A Study of Landfill Content and its Chemical Evolution Using Historical, Geographic and Site Monitoring Data

by

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Abstract

Historical landfills present a paradoxical position where, on the one hand, their impact on the environment is overtly negative and, on the other are seen as potential stores of value. Intrusive investigation to identify a landfill's resource potential is expensive and, due to the heterogeneity of content, has significant uncertainty. This thesis proposes a solution using historical data to identify the quantities and types of materials entering landfills. However, it is accepted that recording of waste data was limited and exactly how much or what data exists is typically answered anecdotally. This study considers the problem from a formal research perspective by collecting and reviewing historical waste data. Data collection focussed on 4 areas: i) contemporary landfill emissions monitoring and waste import data, ii) historical data leading to waste generation, iii) historical disposal data and iv) historical municipal waste composition data. In each case, a significant quantity of data was discovered and reviewed.

Initially, using historical data, 4 discrete epochs were identified where contemporary determinants impacted on i) the materials forming waste ii) the quantities of waste generated iii) waste disposal and iv) regulatory controls. For the first two epochs, waste flows into landfills were determined by waste composition. The third epoch witnessed a transition from composition to regulation with regulation becoming the controlling factor in the fourth. Such classification enabled the data for use i) as surrogates for missing data and ii) as inputs to a system dynamics model that estimates weights for generated and landfilled MSW for England (1.051 billion tonnes and 885 million tonnes respectively) and provides quantities of MSW components disposed to landfill.

Contemporary data included leachate monitoring data for 2 landfills operational since 2005 and 2007 and regulated by the 1999 Landfill Directive whereby the European Union proposed diversion of putrescible waste away from landfill. When decomposing, putrescible MSW reduces the pH of a landfill's environment. Landfill decomposition progresses through a series of phases where this low pH phase or acetogenesis has been shown to inhibit the development of methanogenic bacteria. Analysis of the data identified this low pH phase did not occur at these landfills whereby they provided a possibly singular opportunity to report this anticipated phenomenon to the research community.

Acknowledgements

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Glossary

UK	United Kingdom
MSW	Municipal Solid Waste
WRAP	Waste and Resources Action Programme
WB	World Bank
EU	European Union
EC	European Commission
SRM	Secondary Raw Materials
WtE	Waste-To-Energy
WtM	Waste-To-Materials
LFM	Landfill Mining
LFG	Landfill Gas
CD&E	Construction, Demolition and Excavation
HW	Household Waste
OW	Other Wastes
OECD	Organisation for Economic Co-operation and Development
DofE	Department for the Environment
DEFRA	Department for Environment, Food and Rural Affairs
UKNA	United Kingdom National Archives
RPI	Retail Prices Index
CB	County Borough
BC	Borough Council
UDC	Urban District Council
RDC	Rural District Council
LA	Local Authority
DC	District Council

CC	County Council
GLC	Greater London Council
HM	Her Majesty’s
EEC	European Economic Community
WEEE	Waste Electrical and Electronic Equipment
ELFM	Enhanced Landfill Mining
EPA	Environmental Protection Agency
LCA	Life Cycle Assessment
RMP	Raw Material Potential
IEPA	Irish Environmental Protection Agency
BW	Biodegradable Waste
BMW	Biodegradable Municipal Waste
GDP	Gross Domestic Product
CIPFA	Chartered Institute of Public Finance and Accountancy
WCA	Waste Collection Authority
WDA	Waste Disposal Authority
DETR	The Department of the Environment, Transport and Regions
ONS	Office for National Statistics
EWC	European Waste Catalogue
WSL	Warren Spring Laboratories
WEEE	Waste Electrical and Electronic Equipment
CA	Civic Amenity
MIT	Massachusetts Institute of Technology
MSW_{coll}	Local Authority collected MSW
MSW_{amnd}	Local Authority amended MSW
MSW_{all}	Local Authority all MSW

P	Population (number of persons)
G	Daily MSW Generation Rate (kg/person/day)
D_i	Annual Quantity of MSW Disposed to a Strategy (tonnes/year)
D_{LF}	Annual Quantity of MSW Landfilled (tonnes/year)
D_{inc}	Annual Quantity of MSW Incinerated (tonnes/year)
D_{rec}	Annual Quantity of MSW Recycled (tonnes/year)
D_{oth}	Annual Quantity of MSW to Other Disposal (tonnes/year)
FR_i	% Rate of MSW Flowing to a Disposal Strategy (tonnes/year)
FR_{LF}	% Rate of MSW Flowing to Landfill (tonnes/year)
MF_i	Individual MSW Fraction (tonnes/year)
MF_{Met}	Metal Fraction (tonnes/year)
MFR_i	% Rate of a Component Within the MSW Stream
MFR_{Met}	% Rate of Metal Within the MSW Stream
RQ_i	The Quantity Recycled of Material i (tonnes/year)
RQ_{Met}	The Quantity of Metal Recycled (tonnes/year)
BOD_5	5-Day Biochemical Oxygen Demand (mg/l)
COD	Chemical Oxygen Demand (mg/l)
GCS	Gas Collection System

Chapter 1: Introduction

1.1 Research aims and objectives

The United Kingdom [UK] Environment Agency has on record some 19,763 historic and 1,758 currently authorised landfill sites for England and Wales (UK Environment Agency, 2019). Within Europe, this extends to an estimated 500,000 landfills with circa 450,000 closed before the impact of the Landfill Directive where 80% contain urban or municipal solid wastes [MSW] (EURELCO, 2019).

Waste generation within the UK, peaked during the period 2003 to 2007 (Waste and Resources Action Programme [WRAP], 2012). The 2012 World Bank [WB] review of solid waste management reported solid waste production in cities to total 1.3 billion tonnes annually rising to 2.2 billion tonnes by 2025 (Hoornweg and Bhada-Tata, 2012). A further WB review published in 2018 proposed annual global waste production would increase to 3.4 billion tonnes by 2050 and identified landfilling to be the primary method of waste disposal with landfilling often uncontrolled and where controls exist, for the larger part, are directed at the local level (Kaza et al., 2018). This reflects the situation that occurred within developed economies and more pertinently the United Kingdom [UK] during the periods before and after the Second World War.

Despite advances in MSW management practice since 1990, the 2004/2005 fourth report by the House of Commons Environment, Food and Rural Affairs Committee on Waste Policy and the Landfill Directive emphasized the paucity and unreliability of data particularly in respect of hazardous waste flows (House of Commons Environment, Food and Rural Affairs Committee, 2005). However, the report acknowledges the development and implementation of the WasteDataFlow project (WasteDataFlow, 2004) for the recording of municipal waste. Subsequently and in addition, the European Union [EU] funded SMARTGROUND programme has established a single data base for secondary raw materials [SRM] (SMARTGROUND, 2015).

Dino et al. (2016) propose landfills to represent “new ore bodies or future resources”. However, to consider all waste streams and to account for each material together with their respective quantities presents a serious challenge.

EURELCO estimates 80% of landfilled wastes to be from municipal disposal therefore, this study focusses on MSW. MSW is further defined in Section 3.1.

This project was established to challenge and review the situation occurring before the formal recording of materials disposed into landfill. The scope of this study is to explore methodologies for the prediction and evaluation of the potential resources within landfills by asking three questions:

1. Can historical and geographic data provide a means of predicting landfill mass content?
2. Do significant connections between sampling data and MSW content exist?
3. Have compositional changes in generated MSW impacted upon the wastes disposed into landfills?

