted by the city council indicates that weekly MSW generation in Nottingham was approximately 8.5 kg/Ca in 2001/02 and 2006/07, and 6.7 kg/ Ca in 2016/17, respectively. Therefore, an average 12.8 tonnes of MSW was generated by each LSOA in 2001/02 and 2006/07, and 10.0 tonnes in 2016/17, respectively. The gross weight of refuse collection vehicles in Nottingham was between 26 and 32 tonnes (NCC, 2011). Thus, we assumed that MSW in two LSOA was collected by each trip of a collection. Based on these assumptions, the distance travelled by each trip was estimated (Table 7.3). Bulky waste collection needs order in advance, the distance travelled for collecting bulky waste is assumed as same as that in kerbside collection. There is only one CA site in Nottingham. So, the collection distance is assumed as 0 km, only the transport distance from CA site to a treatment facility is considered.

Table 7.3. The average round trip distance for each collection process (source: Wang, 2020).

Collection system	Vehicle type	Transport distance (km)
Kerbside collection	Road, lorry 16-32 metric tonnes	14
Bulky waste collection	Road, lorry 16-32 metric tonnes	14
CA site collection	Road, lorry	0
Street cleaning	Road, lorry 3.5 metric tonnes (NCC, 2017b)	20 (Turner et al., 2016)
Bring site collection	Road, lorry	20 (Turner et al., 2016)
Door-to-door collection	Road, lorry 16-32 metric tonnes	14
Other collection	Road, lorry	N/A

7.2.3.2. Transport

The distance of inter-facility transport was measured suing Google Map based on the respective locations of the start and end facilities identified from WasteDataFlow (Table

7.4.). The spatial distribution of waste treatment facilities, which receive waste from Nottingham, within the UK in 2016/17 is illustrated in Figure 7.3. The details of waste treatment facilities are shown in Appendix B. The default distance (Table 7.5) was used when the location of a facility cannot be identified.

Figure 7.3. The spatial distribution of waste management facilities and the aerial views of some facilities (source: Wang, 2020).

Table 7.4. Inter-facility transport distances (source: Wang, 2020).

Facility A	Facility B	Transport distance (km)
LSOA	Incineration plant	5.7
LSOA	MRF	8.5
LSOA	Compost plant	18.5
LSOA	Transfer station	5.6
LSOA	Sims Group UK Ltd	5.8
Transfer station	Residual MRF	6.9
CA site	Transfer station	4.8
CA site	Compost plant	25.1
CA site	MRF	11.6
CA site	Wastecycle Ltd	11.6
CA site	Sims Group UK Ltd	0.5
CA site	European Metal Recycling Ltd	0.5
CA site	Electrical Waste Recycling Group	107
CA site	Mercury Recycling Ltd	122
CA site	British Gypsum East Leake	13.2
CA site	Wastecare Limited	116
CA site	Oakwood Fuels Ltd.	34.4
CA site	T Berryman & Sons Ltd	93.7
CA site	Midland Glass Processing	25.9
CA site	Recresco Ltd	25.9
CA site	Samann Environmental Systems Ltd	61.6
CA site	R Plevin & Sons Ltd	110
MRF	Cemex, South Ferriby	110
MRF	Tarmac Limited	84
MRF	Recresco Ltd	173
MRF	SAICA Paper UK Ltd	160
MRF	Monoworld Ltd	129
MRF	J And A Young (Leicester) Limited	31
MRF	Roydon Group Plc	159
MRF	Tandom Metallurgical (Midlands) Ltd	90
MRF	European Metal Recycling Ltd	100
MRF	Novelis UK Ltd	143
MRF	Tom Martin & Co Ltd	213
MRF	U R M (UK) Limited	90.4
MRF	FCC Recycling (UK) Limited	33.8
MRF	Castle Cement Limited	70.6
MRF	Sims Group UK Ltd	11.7
MRF	Eastcroft EfW	6.9

Continue of Table 7.4

Facility A	Facility B	Transport distance (km)
Eastcroft EfW	Sims Group UK Ltd	6.3
Eastcroft EfW	Johnsons Aggregates Ltd	12.1
Eastcroft EfW	FCC Recycling (UK) Limited	71.5
Nottingham	ACE UK (WF6 2EP)	102
Nottingham	Viridor Waste Management Ltd	65
Nottingham	U R M (UK) Limited	86.2
Nottingham	H J Enthoven Limited	95.1
Nottingham	Abitibi Consolidated Ltd (ACRE)	444.2
Nottingham	Kappa SSK Ltd	79.3
Nottingham	Smurfit Townsend Hook	57.1
Nottingham	T Berryman & Sons Ltd	89.6

Table 7.5. Default A to B vehicle distances (km) and vehicle types for transport of waste (source: Wang, 2020; Turner et al., 2016).

Geographic area	Distance (km)	Vehicle types
City	6	Road, lorry
County	25	Road, lorry
Regional	50	Road, lorry
National	250	Road, lorry
International (Europe)	750	Road, lorry
	50	Rail, freight
International (Global)	300	Road, lorry
	15,000	Ocean, transoceanic freight ship

Detailed documentation of waste flow and detailed information on waste treatment facilities are only available for the year 2016/17. The destination of secondary materials recovered from MRF was unknown for the year 2006/07. There was little information on the waste flow in 2001/02. Therefore, some assumption was summarized as follows:

- Landfill site which could not be identified was assumed to be located within Nottinghamshire County.
- Eastcroft EfW was the only one incineration plant in all scenarios.

- Locations of reprocessors which could not be identified were assumed to be located within the UK.

7.2.3.3. 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) (Appendix C). This method calculates the total mass of methane potentially generated based on the mass and composition of landfilled waste as listed in Table 7.6.

Table 7.6. Composition of MSW and the landfilled waste (%) (source: Wang, 2020).

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: sourced from Burnley (2001); b: Waste composition was estimated based on results of MFA in Chapter 6. c: sourced from IPCC (2006). d: Putrescible includes garden waste and food waste. Values in brackets () are the default values set by IPCC (2006).

7.2.3.4. 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 7.7) using equations provided by the IPCC (2006) (Appendix C). 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.

Table 7.7. Composition of waste incinerated at Eastcroft EfW (source: Wang, 2020).

	2001 [a]	2006 [b]	2016 [c]	Future 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: sourced from Burnley (2001). b: sourced from WRL (2008). c: sourced from NCC (2013a). d: Waste composition was calculated based on results of MFA in Chapter 6. e: IPCC (2006). f: LHV was calculated using the regression model Eq. 5.14 built in Chapter 5.

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. 5.14, developed in Chapter 5.