These questions open a significant research field that requires far more than one thesis to resolve. To moderate this issue, seven objectives were established:

1. To undertake a literature review of landfilling to identify and assess the directions in which research has developed and in particular the current focus in respect of:
    i. Landfills: composition, resource content and value.
    ii. Methodologies for the identification of landfill content.
    iii. MSW generation modelling.
    iv. The evolution of chemical processes within landfills.
2. To develop a waste repository resource inventory for England.
3. To identify and evaluate historic data and socio-economic factors influencing MSW generation as indicators to landfill content.
4. To develop a model for the identification of materials within landfilled MSW so as to benefit decision taking.
5. To investigate the physical and chemical relationships between decomposition processes, leachate samples and landfill mass content.
6. To determine whether landfill leachate composition has changed as a result of changes in landfill practice.
7. Identify the material content of landfills using the model output.

1.2 The landfill paradox

In the Preface to their recent publication *Solid Waste Landfilling* Professors Cossu and Stegmann identify the requirement for landfills to provide a necessary geological depository or sink. Whilst the move towards the circular economy, away from a linear one, is a major policy driver within developed economies, the landfill should be considered “as a fundamental inevitable tool for use in closing the material loop” (Cossu and Stegmann, 2019 p.xv) that is landfills are very necessary and should be constructed and operated to the highest standards (Cossu and Stegmann, 2019).

Historically, waste disposal to landfill has created a legacy of negative environmental issues due, for the most part, to reactive waste disposal strategy and practice. With the birth of the environment movement in the late 1960s/early 1970s, waste and its disposal methods began to be viewed as flawed strategies (Jones and Tansey, 2015). Environmental uncertainties have subsequently become heightened not only because of the unknown chemical content of many landfills, their potential to emit greenhouse gases but also as a result of historic land reclamation policies where landfill sites were chosen for their proximity to coastal and wetland areas where erosion and flooding is becoming a major concern (Bawden, 2016, Brand & Spencer, 2017; Beaven et al., 2020). A developing juxtaposition proposes existing landfills can now be conceptualised as material stores where the outcome can be considered as either waste-to-energy [WtE] or waste-to-materials [WtM] (Hogland, et al., 2010, Danthurebandara et al., 2013, Jones et al., 2013). UK specific research is also being undertaken (Wagland, et al., 2019).

A necessary sink and/or a necessary evil? A store of value or a store of future contamination? These two paradoxes present a series of quasi-philosophical questions together with complex dilemmas for the engineer. From the Engineers perspective, to undertake either remediation or landfill mining (LFM) it will be necessary to ascertain both the hazard and resource potential contained within respective sites. Identification of the content of any landfill mass, whether in terms of value or hazard, is complicated by the heterogeneity of MSW and, following disposal, the significant transformations arising from chemical and biological processes that occur during stabilisation.

1.3 Waste in context

1.3.1 Defining and categorising waste

The European Waste Directive 2008/98/EC defines waste as any substance or object which the holder discards or intends or is required to discard (European Union [EU], 2008). This definition encompasses a significant range of materials derived from different societal generators which are classified as commercial and industrial (C&I) wastes, from the tertiary sector, construction, demolition and excavation (CD&E) wastes through to household wastes [HW] and other wastes (OW) which comprise primary and secondary sector wastes. The Organisation for Economic Co-operation and Development [OECD] categorises wastes as being household, biological, industrial or solid (OECD, 2003). Environmental commentators have tended to the more emotive where “the notion of waste generally refers to an imbalance” and reflects “the very profligacy of modern living” (Scanlan 2005, p.22) is representative of a great number of publications.

The definition and classification of wastes is necessary but not straightforward (Pocock et al., 2009). Wastes are discharged to the atmosphere (gaseous), to bodies of water (mainly liquid) or to the land (solid and sludges). This thesis is directly concerned with wastes disposed on to land. The EU definition of waste is accepted however, waste classification often results from subdivision into source, type or potential for reuse (Wilson, 1981). In one sense it is more practical to exclude the wastes not considered by this thesis which are basically industrial and construction and demolition wastes. However, some wastes from commercial or light industrial sources must be included as these were collected and disposed alongside household waste and are considered, historically, as forming the municipal solid waste [MSW] stream therefore defined by source. Furthermore, this is a narrower definition of MSW to that adopted by the EU which is determined by material type.

MSW composition has changed considerably and will continue to change (Coggins, 2009). This project encompasses the period from the termination of World War Two to the present. At the commencement of this period household MSW in the UK comprised, for the most part, the ashes from fire grates (Dawes, 1953, Higginson, 1964, Stirrup, 1965). Latterly and following the implementation of the Landfill Directive 1999/31/EC, disposal operatives are required to handle an increasing range of stated recyclables in addition to what collection authorities advise to be ‘rubbish’. In one sense, these statements move beyond ‘simple’ categorisation but what of the intervening period?

1.3.2 Income, consumption and increasing MSW generation

In a 1952 paper presented by J. C. Dawes estimated that local authorities were collecting and disposing of 10 million tons of domestic refuse (Dawes, 1952). Five years later, J. C. Wylie proposed refuse generation had risen to an estimated 13 – 15 million tons (Wylie 1957). By 1974/75 MSW for disposal had risen to 23,743,000 (Department for the Environment [DoE], 1976) further increasing to 25,833,000 tonnes by 2002/03 (Department for Environment, Food and Rural Affairs [Defra], 2018). This increase has been attributed to a number of factors by a plethora of writers. Significant increases in consumption levels and changes in consumer preferences, the increase in population and with-it urbanisation, purchasing power and rapidly advancing technology are imputed as root causes. Even the adoption of the larger 240 litre capacity wheelie bin, introduced to mechanise waste loading during collection, fostered and endowed additional waste disposal at the household (Barton et al., 1986; Parfitt *et al.,* 2001). To quote Wylie “we can measure our progress through the centuries by how fast our dustbins are filled” (Wylie 1959, p.9).

> “*Consumption is a motor of waste production . . . To understand waste, it is necessary to understand consumption in-depth and in particular the driving forces for consumption”* (Ekström 2015, p.2).

Whilst the consumer as an entity has always existed, what evolved was the throwaway society where impulse drove commodity replacement and with it the development of marketing and packaging (Packer, 1960; Strasser, 1999; Tammemagi, 1999; Campbell 2015a; Bonneuil & Fressoz, 2017). However, this may be too simplistic as, of equal significance is the desire to improve living standards (Duesenberry, 1967).

Consumerism reflected the change in living standards coupled with the positives derived from increased job security and state provision for the unemployed (Marwick, 2003; Scott, 2007). Average income levels in real terms increased from

£5.08 per week in 1950 to £84.00 in 1980. Both the desire for and availability of durable goods increased during the period. Durable goods include furniture, electrical appliances (television sets, radios etc) but do not include motor vehicles or motorcycles. Whilst per capita consumer expenditure increased from £188 per annum in 1950 to £2410 in 1980 expenditure on durable goods increased from £8 to £115 over the same period. The increase for cars and motorcycles was somewhat larger, £2 increasing to £145. Section 1.3.1 identified the major component of post-war MSW to be ash. Increased expenditure effected a change in the composition of household MSW. In addition, the rise of consumerism brought with it a revolution in retailing where loose goods gave way to the era of packaged ones and large, supermarkets replaced the high street grocer which were then replaced by out-of-town superstores (McArthur et al., 2016).

Greater detail in respect of household purchases can be obtained from the UK National Archives [UKNA]. The Retail Prices Index [RPI] basket of goods and services serves as an indicator of how household expenditure evolved hence, how the content of dustbins changed. The 1952 listing contained only 3 electrical items, a vacuum cleaner, an electric fire and an iron. Neither wine nor coffee were included, and available fruit and vegetables were either fresh in season, dried or canned. By 1960 electrical items numbered 11 but out of season fruit and vegetables were now available due to importation or could be obtained as frozen. By 1980 electrical items had increased to 19 with the inclusion of colour television and hi-fi equipment. Many other items became available but of significance was the addition of household chemicals and the loss of coal and canned war-time staples. Into the 1990s and 2000s the list includes video recording equipment and tapes, sporting essentials, takeaway meals, mobile phones and computing equipment (UKNA, 2016).

Whilst motor vehicles have traditionally been resold and then recycled through breakers yards certain components (tyres, seats, facias etc) were landfilled. Consumer durables were often resold or passed-on to others but, at the end of their useful life helped to change the composition of municipal solid waste and, for the most part was landfilled. Table 1.1 identifies the changes to the major components forming the municipal household waste stream. These data are taken from sampling experiments undertaken between 1937/38 to 2006/07 and

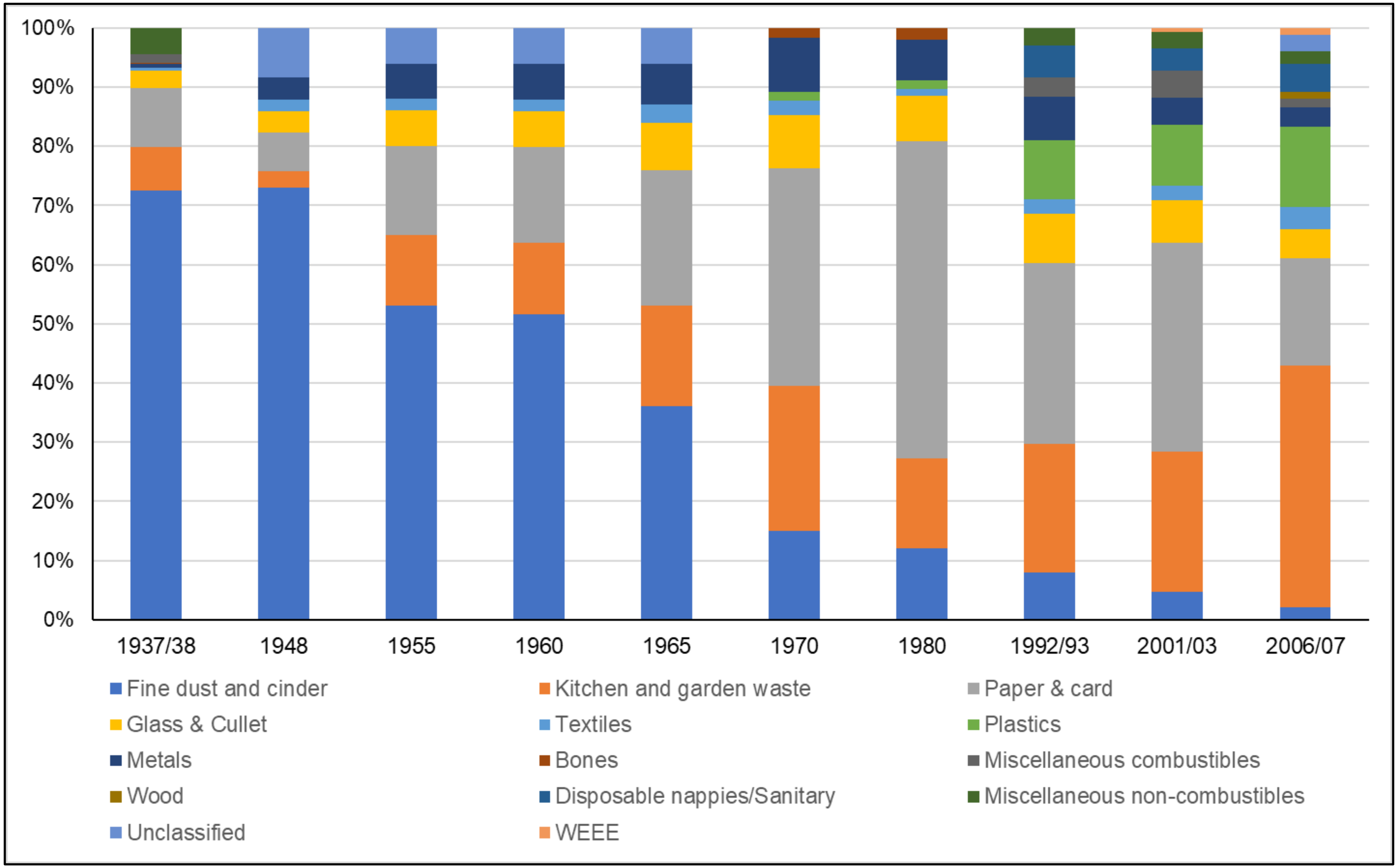


Figure 1.1: MSW composition as percentages generated 1937/38 – 2006/07. Sources: 1937/38 &1948: A.E. Higginson, 1960. Post-war Average: F.L. Stirrup, 1965. 1955, 1960 & 1965: Flintoff and Millard, 1969. 1970: Pencol, 1972. 1980: Neal, 1979. 1992/3 Parfitt 2001. 2001/3: Burnley 2006. 2006/7 DEFRA

reflect the changes in human consumption and also the flow of materials into the waste collection and disposal mechanism. Two almost contradictory outcomes are associated with the changes in MSW composition. The first, was the progressive and substantial decrease in the generation of dense materials, ash and cinders, to be replaced with lighter materials paper, card and plastic. The likely outcome should be a reduction in the weight of MSW with a concomitant increase in volume, However, both volume and weight increased significantly.

1.3.3 Local authorities and waste strategies

The history of solid waste management in England is effectively described by David Wilson from the Middle Ages although successive endeavors before the mid-nineteenth century are better represented as street cleansing (Wilson, 2007). With the implementation of public health statutes in the UK from 1848, superfluous items or for that matter, whatever was deemed as no longer required could easily be burnt (Girling, 2005) or deposited in landfill sites although these were uncontrolled with their content unrecorded and presenting future generations with an unknown, nevertheless heterogeneous material mass (Jones *et al.,* 2013). Landfill sites, for the most part, were conveniently located and often offered an additional land reclamation function. The landfill site became a repository for anything organic, metallic, demolished, excavated and much more for the simple reason landfilling was cheap to administer and operate.

MSW collection and disposal was organized by local government under what must be considered a notionally voluntary capacity (Stokes *et al.* 2013). With no effective central government involvement there was little strategic waste planning at any level. Until the early 1970s, regulation in respect of collection and disposal was all but non-existent. The 1848 and 1875 Public Health Acts laid the foundation for the municipal activity known as refuse collection and disposal, but the Act offered little in respect of guidance or how respective systems should be operated or controlled. The 1936 Public Health Act consolidated previous regulation and provided for local government to employ waste specialists should they choose to do so. Nevertheless, what constituted MSW, or refuse was often determined in the Law Courts (Ministry of Housing and Local Government, 1967).