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 used 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, 2016a). Based on these assumptions, the quantity of natural gas and associated GHG emission saved by EfW were quantified.

7.2.3.5. Recycling

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

7.2.3.6. Composting

GHG emissions from composting were calculated after excluded the 36% non-compostable fraction (NCC, 2013a). Details of LCI for composting are presented in Table 7.8. 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.

Table 7.8. LCI for composting (source: Wang, 2020).

	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)

7.2.3.7. MRF

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 consumptions 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 consumptions in a Residual MRF are 2 kg/t and 44 kWh/t, respectively (Pressley et al., 2015; Turner et al., 2016).

7.2.3.8. 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) (Appendix C).

7.2.3.9. Anaerobic digestion

Biogas production with a yield of 20% by weight of which 63% is methane in an anaerobic digestion process, was assumed (Zaccariello et al., 2015, Turner et al., 2016). The LHV of biogas is 30 MJ/kg (DEFRA, 2016a). 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 anaerobic digestion process is presented in Table 7.9. Avoided GHG emissions from production of 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 (Hill et al., 2011).

Table 7.9. Life cycle inventory data for the anaerobic digestion process (source: Wang, 2020).

	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 (2016a)
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)

7.2.4. 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 the amount of GHG emission and its equivalency factor. The total GWP of 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 for 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).

7.2.5. 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 tonne of MSW (expressed as GWP_{100} per tonne 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 understanding 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 using the LHV predictive model (Eq. 5.1) built by Khan and Abu-Ghararah (1991) to estimate the LHV of incinerated MSW. Sensitivity analysis 2 was carried out by varying the DOC in landfilled waste, the content of N, P, K in composted organic waste (Table 7.10), and the LHV and fossil carbon of RDF.

Table 7.10. Nutrients (N, P, K) content and potential of inorganic fertilizer replacement of different compost (source: Boldrin et al., 2009).

	Nutrients Content (kg/t wet compost)			Inorganic fertilizer replacement (kg/t treated waste)		
	N	P	K	N	P	K
Food waste	6.0-21.5	1.8-4.7	6-13.4	0.5-5.2	0.6-1.9	2.4-5.4
Garden waste	3.9-8	1-4	5-13.8	0.5-3.4	0.6-2.8	3.5-9.7

7.3. Results and Discussions

7.3.1. Historical GWP_{100} of MSW management

7.3.1.1. GWP_{100} per tonnes of MSW

The LCA results are presented in Figure 7.4 – 7.6 and Table 7.11. Figure 7.4a 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 211.3 kg CO_2-eq./t of MSW was achieved in S3, which was half of that in S2 (Figure 7.4a). 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 for the equivalent GWP_{100} from the consumption of virgin materials and fossil fuels.

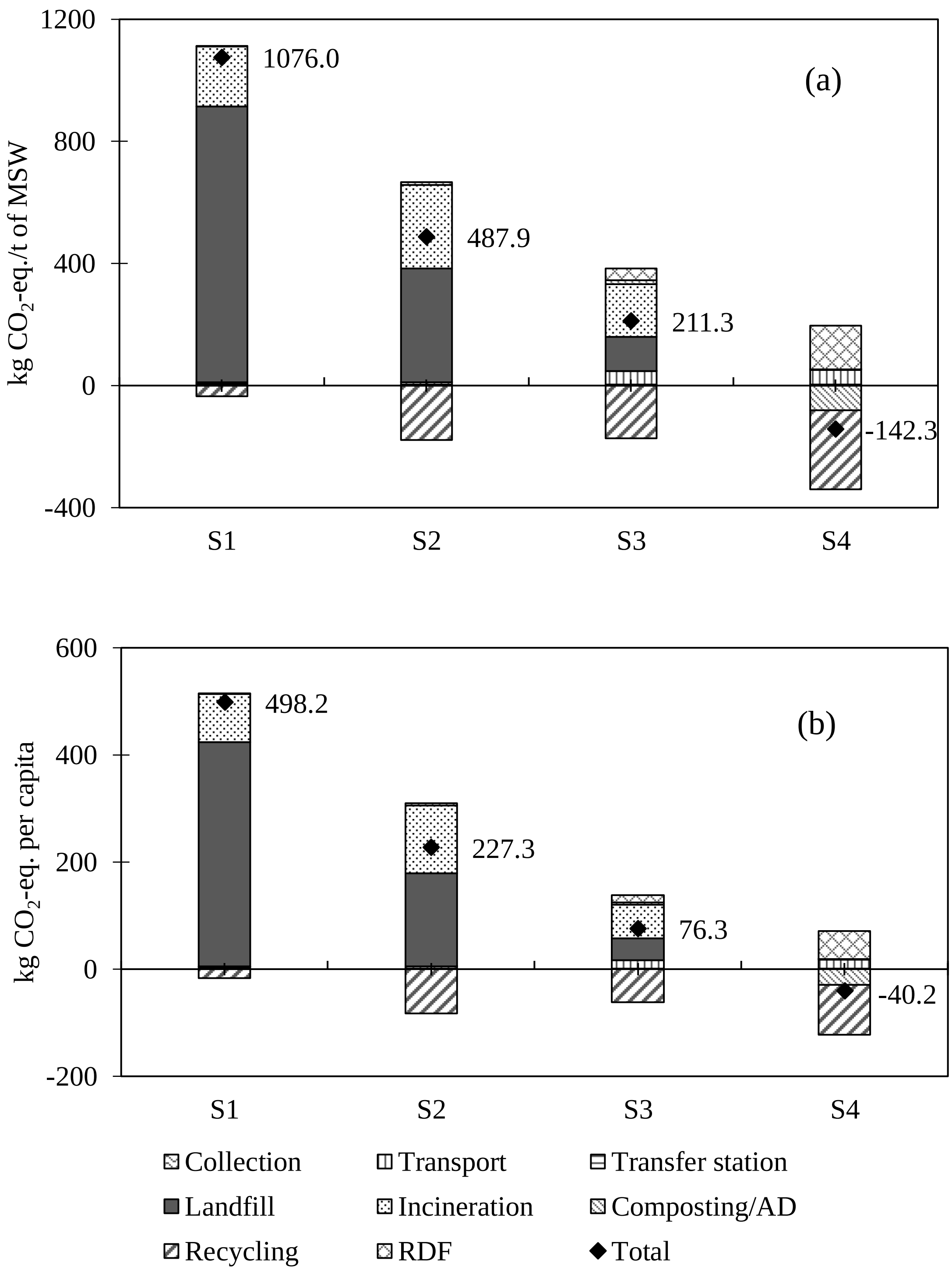


Figure 7.4. The GWP_{100} of MSW management scenarios in Nottingham (source: Wang, 2020). (a): GWP_{100} per tonne of MSW. (b): GWP_{100} per capita. AD represents anaerobic digestion.