Before and during the Second World War MSW collection and disposal in the England and Wales was undertaken by fourteen hundred and sixty Local Authorities (Ministry of Housing and Local Government, 1967). By 1970 this had reduced to one thousand, one hundred and sixty-five formed from county boroughs [CBs], borough [BCs], urban [UDCs] and rural district councils [RDCs] (Department of the Environment, 1971). This remained the case until 1974 when local government was reorganised reducing the number of Local Authorities [LAs] to four hundred and twenty-two. Waste collection became the remit of the new district councils [DCs]. Disposal was transferred to county councils [CCs]. London's waste has been reviewed a number of times. As the metropolis expanded the County of London was replaced by the Greater London Council [GLC] in 1965, which was then dissolved in 1986.

Financial sensitivity was often the key driver for both the collection and disposal of MSW. Contemporary waste professionals were aware of the requirement to improve disposal methods. Published public accounts for England and Wales show a significant increase in LA expenditure between 1935 and 1980 from £454.8 million to £25,264.9 million, an increase of some 5,555%. Figure 1.1 shows this significant increase against a 1980 base year.

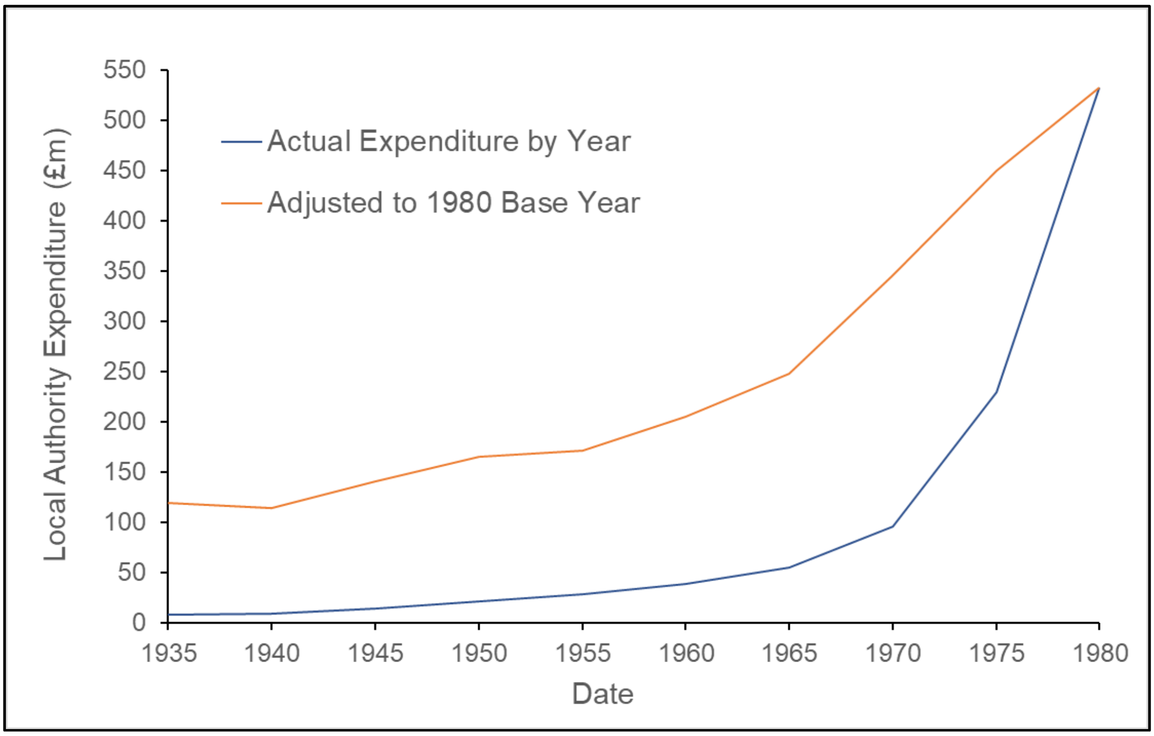


Figure 1.2: Local Authority expenditure on MSW collection and disposal 1935 – 1980. Real term adjustment provided by the Bank of England, 2020.

During the period collection and disposal costs remained at approximately 1.9% of total expenditure (Mitchell, 2011). The statistics cited by Mitchell are for England and Wales and represent both collection and disposal. A further breakdown is not available. However, it was proposed that for the financial year 1966/67 the costs to LAs of disposal totalled some £13.5 million added to which the costs of collection some £45.5 million (Department of the Environment, 1971). In addition to the increasing financial burden of collecting and disposing of waste, many LAs were affected by supply issues. These included restrictions on the supply of materials, particularly steel, which impacted on the requirement for waste bins and collection vehicles. MSW operatives were also in short supply due mainly to the nature of the work (Finn, 2016). This was the period of the "Stop Go" economy and by the mid-1960s early 1970s saw a number of developing social problems which continued into the 1980s (Worswick, 1952; Marwick, 2003; Scott, 2007).

Options for disposal by LAs were limited. Data published by the Ministry of Housing and Local Government (1967) identified the near universal adoption of landfilling by LAs:

- Direct tipping to land 90.4%
- Composting 0.3%

With some LAs adopting pre-treatments:

- Separation/incineration 7.6%
- Direct incineration 0.7%
- Pulverization 1.0%

Pre-treatments were employed to reduce the volume of collected municipal solid wastes, following which the residues were invariably landfilled (Department of the Environment, 1971). Additional data will be provided in Chapter 3 to reinforce the example taken from 1966/67.

Volume reduction was employed where MSW was required to be transported in bulk following deposition and treatment at a dedicated transfer station. Treated MSW was then transported by rail, barge or larger goods vehicles unsuitable for the collection operation. Data for pre-treatment operations is extremely limited. It was recorded that approximately eight percent of waste was handled by transfer stations in 1975 and that increased to at least ten percent by 1980 (Wilson, 1981). Pre-treatment was popular within the London Boroughs as potential disposal sites were more valuable as development land (Flintoff, 1950). The term “separation of materials” as used in contemporary literature is somewhat nebulous and should not be confused with current household recycling. Separation was usually direct – the recovery of metals, usually ferrous by means of magnets or indirect which were materials better suited for combustion. Research was underway at the Warren Spring Laboratory for the post-recovery of mainly non-ferrous metals from incinerator residues (Ministry of Housing and Local Government, 1967). Public Cleansing Returns give an indication of the type of materials and weights salvaged. Of significance is the quantity of paper reclaimed and the continuing reduction in the quantities reclaimed. Table 1.2 reproduces these data.

Table 1.1: Materials salvaged by LAs from collected waste over four financial years 1963/4 to 1966/67 (Ministry of Housing and Local Government, 1967).