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 Figure 7.5 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).

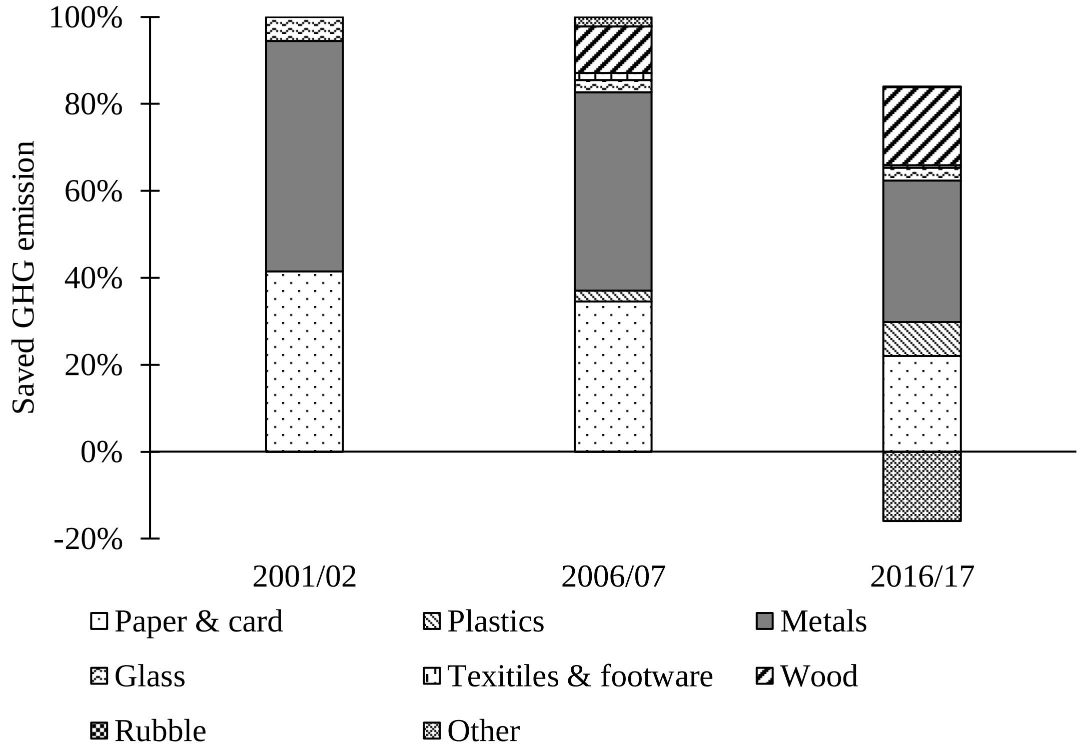


Figure 7.5. The fraction of GWP_{100} saved by recycling different materials (source: Wang, 2020).

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 the case of heat production and 52.1% in the 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 gently declining trend during the same period (Table 7. 11). The reduction in GWP_{100} from the 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 travelled 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 7. 11), due to two factors: more materials were recycled, and reprocessing facilities have usually located some distance from Nottingham. For example, recycled glass was transported 173 km for reprocessing and paper was transported 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. A 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.

Table 7.11. GWP_{100} added by collection and transport (unit: kg CO_2-eq.) (source: Wang, 2020).

		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

7.3.1.2. 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 (Figure 7.5b). This is due to the improvements in MSW management discussed in section 7.3.1.1, as well as efforts in waste prevention. MSW generation per capita decreased from 463 kg to 361 kg during the same period (Figure 6.3 in Chapter 6). 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 7.11), the reason for which has also been detailed in section 7.3.1.1.

7.3.2. 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 tonne of MSW and GWP_{100} per capita reduce to just – 142.3 kg CO_2-eq. and – 40.2 kg CO_2-eq (Figure 7.5b), respectively. Anaerobic digestion 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 7.11).

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). Recycling material from mixed residual waste could reduce the GWP of MSW management, 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. Thus, enhancing source separation and public participation will be crucial to improve the quality of secondary products.

7.3.3. Sensitivity analysis

As presented in Figure 7.7 and Table 7.12, sensitivity analysis results indicate that the variations in the LHV prediction model and waste composition affect the estimated GWP_{100} values, but not the downward trend.

To assess the sensitivity of LCA results affected by the LHV predicting model, the model developed by Khan and Abu-Ghararah (1991) (Eq. 5.1 in Chapter 5), using global data collected and the same explanatory variables as Eq. 5.14 in Chapter 5, 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 Figure 7.6 illustrated, both the LHVs and associated GWP_{100} of incinerated waste in all three scenarios change significantly when using Eq. 5.1. However, this model has 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. 5.14) is recommended to estimate the LHV of MSW. Nevertheless, the GWP100 of MSW management in Nottingham is estimated to have reduced during the study period, irrespective of the model adopted.