Year	Total (tonnes)	Wastepaper (tonnes)	Scrap metal (tonnes)	Other materials (tonnes)
1963/4	415,000	256,500	65,950	92,550
1964/5	352,300	225,440	58,340	68,520
1965/6	389,900	256,800	60,130	72,970
1966/7	317,600	227,960	57,110	32,530

Uncontrolled and indiscriminate landfilling were known to be deleterious to those living adjacent to the disposal sites. Not only were uncontrolled sites unsightly as depicted in Figure 1.3 but they were infested with vermin, scavenging birds and insects. They also created nuisance issues with windblown litter and smoke from fires as well as being highly odorous due, for the most part, to the decay of putrescible wastes. The Ministry of Housing and Local Government were also conducting experiments on the polluting effects of landfilled MSW on groundwater (Ministry of Housing and Local Government, 1961). To overcome, but not necessarily eliminate, these issues the practice of controlled tipping resulted from nine recommendations or “precautions” issued by the Ministry of Health so as “to avoid giving offence and without risk to the public health” (Bevan 1967, p.6).

Figure 1.3: Uncontrolled tipping site reproduced from Department of the Environment (1971, p. opposite 98)

The first recommendation, formed from five sub-precautions, identified the need to layer and cover refuse within twenty-four hours of deposition. Controlled tipping was pioneered in Bradford during the 1920s and further developed as a result of original research undertaken in 1932 by Jones and Owen. Their research, known as the Manchester Experiment, identified the significance and chemical products of bacterial action on the waste mass. Furthermore, their work examined specific temperature ranges necessary to destroy pathogens contained within the decomposing heterogeneous waste mass.

Figure 1.4 illustrates an operational controlled site during the 1960s. Controlled sites differ significantly from modern engineered landfills which are also known as sanitary landfills. In a contemporary context, controlled sites may be considered as dispersal sites having no effective barrier to prevent egress of landfill leachate into the surrounding environment. Engineered sites operate as containment sites and are constructed over an impervious barrier which lies under a drainage system designed to collect and distribute leachate.

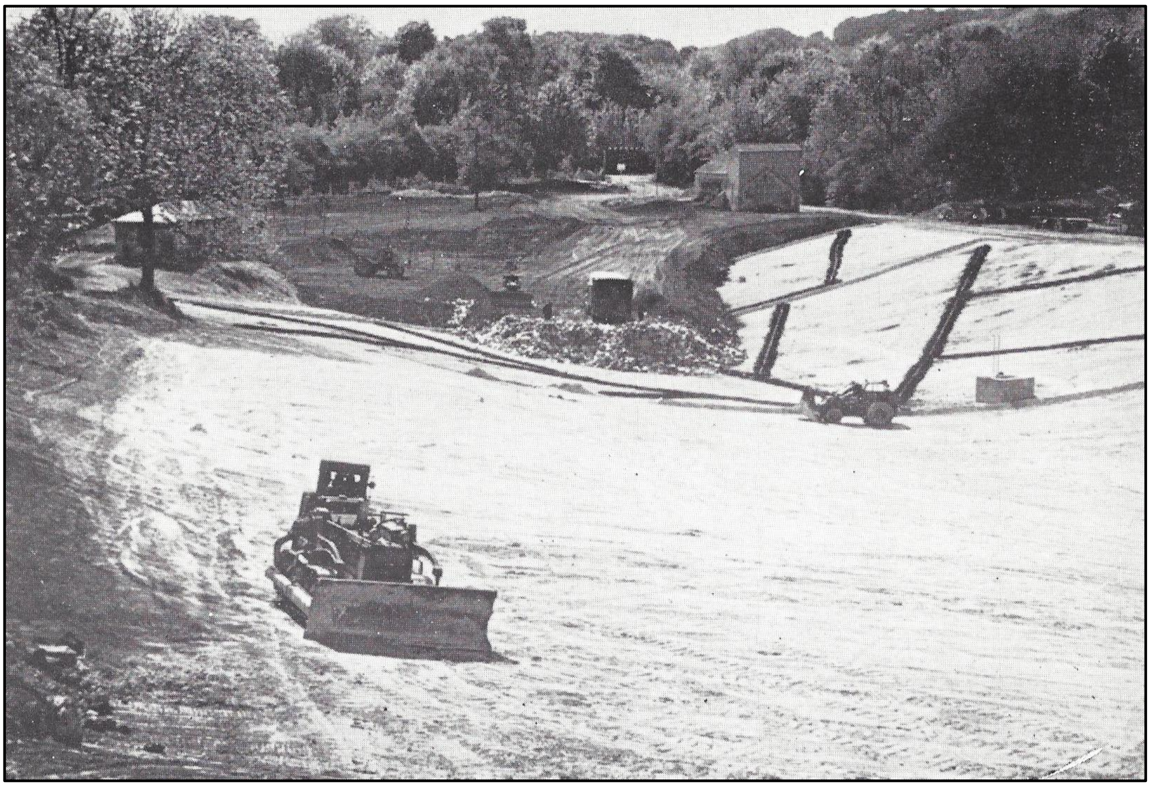


Figure 1.4: An operational controlled waste tip located in a disused chalkpit reproduced from Department of the Environment (1971, p. opposite 98)

In most respects, these descriptions form the boundaries of two conflicting disposal philosophies which cannot exist at the practical level (Knox, 1989). However, dispersal methodologies in landfill disposal are now considered as unacceptable. Containment is provided by an engineered barrier forming the base of the landfill. Gases and leachate generated during the stabilization process are collected. Gas is either flared or scrubbed to enable discharge into the National Gas Grid. Leachate undergoes either on-site treatment or is discharged to the local sewage treatment works.

The economics of refuse collection and disposal remained central to all LAs. Whilst there existed "the moral obligation for putting to the wisest use a vast bulk of material of heterogeneous content" (Flintoff, 1950 p.103). This moral obligation was never realized principally because of the practicalities of achieving it (Flintoff, 1950; Department of the Environment, 1971). However, two additional factors advanced the adoption of landfilling. First, the lack of any national strategy to the contrary (Department of the Environment, 1971) and second, the consensus that landfilling provided a land reclamation benefit particularly in the establishment of recreational facilities (Flintoff, 1950; Dawes 1953; Stirrup, 1965; Ministry of

Housing and Local Government, 1967; Bevan 1967; Department of the Environment, 1971; Bridgewater and Lidgren, 1981; Wilson 1981). Figure 1.5 (a) and (b) illustrate how landfilling could be turned to the ‘public benefit’.