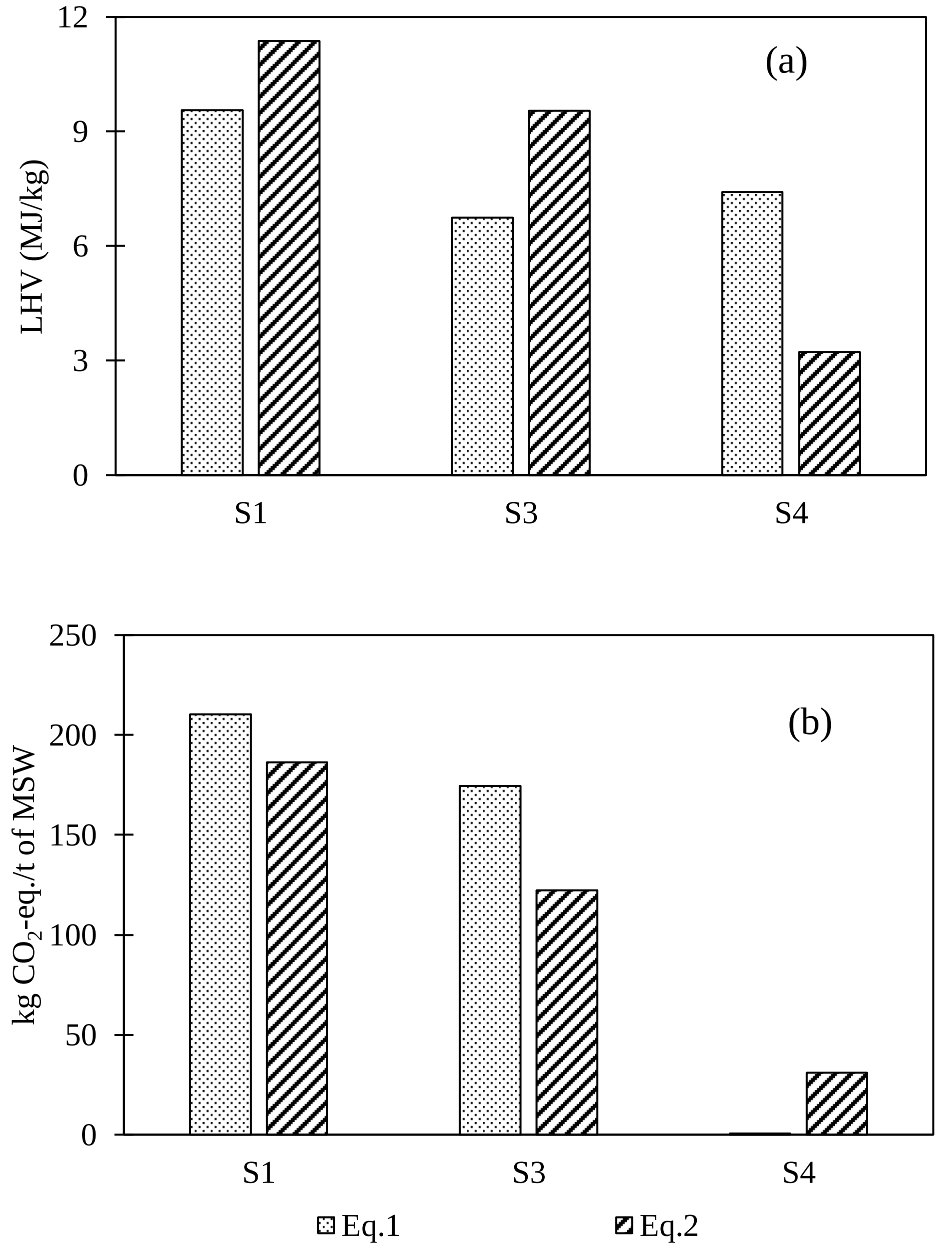


Figure 7.6. Variation of estimated LHVs (a) and GHG emissions (b) of incinerated waste when different LHV models were used to estimate its LHV (source: Wang, 2020).

The DOC (Table 7.6), N, P and K (Table 7.10) 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 7.12 illustrates the minimum and maximum GHG emission from managing 1 tonne of MSW when the variations in waste composition are taken into consideration.

Table 7.12. Effect of waste composition variation on GWP_{100} (unit: kg CO_2-eq./t MSW) (source: Wang, 2020).

	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 /anaerobic digestion	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

7.3.4. Limitations and future endeavours

The assessment of life cycle impacts of MSW management in this study focused on the global warming potential at this stage because reduction of global warming through the reduction of the landfill of biodegradable waste was emphasized in the EU Landfill Directive. However, MSW is a complex mixture and multiple toxic and non-toxic by-products are produced through all the management processes from collection to the final disposal. Only assessing the impact on global warming of MSW management system cannot present the overall performance of it. More impacts on environment and human health should be considered. Therefore, impacts in terms of acidification potential, eutrophication potential, human toxicity potentials, ecotoxicity potential, photochemical oxidation potential, ozone layer depletion potential, energy consumption, and abiotic depletion potential, etc. will be considered and assessed in my future research works.

Only one future scenario has been set and assessed because this study focused on the transition of MSW management under the guidance of EU waste directives. Missing the comparison among potential scenarios might influence the identification of the most promising scenario and/or measurement. Assessing and comparing the life cycle impacts of different scenarios could improve the comprehensiveness of this study and for better decision making. Besides, more impacts on environment and human health should be taken into consideration when assessing these scenarios.

7.4. Conclusions

To assess the effectiveness of waste regulations and the evolution of MSW management under the guidance of these regulations, in this chapter, 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 were 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 centred 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, pre-treating 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 needs to be enhanced to ensure source separation. Besides, attention should be paid to the quality of recycled and recovered materials.

Chapter 8 Discussions and Conclusions

This chapter brings together the conclusions of the findings and the implications of the whole study for both research and practice. In particular, this study provides a clear picture of the change of waste flows and the decreased GWP contribution in Nottingham to the globe due to the evolution of waste management strategies corresponding to EU and national waste management regulations. The effectiveness of waste prevention has been quantitatively assessed and demonstrated to be an important waste management strategy. Besides, robust LHV predictive models have been built based on the wet-physical composition of MSW using datasets collected worldwide. Those models have the advantages to be internationally applied considering the geographical coverage of the data based on which the models were derived. As collecting data on the wet-physical composition of MSW is relatively more convenient and less costly than chemical analysis, the models also provide a convenient way for LHV estimation from the perspective of data accessibility. As Table 8.1 illustrated, all research questions and objectives listed in sections 1.1 – 1.3 were explicitly answered. The key findings, contributions and limitations of this study, the lessons can be learned from this study, as well as the future endeavours, are discussed in the flowing sections.

Table 8.1. Summary of all explicitly answered research questions and objectives (source: Wang, 2020).

Research questions objectives	Main findings	Related sections
EU, national and local waste regulations	National and local waste regulations were developed to respond EU waste directives. Recycling practices were emphasized and recycling targets were improved step by step.	Chapter 4
MSW management situations in Nottingham	The structure of MSW management system became complex. Landfill became the least favorable management option.	Section 3.4 and Section 4.3.
Waste streams at different stage of evolution	Pathways of waste flows became complex. Waste streams were diverted to the upper layers of the waste management hierarchy.	Sections 6.3.1 - 6.3.5.
Performance on waste flows	It has been improved from the aspects of prevention, recycling and energy recovery against indicators selected based on waste management hierarchy.	Sections 6.3.1 - 6.3.5.
Performance on life cycle impacts	Global warming potential of MSW management has been significantly reduced.	Section 7.3.1
Evaluation of waste prevention	Waste generation per capita and global warming potential per capita reduced notably	Section 6.2.1, Section 6.3.1, Section 7.2.1 and Section 7.3.2.
LHV predictive model	Internationally applied LHV predictive models were built. Separating food waste improves the LHV of incinerated waste. Selection of model building method should base on the quality and size of data and the skills of researchers.	Chapter 5
Successes and failures	Prevention campaigns successfully reduced the waste generation. Public education on waste separation at source need be catch up. Recycling rate of textile is low. Efficiency of waste incineration was relatively low.	Section 6.3 and Section 7.3.
Future improvements	Enhancing public education. Separating food waste textile at source. Updating treatment facilities.	Section 6.3 and Section 7.3.
Reference to other cities	Setting MSW management targets. Separating waste at source. Applying multiple management and treatment strategies.	Section 8.3.

8.1. Findings

A systemic review of waste regulations, policies and activities in EU, England and Nottingham, as well as the associated achievements, are presented in Chapter 4. The enforcement of EU waste directives has stimulated the update of national and local waste management regulations, plans and strategies. MSW management in European countries had changed significantly since the enforcement of the EU Landfill Directive, especially after the enforcement of the Waste Framework Directive which circulated the waste management hierarchy as the guiding principle to practice MSW management. Since the announcement of the Waste Framework Directive, achievements have been gradually made from the aspects of prevention, materials recycling, material and energy recovering, and reduction of landfilled waste as the waste management hierarchy indicated among EU member states. In terms of the waste policies and measurements of MSW in Nottingham, three main waste management strategies had been published by the local authority to guide the practice of waste prevention, separation, recycling, treatment and disposal. A series of actions and measurements had been undertaken, and facilities had been installed, to facilitate the improvement of MSW management. As a result, MSW management in Nottingham changed from a simple model of combined landfilling and incineration with energy recovery to a combination of source separation, recycling, pre-treatment before landfilling, production of RDF, composting, incineration with energy recovery and landfilling with complex waste flows.

To accurately estimate the energy content of MSW so as to quantify the saved GHG emission by energy recovery, LHV predictive models have been built in Chapter 5. These models provide a possibility to estimate the energy recovery potential from MSW at a globe scale. However, building an LHV predictive model that can be conveniently applied worldwide without high cost is beneficial but it is difficult to collect sufficient and high-quality data to build this type of model because of the lack of related research and inaccessible data source. Besides, the inconsistency in data due to the variation in waste classification and the original purpose of data collection also causes uncertainties of the models built based on these datasets. The two model building methods, multiple regression and ANN, have their strengths and weakness to build LHV predictive models. It is important to think through when choosing a method to build an LHV predictive model. Besides, the experience in building the LHV predictive models indicates that separating food waste from incinerated waste contributes to the improvement of combustion efficiency.

The performance of the MSW management in Nottingham was assessed using MFA and LCA. Both the MFA and LCA results presented in chapters 6 and 7 confirmed the improved performance of the MSW management in Nottingham. MSW generation per capita has been gradually reduced during the study period (from April 2001 to March 2017). Material recycling and recovery have been improved through the measurements of source separation and treatment in MRF. Energy recovery also has been improved through the production of RDF. All these improvements contribute to the transformation from linear urban metabolism to a circular one in Nottingham and the development of circular economy in the UK by providing renewable material and energy resources from waste. However, more efforts need to be made to meet national and local waste management targets. The future improvements, in this case, can be made by separating food waste and textile waste at sources and replacing open windrow composting by anaerobic digestion.

To further reduce the GWP of MSW management in Nottingham, the EfW technologies need to be updated to improve their energy recovery efficiency, and domestic reprocessing of secondary materials need to be enhanced to reduce the GHG emissions from transportation.

In addition to improving the recycling rate and reducing GWP of MSW management, attention should also be paid to the quality of secondary products derived from waste. In Nottingham, a certain amount of materials (about 2,400 tonnes in 2016/17) were recycled from residual waste and more materials should be recycled from residual waste to improve the recycling rate so as to meet the recycling targets. The improved recycling rate does increase the quantity of secondary material brought back to the metabolism process thus contributes to the circular economy, as well as reduces the GWP of MSW management. However, hazardous or undesirable substances can be introduced to recycled material after contaminated by residual waste. This would reduce the quality of the products made of recycled materials, even introduce risk to human health. Besides, the declined recycling rate at MRF with the expansion of kerbside collection indicates that separation at source needs to be enhanced. In this case, improving the quantity and quality of material recycling relays on public participation, and improving public awareness will be the key to improvement. Thus, public education, supportive facilities and information for source separation is the fundamental step for establishing sustainable MSW management.

8.2. Contributions

This study offers a comprehensive analysis of the transformation of MSW management in a selected British city under the guidance of EU and national waste regulation. The evolution process of MSW management in Nottingham reveals the way an intergovernmental organization drives the changes in environmental practices. As the related topic has so far not been explicitly discussed before. This study has contributed to the field of MSW management, environmental management and urban metabolism from theoretical, practical and political aspects.

8.3.1. Theoretical contributions

This study expands previous work on MSW management theory in the following ways.

Firstly, the present research builds the LHV predictive models which can be used internationally. The models are built based on the physical composition of MSW at wet-basis, which can be easily applied in estimating LHV of MSW because collecting data of the physical composition of MSW is easy and less costly. The comparison between the two model building methods (multiple regression analysis and ANN) indicates that their compatible performance in building LHV predictive models. Although ANN could capture the non-linear relationships between LHV and explanatory variables, ANN model works as a black box. It is difficult to verify or validate the model based on knowledge of the chemical mechanism involved in incineration. Besides, it is difficult to identify and ensure the size of training data.

Secondly, this study presented a method to quantitatively assess the effectiveness of waste prevention that is quantifying the per capita waste generation and the associated environmental burdens. The LCA results demonstrated that waste prevention is superior to other management options regarding GWP reduction and should be integrated into any MSW management system.

Thirdly, the enforcement of waste policies and appropriately setting management goals incentive the transformation of MSW management strategies, and thus improve the performance of MSW management.

8.3.2. Managerial contributions

An important contribution of this study is demonstrating that separating food waste at source and treating it using biological treatments is beneficial for improving the performance of MSW management. Thus, the circular style of urban metabolism may be better realized.

It is also important to note that, successfully implementing the separate collection of waste relies on public participation. Therefore, public education needs to catch up with the evolution process of MSW management. Besides, accessible facilities for source separation and knowledge on waste prevention should be provided to the public.

Another important contribution of this study is that more attention should be paid to improving energy recovery efficiency in EfW facilities, enhancing domestic secondary material processing and assuring the quality of recycled materials. With the improvement of material recycling, the environmental burdens caused by the transportation of recycled materials become visible. So, after improving the waste treatment processes, the logistics of waste collection become the nest issues and research focus in the area of waste management.

8.3.3. Policy implication

MSW is a complex mixture. Even materials within the same category have significantly different physical and chemical properties and their management strategy should be different. Taking plastic products and paper products as examples. The proper way to treat plastic film and toilet paper is incineration, while recycling is a better choice for plastic bottles and books. Therefore, policies should be undertaken to refine the classification of these waste and to educate the public to separate these wastes.