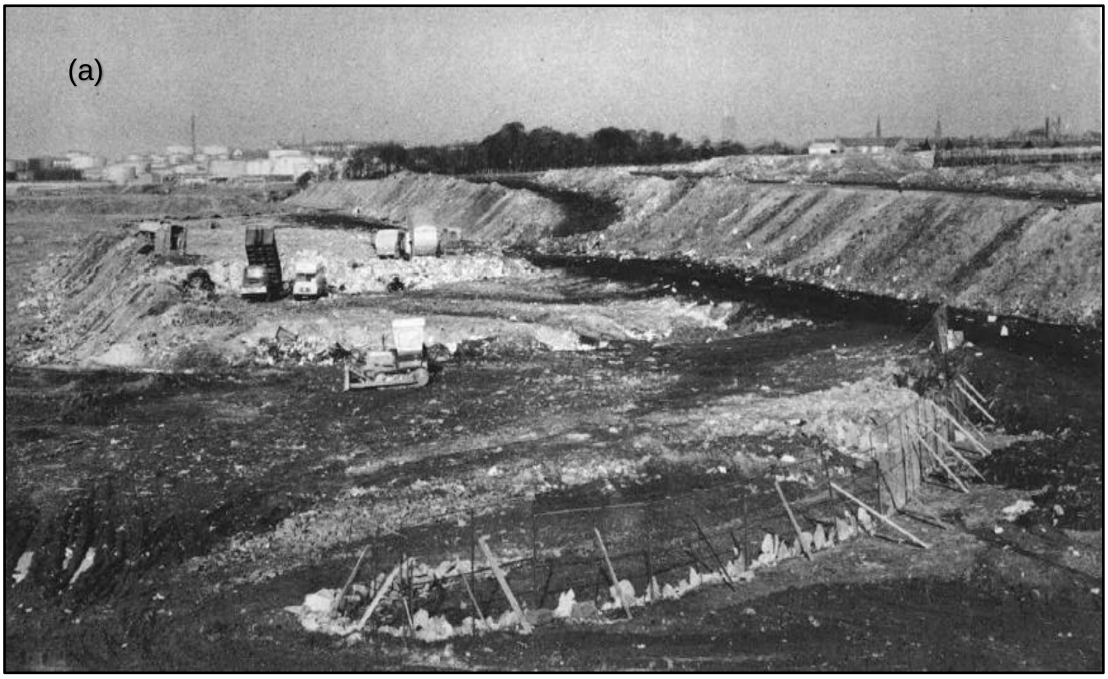


Figure 1.5 a & b: City of Liverpool, Otterspool Riverside Promenade 1929 – 1969. Phase 1 (Plate a): 43 acres, phase 2: 113 acres. Circa 6 – 7 million tons of household waste tipped to reclaim part of the River Mersey shoreline (Plate b). Reproduced from publicity publication dated May 1971.

The practical solution offered by landfilling dominated LA disposal strategies until EU directives 91/442/EEC, 96/61/EC and the Landfill Directive 1999/31/EC.

1.3.4 The advance to regulation

Section 1.3.3 identified the ‘notionally voluntary’ approach to waste collection and disposal in the UK. Before 1972, legislation provided local authorities measures to control waste should it result in a public nuisance. However, the 1947 Town and Country Planning Act provided new regulations on land use and with it control of the siting of landfills. The 1956 Clean Air Act indirectly impacted upon waste disposal by prohibiting the burning of domestic waste and the use of coal in smoke control areas (HM Government, 1956). Direct regulatory control of waste and its disposal resulted with the 1972 Deposit of Poisonous Wastes Act and the 1974 Control of Pollution Act. Under the latter, responsible authorities were required to investigate, prepare and review the disposal of controlled wastes and how such disposal could be achieved. The Act required consultation with water and other local authorities. Furthermore, it introduced licensing for the disposal of controlled wastes. Controlled wastes were defined as “*a kind which is poisonous, noxious or polluting . . . is likely to give rise to an environmental hazard*” (HM Government, 1974). Unfortunately, the act made no provision in respect of disposal records. This legislation was superseded by the 1990 Environmental Protection Act. Schedule 2B of the act set out a broad definition of waste which, basically, is any material "*which the holder discards or intends or is required to discard*" and controlled waste is “*household, industrial and commercial waste or any such waste*" (HM Government, 1990). The act also introduced, under section 34, a “*Duty of Care*” applicable to producers and handlers of waste. This established a paper-trail for the transfer and disposal of wastes. However, this did not resolve the issue in respect of waste records. Subsequently, policy has been geared towards waste reduction and ultimately the drive towards zero waste. Figure 1.2 provides a complete delineation of legislation and directives germane to this research as a timeline where the bulk of regulation occurs after EU Directive 91/156. Membership of the EU has required implementation of its directives and regulations not only in terms of waste policy but, and more fundamentally, the adoption of pollution prevention and control measures. Furthermore, EU waste policy has sought to reduce both waste generation and its environmental impact.

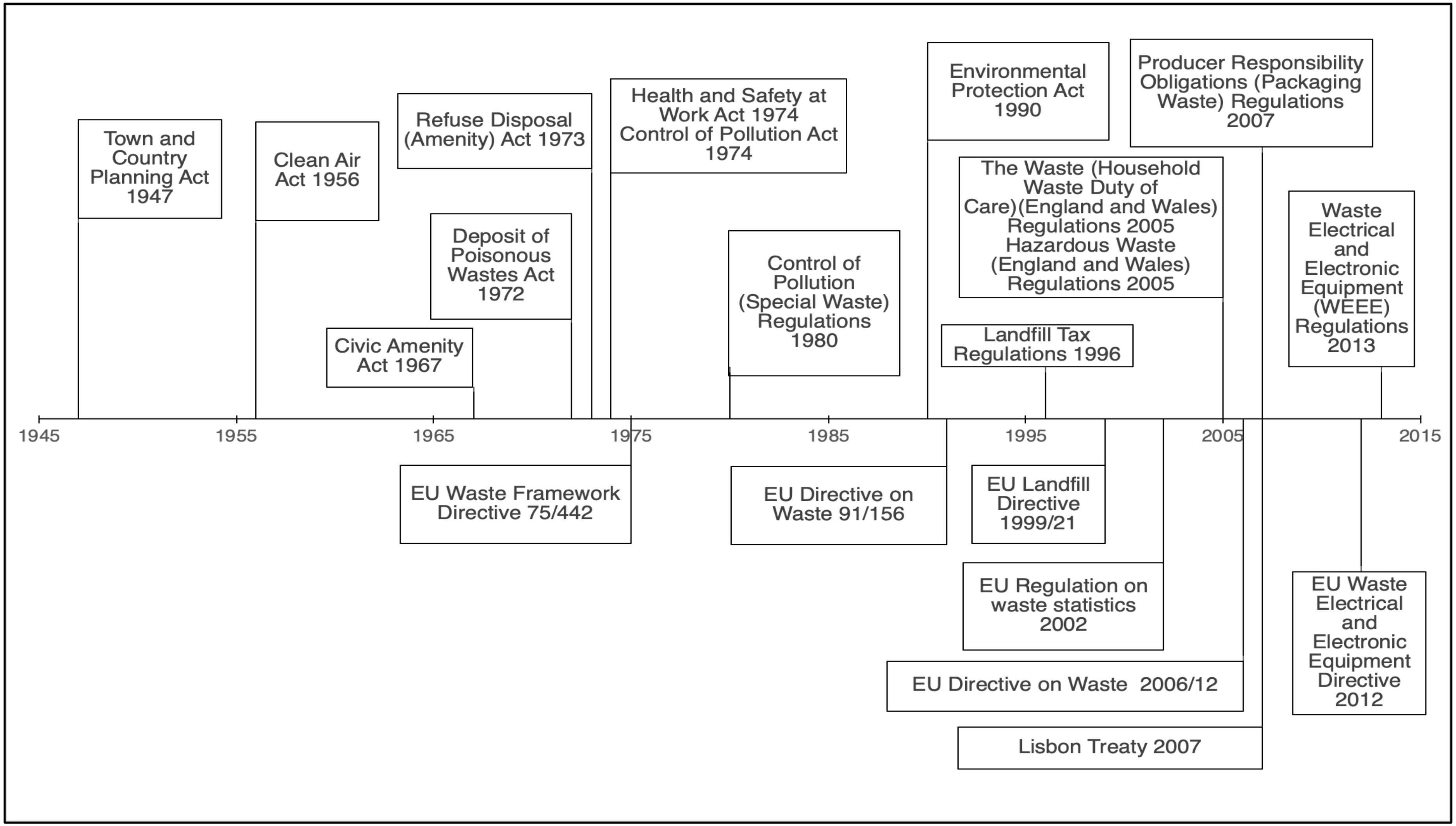


Figure 1.6: Timeline of the regulatory framework impacting on MSW flows to landfills. UK waste legislation (above x-axis) and EU Directives (below x-axis). UK legislation [Online]. Available at: https://www.legislation.gov.uk/ukpga Accessed [28th February 2019]. EU Directives [Online]. Available at: https://eur-lex.europa.eu/homepage.html Accessed [27th February 2019].