Public participation is the foundation and key to build a sustainable MSW management system because citizens are the executors for waste prevention and source separation. However, the public education and/or stimulation were less emphasized in waste management policies. Therefore, we recommend authorities to take public participation into consideration in policy making, and to establish reward and punishment policies to stimulate the public participation in MSW management.

In previous waste management strategies, target setting was always quantity-oriented and the quality by-products derived from waste were less mentioned. For example, recycling and composting 50% of household waste, but the quality of recycled materials and compost derived from waste were seldom mentioned. Besides, the "recycling rate" is not clearly defined as the materials collected for recycling, or the materials processed for recycling, or the materials actually produced to new products. Therefore, concepts should be clearly clarified in waste management policies and we wish quality-oriented policy could be made in the future.

8.3. Learned Lessons

Through investigating and assessing the transition of MSW management in Nottingham, some experiences and lessons can be learned and transposed to other cities to facilitate their decision making on MSW management.

Well developed waste policy and targets, as well as available facilities, are the foundations supporting the transition of MSW management in Nottingham. Waste prevention and recycling were emphasized by the local authority and associated targets were set in local waste management strategies. This stimulated and impelled the establishment of diverse but coordinated waste management facilities to support the prevention and recycling of waste. For example, the “Freecycle” and “Exchange” networks, KCS and MRF. Establishing community exchange networks of second-hand products might be a good start for Chinese cities to prevent waste.

With the initiation and strengthen of KCS, the recycling rate in Nottingham improved year by year. Source separation collection, which just initiated in several Chinese cities, is the key to improve material recycling. Establishing an accessible and easy-to-operate source separation scheme is essentially important for Chinese cities to establish sustainable MSW management. However, differences on the characteristics of MSW and the structure of settlements between western and Chinese cities requires local authorities to modify the waste classification system and way to collect waste before applying KCS in Chinese circumstance.

Public participation is the key to achieve a sustainable MSW management. Public education should be the first step of the establishment of sustainable MSW management. Besides, building accessible and suitable supportive facilities, and providing available information and knowledge on waste separation encourage public’s participation in the separate collection.

Furthermore, technologies integrated in an MSW management system determine the impacts and efficiency of the whole system. They not only affect the production of pollutants from the treatment processes, but also determine the quality of the by-products derived from waste. The combination of these technologies might turn the system from an energy consumer to a provider, from a GHG emitter to a saver. Therefore, suitable technologies should be taken into the consideration of design and operation of an MSW management system.

8.4. Limitations and Uncertainties of Study

While the present study has provided theoretical and empirical contributions, it has some limitations and uncertainties. One of these is that the performance of LHV models is relatively lower than locally build models because of the uncertainties in training data. Besides, most of the training data were collected from developing countries. This might constrain their application in developed countries.

Because of data deficiency, the performance of MSW management before the enforcement of the EU Landfill Directive cannot be comprehensively assessed in equal detail to that after the enforcement. This makes the results of comparison less accurate. Besides, the analysis of material flows in each single management process and the material flows at substance level have not been performed. This might affect the performance of assessment on the qualitative aspects of MSW management and identification of some specific impacts on environment or human health. However, it is clear that the MSW management was improved during the last two decades.

Only one future scenario has been assessed and only GWP has been quantified. This is not enough for comprehensive assessment of an MSW management system because the sustainability of an MSW management system should be assessed at environmental, social and economic dimensions. Comparison among different scenarios is need for the identification of the most promising scenario, and multiple impacts of MSW management should be assessed. Besides, economic investment is the key to establish and operate an MSW management system. Life cycle cost of the MSW management system has not been assessed at this stage.

8.5. Future Endeavours

Based on the limitations and uncertainties of this study, several future research opportunities are identified and efforts can be made from the following aspects. Firstly, the LHV predictive model was based on second-hand data; it would be interesting to cooperate with other researchers to collect more first-hand data with a more consistent approach so as to improve the accuracy of the LHV predictive model, and to build a model which could be applied at the global scale. Secondly, the suitability of ANN in building LHV predictive model needs to be further assessed and discussed by varying the size of input training data. Thirdly, Extending the MFA to substance level and to single management practice. Lastly, the assessment of the performance (or sustainability) of the MSW management should be extended to a broader perspective that covers more environmental impacts and impacts on social and economic dimensions, and more scenarios.

In addition to the extension of this study, future research related to MSW management can be addressed in the following aspects:

- Comprehensive study on the physical and chemical characteristics of waste streams, as well as the source of these waste streams, is needed because the design of integrated MSW management is based on the materials consists of MSW.

- Since waste management is the key to achieve circular metabolism or circular economy, the connection between waste management sector and other sectors which produce waste or utilize resource from waste need to be fully explored so as to improve the efficiency of metabolism process.
- Under the context of globalization, links among cities are getting closer and material and energy exchange between cities are becoming more frequent. In the future study, the system boundary of MSW might need to expand.
- Waste classification at source and separate collection have been implemented in some Chinese cities. Waste management at the downstream after collection needs to catch up to harness the resources from waste and to safely dispose of separated waste. Besides, the interaction among stakeholder also need to be addressed to secure the separate collection. Furthermore, the influence and consequence of this new collection strategy need to be monitored so that the drawbacks could be detected and corrections can be made in time.

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Appendix A Abbreviations of European Countries

AT	Austria
BE	Belgium
BG	Bulgaria
CZ	Czech Republic
DK	Denmark
DE	Germany
EE	Estonia
IE	Ireland
GR	Greece
ES	Spain
FR	France
HR	Croatia
IT	Italy
CY	Cyprus
LV	Latvia
LT	Lithuania
LU	Luxembourg
HU	Hungary
MT	Malta
NL	Netherlands
PL	Poland
PT	Portugal
RO	Romania
SI	Slovenia
SK	Slovakia
FI	Finland
SE	Sweden
UK	United Kingdom
IS	Iceland
NO	Norway
TR	Turkey
CH	Switzerland

Appendix B Database of Waste Treatment Facilities

Table B.1. Facilities that managed MSW generated in Nottingham in 2006/07.