1.4 A proposed framework for modelling landfill content

1.4.1 Introduction

Historical MSW data collected for this study are required to fulfil two functions: i) provide effective indicators in respect of MSW generation and its material components together with identifying the quantities of MSW flowing to disposal alternatives and ii) operate as model inputs where the provenance and statistical validity of the data can be established. To function effectually, data should be representative of the population that is, it should be unbiased. Bias and representativeness of collected MSW data are considered in Chapter 3, Section 3.4 where the approach is to consider data sets as if they are samples by examining both their size and how each is representative of its population.

The issue with bias is exacerbated by the number of changes occurring across the study period. Figure 1.3 summarizes these changes and introduces a series of determinants and variables which are considered as either primary, secondary or latent. The final deposition of materials into landfills is dictated by a series of characteristics or a set of variables determined by these prevailing environments.

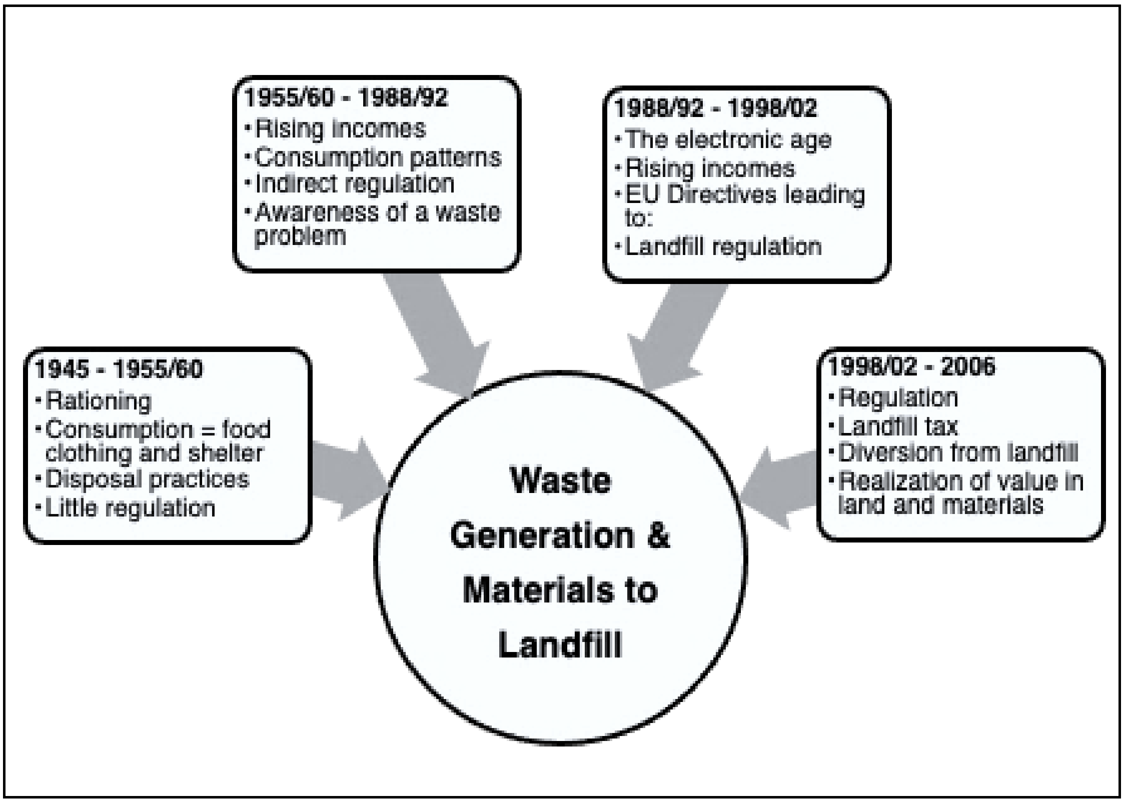


Figure 1.7: Major determinants impacting on waste generation and the materials flowing to English landfills 1945 - 2007.

1.4.2 Four epochs of landfilling

Collating respective determinants yields an original approach where four discrete time-periods or epochs are established. Figure 1.3 identifies each set of determinants that are effective MSW generation and disposal drivers for a given epoch. Furthermore, and by incorporating such a novel arrangement, data is managed within specific epochs which aids its justification as surrogate data. Given the starting and ending boundaries of 1945 and 2007, disposal options were dominated by landfilling although this continued until 2013/14 when landfilling, recycling and incineration each received approximately 10 million tonnes of MSW (Eurostat, 2017). Furthermore, and between the outer limits, the commencing and finishing dates between each period require some flexibility. The primary and secondary determinants for each period are identified below:

- Post-war Austerity: 1945 – 1955/60, the period of Austerity as defined by economic historians (Zweiniger-Bargielowska, 2000). Rationing determined consumption, average incomes were low and relatively stagnant (Figure 1.1) and for the most part, waste composition at the household level was fire-grate ash. Waste regulation was in most respects limited to the Public Health Acts however the 1947 Town and Country Planning Act allowed LAs control over site location. Over 90% of waste was disposed into landfills.

- The New Consumerism: 1955/60 – 1988/92, the period identified as the advent and development of the consumer society (Packer, 1960; Strasser, 1999; Tammemagi, 1999; Campbell 2015a; Bonneuil & Fressoz, 2017). Incomes increased significantly and the affordability of ‘white goods’ would provide consequent additions to the waste stream. At the commencement of the period, regulation impacted indirectly – the Clean Air Acts 1956 & 1968 and the provision of the National Gas Grid removed the reliance on solid fuels for domestic heating and so changed the composition of household MSW where paper and card, mainly packaging wastes, together with kitchen and garden (organic) waste become its main constituents. Her Majesty’s Government convened four reports from the Royal Commission on Environmental Pollution which initiated controls over the disposal of poisonous wastes.

The UK's membership of the European Economic Community [ECC] commenced on 1st January 1973. The first of the ECC pollution control Directives was issued in 1975 along with the adoption of the 'polluter pays' principle (European Union, 2020).