Facility type	Facility name	Facility location (country)	Facility postcode	Material
Reprocessor	Abitibi Consolidated Ltd (ACRE)	UK	EH21 6SY	Paper
Reprocessor	Kappa SSK Ltd	UK	B7 5RE	Mixed paper & card
Reprocessor	Smurfit Townsend Hook	UK	B78 1TS	Mixed paper & card
Reprocessor	R Plevin & Sons Ltd	UK	OL5 9NG	Wood
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Wood
Reprocessor	Samann Environmental Systems Ltd	UK	S26 6NQ	Rubble
Reprocessor	R Plevin & Sons Ltd	UK	OL5 9NG	Rubble
Reprocessor	Recresco Ltd	UK	NG17 8AP	Mixed glass
Reprocessor	Midland Glass Processing	UK	NG17 8AP	Mixed glass
Reprocessor	T Berryman & Sons Ltd	UK	WF9 3NR	Mixed glass
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Mixed paper & card
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Mixed cans
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Plastics
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Wood
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Rubble
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Other Scrap metal
Reprocessor	Mountstar Metal Corporation Limited	UK	NG7 2SD	Other Scrap metal
Reprocessor	Sims Group U K Limited	UK	NG7 2SD	Fridges & Freezers
Reprocessor	Sims Group U K Limited	UK	NG7 2SD	Other Scrap metal
Reprocessor	Sims Group U K Limited	UK	NG7 2SD	Automotive batteries
Reprocessor	Sims Group U K Limited	UK	NG7 2SD	Other electrical goods
Reprocessor	Waste Notts Ltd	UK	NG7 8PU	Rubble
Transfer station	Nottingham City Council	UK	NG2 3AH	MSW
Landfill	Biffa Waste Services Ltd	UK	NG9 3GJ	MSW
Landfill	Waste Notts Ltd	UK	NG22 8UB	MSW
Landfill	Waste Notts Ltd	UK	NG7 8PU	MSW
Landfill	Viridor Waste Exeter Limited t/a Vindor Waste Management Lim	UK	S44 5HS	MSW
Landfill	Albion Waste Management Ltd	UK	S9 3HY	MSW

Continue of Table B.1

Facility type	Facility name	Facility location (country)	Facility postcode	Material
Landfill	Yorkshire Water Services Limited	UK	LS9 0PJ	Fly ash
Incineration with energy recovery	Waste Recycling Group Ltd	UK	NG2 3JH	MSW
Residual waste MRF	Wastecycle Ltd	UK	NG4 2JT	Residual waste
MRF	Wastecycle Ltd	UK	NG4 2JT	Mixed recyclables
Windrow or other composting	Leverton A & P	UK	NG6 0BT	Green waste only

Table B.2. Facilities that managed MSW generated in Nottingham in 2016/17.

Facility type	Facility name	Facility location (country)	Facility postcode	Material
Charity	Oxfam	N/A	N/A	Textiles & footwear
Charity	Planet Aid	N/A	N/A	Textiles & footwear
Exporter	Wastecycle Ltd	UK	NG4 2JT	RDF
Exporter	Wastecycle Ltd	UK	NG4 2JT	Mixed Plastics
Exporter	Wastecycle Ltd	UK	NG4 2JT	Mixed paper & card
Exporter	Wastecycle Ltd	UK	NG4 2JT	Mixed cans
Hazardous landfill	FCC Recycling (UK) Limited	UK	S9 1HL	Fly ash
Incineration with energy recovery	FCC Environment formerly WasteNotts (Reclamation) Ltd	UK	NG2 3JH	Incineration with energy recovery
Incineration with energy recovery	Cemex, South Ferriby	UK	DN18 6JL	RDF
Incineration with energy recovery	Tarmac Limited	UK	SK17 8TG	RDF
Incineration with energy recovery	Castle Cement Limited	UK	PE9 3SX	RDF
Non-hazardous landfill	FCC Recycling (UK) Limited	UK	NG24 3JJ	
Reprocessor	Johnsons Aggregates Ltd	UK	NG11 6QN	Incinerator Bottom Ash
Reprocessor	U R M (U K) Limited	UK	WF9 3NR	Mixed glass
Reprocessor	Recresco Ltd	UK	CH65 1AB	Mixed glass
Reprocessor	SAICA Paper UK Ltd	UK	M31 4QN	Mixed paper & card
Reprocessor	RRG Rheinische Recycling GmbH	Germany	N/A	Mixed cans

Continue of Table B.2

Facility type	Facility name	Facility location (country)	Facility postcode	Material
Reprocessor	Monoworld Ltd	UK	MK44 1NB	Mixed Plastics
Reprocessor	Roydon Group Plc	UK	M27 8LU	Mixed Plastics
Reprocessor	J And A Young (Leicester) Limited	UK	LE11 5RH	Mixed Plastics
Reprocessor	H J Enthoven Limited	UK	WS10 8JR	Post consumer, non automotive batteries
Reprocessor	ACE UK	UK	WF6 2EP	Composite food and beverage cartons
Reprocessor	British Gypsum East Leake	UK	LE12 6HX	Plasterboard
Reprocessor	Other/Exempt	UK	N/A	Textiles & footwear
Reprocessor	Other/Exempt	UK	ME3 9ND	Mixed Plastics
Reprocessor	Sims Group U K Ltd	UK	NG7 2SD	Other Scrap metal
Reprocessor	Sims Group U K Ltd	UK	NG7 2SD	WEEE - Large Domestic App
Reprocessor	Sims Group U K Ltd	UK	NG7 2SD	WEEE - Fridges & Freezers
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Mixed glass
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Wood
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Rubble
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Soil
Reprocessor	Wastecare Limited	UK	LS25 1NB	Post consumer, non automotive batteries
Reprocessor	Electrical Waste Recycling Group	UK	HD5 0JS	WEEE - TVs & Monitors
Reprocessor	Oakwood Fuels Ltd.	UK	NG22 8UA	Mineral Oil
Reprocessor	Mercury Recycling Ltd	UK	M17 1HW	WEEE - Flourescent tubes and other light bulbs
Reprocessor	European Metal Recycling Ltd	UK	NG7 2SF	Other Scrap metal
Reprocessor	European Metal Recycling Ltd	UK	NG7 2SF	Automotive batteries
Reprocessor	Viridor Waste Management Ltd	UK	S4 7WT	Mixed glass
Reprocessor	NWR Niederrheinische-Wertstoff-Ruckgewinnungs MbH	Germany	N/A	Mixed Plastics
Reprocessor	Outside UK-nonEU	China	N/A	Mixed Plastics
Reprocessor	Sims Group U K Ltd	UK	NG7 2SD	Metals from Incinerator Bottom Ash
Reprocessor	OUD Papier Centrale	Belgium	N/A	Mixed paper & card

Continue of Table B.2

Facility type	Facility name	Facility location (country)	Facility postcode	Material
Reprocessor	Solidus Solutions BV	Netherlands	N/A	Mixed paper & card
Reprocessor	N/A	China	N/A	Mixed paper & card
Reprocessor	N/A	UK	N/A	Mixed cans
Reprocessor	European Metal Recycling Ltd	UK	WS10 8LW	Mixed cans
Reprocessor	Novelis UK Ltd	UK	WA4 1NN	Mixed cans
Reprocessor	Tandom Metallurgical (Midlands) Ltd	UK	B18 7AA	Mixed cans
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Wood
Reprocessor	Wastecycle Ltd	UK	NG4 2JT	Rubble
Reprocessor	Sims Group U K Ltd	UK	NG7 2SD	Other Scrap metal
Reprocessor	Smurfit Kappa Ltd	Netherlands	N/A	Mixed paper & card
Reprocessor	Eska Graphic board	Netherlands	N/A	Mixed paper & card
Reprocessor	Tom Martin & Co Ltd	UK	PR5 8AE	Mixed cans
MRF	Wastecycle Ltd	UK	NG4 2JT	Mixed recyclables
Residual waste MRF	Wastecycle Ltd	UK	NG4 2JT	Residual waste
Windrow or composting	Simpro Ltd	UK	NG25 0RG	Green garden waste only
Windrow or composting	RM Wright Straglethorpe	Switzerland	N/A	Green garden waste only

Appendix C Models for Calculating GHG Emissions

Methane emission from landfill

The amount of methane emitted from landfill can be estimated based on Eq. C1 reported by Fong et al. (2015):

$$ME_D = (MSW_x \times MCF \times DOC \times DOC_F \times F \times 16/12 - R) \times (1 - OX) \qquad \text{(C1)}$$

Where:

ME_D:	Methane emission.
MSW :	Mass of solid waste sent to landfill in inventory year.
MCF :	Methane correction factor. Default value 1 for managed landfill is used.
DOC :	Fraction of degradable organic carbon.
DOC_F :	Fraction of DOC dissimilated. Default value 1 is used in this paper.
F :	Fraction of CH_4 in landfill gas.
$16/12$:	Conversion of C to CH_4.
R :	Recovered CH_4.
OX :	Oxidation factor (IPCC default is 0).

IPCC guidelines provide the following Eq. C2 to estimate the content of degradable organic carbon in MSW and those default values for those parameters involved from IPCC guidelines are adopted for the estimation of DOC in MSW (Fong et al., 2015).

$$DOC = (0.15 \times A) + (0.2 \times B) + (0.4 \times C) + (0.43 \times D) + (0.24 \times E) + (0.24 \times W) \quad \text{(C2)}$$

Where:

DOC:	Degradable Organic Carbon.
A:	Fraction of solid waste that is putrescible including food waste, garden waste and plant debris.

B: Fraction of solid waste that is garden waste and plant debris

C: Fraction of solid waste that is paper.

D: Fraction of solid waste that is wood.

E: Fraction of solid waste that is textile.

W: Fraction of solid waste that is other waste.

CO_2 emission from EfW

Given that CO_2 capture system is not in place in most waste incineration plants worldwide, the quantity of CO_2 emitted from incinerator could be calculated based on the mass and composition of incinerated waste using Eq. C3 (IPCC, 2006).

$$CO_2\ Emissions = m \times \sum_i (WF_i \times dm_i \times CF_i \times FCF_i \times OF_i) \times (44/12) \qquad \text{(C3)}$$

Where:

CO_2 emissions:	Total CO_2 emission from incinerated solid waste.
m :	Mass of incinerated waste. It is 1 tonne in this paper.
WF_i:	Fraction of waste consisting of type i matter.
dm_i :	Dry matter content in the type i matter.
CF_i :	Fraction of carbon in the dry matter of type i matter.
FCF_i :	Fraction of fossil carbon in the total carbon component of type i matter.
OF_i :	Oxidation fraction or factor. The default value 1 for incineration is used.
i:	Matter type of the solid waste incinerated such as paper, plastics, textile, etc.
$44/12$	Conversion of C to CO_2.

Appendix D England Carbon Metric Report 2012

Table D.1. GHG emission savings via recycling versus landfill (CO_2-eq.kg/tonne)

Material Group	Material Type	Recycling	Reuse
Glass	Green glass	392	0
Glass	Brown glass	392	0
Glass	Clear glass	392	0
Glass	Mixed glass	233	0
Paper &Card	Paper	811	0
Paper &Card	Card	894	0
Paper &Card	Books	811	0
Paper &Card	Mixed paper & card	873	0
Paper &Card	Yellow Pages	0	0
Metal	Steel cans	1,799	0
Metal	Aluminium cans	9,267	0
Metal	Mixed cans	3,965	0
Metal	Other Scrap metal	2,239	0
Metal	Aluminium foil	9,267	0
Metal	Aerosols	0	0
Metal	Fire extinguishers	0	0
Metal	Gas bottles	0	0
Metal	Bicycles	0	0
Metal	Metals from Incinerator Ash	3,550	
Plastic	Plastics	1,215	0
Plastic	Mixed Plastic Bottles	1,156	0
Plastic	PET	1,705	0
Plastic	HDPE	1,161	0
Plastic	PVC	888	0
Plastic	LDPE	1,098	0
Plastic	PP	948	0
Plastic	PS	1,240	0
Plastic	Other plastics	688	0
Wood	Wood	1,276	1,425
Wood	Chipboard and mdf	1,276	1,425
Wood	Composite wood materials	1,276	1,425
WEEE	WEEE - Large Domestic App	1,266	0
WEEE	WEEE - Small Domestic App	1,482	0
WEEE	WEEE - Cathode Ray Tubes	0	0

Continue of Table D.1

Material Group	Material Type	Recycling	Reuse
WEEE	WEEE - Flourescent tubes and other light bulbs	0	0
WEEE	WEEE - Fridges & Freezers	656	0
Batteries	Automotive batteries	563	0
Batteries	Post consumer, non automotive batteries	563	0
Tyres	Car tyres	1,910	2,900
Tyres	Van tyres	1,910	2,900
Tyres	Large vehicle tyres	1,910	2,900
Tyres	Mixed tyres	1,910	2,900
Furniture	Furniture	921	921
Rubble	Rubble	9	10
Soil	Soil	0	0
Plasterboard	Plasterboard	139	0
Oil	Vegetable Oil	0	0
Oil	Mineral Oil	725	0
Other	Bric-a-brac	0	0
Composite	Composite food and beverage cartons	-1,730	0
Composite	Ink & toner cartridges	0	0
Composite	Mattresses	0	0
Composite	Video tapes, DVDs and CDs	0	0
Paint	Paint	2,840	0
Textiles	Textiles & footwear	2,028	5,631
Textiles	Textiles only	5,987	5,987
Textiles	Footwear only	4,385	0
Textiles	Carpets	0	0
IBA	Incinerator Bottom Ash	3	
Other	Other materials	0	0
Other	Bric-a-brac	0	0
Other	Absorbent Hygiene Products (AHP)	0	0