- Directives and WEEE: 1988/92 – 1998/02, 1990 saw the passing of the Control of Pollution Act which superseded the 1974 Act and initiated direct controls over landfilling and landfilling practices. Although licensing was introduced under the previous Act, Part 11, under Section 35, strengthened the licensing controls and, under Section 34, saw the introduction of the Duty of Care code of practice in respect of the management of wastes (UK National Archives, 2011). In summary, regulation to this point determined how municipal wastes were managed and how landfills were constructed and operated. MSW composition followed a similar pattern to the previous period with a further reduction in fines and ash content and increases in packaging and organic wastes. However, the 1980s saw the UK move into the electronic age with home computers and the sale of the first mobile phones. This resulted in the inclusion of waste electrical and electronic equipment [WEEE] in the MSW stream. This period witnessed both economic stagnation and growth which led to the phenomenon of "hyper consumption" that is replacement for replacement sake (Campbell, 2015b). Such a practice might improve the value of materials disposed into landfills – landfilling remaining as the major disposal route (Eurostat, 2017) despite the drive to initiate disposal alternatives, develop recycling and the introduction of a landfill tax in 1996.
- MSW Becomes Resource: 1998/00 – 2007, In this final period regulation determines not only the mechanics of waste management; that is its processes but also what constitutes a waste or potential resource. Clearly there are still some socio-economic drivers, but these were present during the previous period. From the adoption of the 1999 Landfill Directive regulatory measures dominate waste disposal strategy. Two elements separate this period from the previous. The first

is the growth of diversion of wastes away from landfill. However, landfilling exceeded 20 million tonnes annually until 2006/07 whilst diversion whether as recycling, composting or incineration accounted for approximately 15 million tonnes. Diversion in total exceeded landfilling from 2009/10 but it was not until 2013/14 recycling exceeded landfill (EUROSTAT, 2017b). Whilst increases in taxation also provided an effective driver with landfill tax on 1st April 2019 rising to £91.35 per tonne of waste for the disposal of domestic MSW, the second element was the statutory requirement for the recording of the type and quantity of generated wastes.

1.4.3 Modelling landfill content with system dynamics

From its generation to its final disposal, whether in the household, business or the outdoor environment, the management of waste relies on a complex and integrated engineered network or system. Such systems are often modelled mechanistically, that is by identifying specific components or sub-units and building from the bottom upwards (Forrester, 1961). In defining system dynamics, the acknowledged creator of the methodology J W Forrester proposed it as a tool for “top management problems” (Forrester 1961, cited in Sterman 2000, p.41). Since then, the method has been applied to most academic fields where a ‘system’ exists (Sterman, 2000; Dyson and Chang, 2005). A system is any set of interacting components whose structure transmits actions or events which then have an effect on other members of the set. In this study the system is built on the system dynamics software Vensim®, a computer-based modelling system. The building blocks of system dynamics softwares are stocks, flows and variables (also known as converters) which are represented by simple, geometric shapes. In Vensim® the standard shapes are rectangular boxes which represent stocks, circles which represent variables and broad, black arrows which represent flows. This study is concerned with the material composition of the large number of historical landfill masses that exist. The methodology is to replicate the MSW system that flowed into landfills and quantify these using contemporary data where these exist and include estimated or surrogate data to fill gaps. Contemporaneous data are used to compile a series of scenario matrices so as to simulate the content of individual landfills, groups of landfills within a specific region or the total resource content for England.

The proposed system dynamics model comprises 3 sub-units: i) MSW generation, ii) MSW disposal and iii) MSW composition. In addition, the modelling framework is presented with 2 tiers of complexity. Tier 1 models MSW generation and disposal with Tier 2 combining the three sub-units. Figure 1.4 is a simplification of the system dynamics model and its sub-units. MSW generation is coloured blue, disposal pink and composition red with these colours maintained throughout the study. The stock (Total Annual Waste) is connected by a flow, a broad, black arrow which, in this case, represents the MSW flowing into the 1 of 4 disposal options (also identified as stocks).

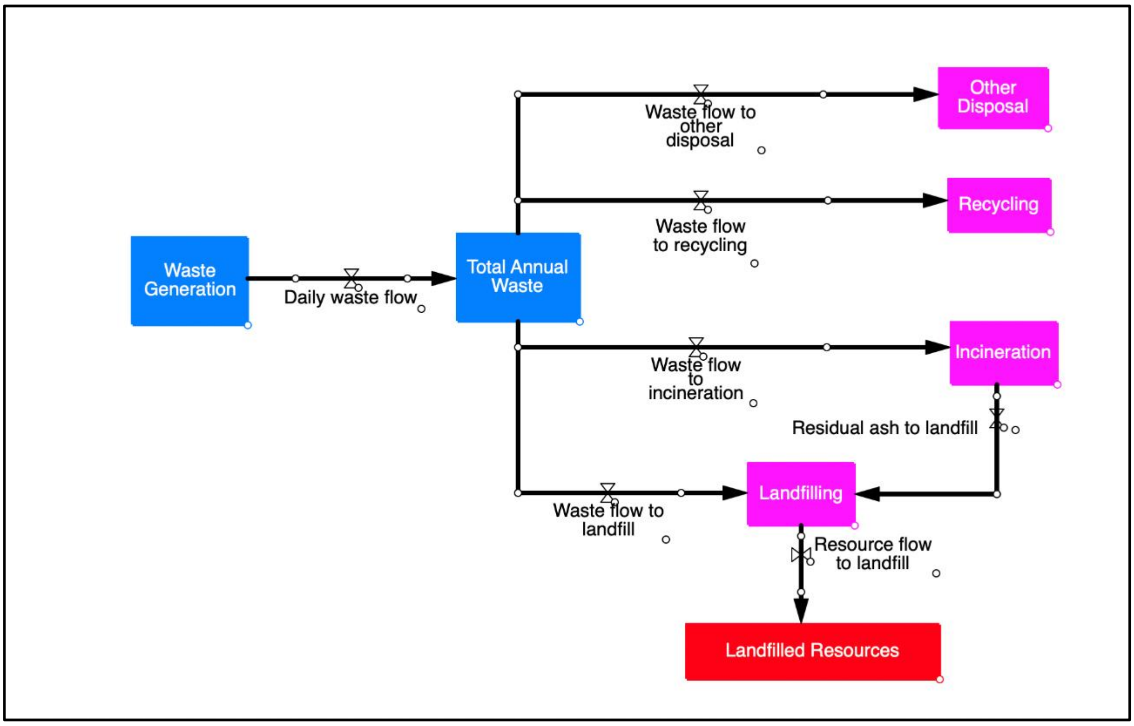


Figure 1.8: Abridged framework of the proposed system dynamics model. The model is built from stocks (the boxes), and flows (black arrows). The colours represent 3 sub-units: MSW generation, MSW disposal and MSW composition or landfill content.

1.5 Specific and non-specific data

Data used as inputs to the model variables falls into 2 classes: i) specific data where the data was sampled from or refers directly to a stated location or region and ii) non-specific data, usually in place of missing data or occasionally where data exists but is considered to be an outlier, then an average is taken over a data set or a surrogate value used which is based upon a similar background.

1.6 Thesis Overview

This thesis comprises seven chapters. This Introduction (Chapter 1) identifies the three research questions and this study's research objectives which include a proposed model to aid the identification of materials in landfills.

Chapter 2, the literature review for this study comprises two elements. The first element fulfils the function of what is expected for a thesis of this type whereby a structured review of relevant publications is presented. The second element required an extensive data gathering exercise and, whilst formed part of the overall review, employed a different methodology and is therefore dealt with separately and forms the focus of Chapters 3 and 4. Furthermore, a first review of the literature was completed during the early stages of the study (mid-2015) and then repeated during late-2019 early 2020 with the review focusing on 4 areas of research and identifying a gap in respect of landfill content estimation. Data gathering has continued throughout the study.

Chapter 3 examines historical data and evaluates two data elements, MSW generation and because these data were often published jointly, disposal data.

Chapter 4 investigates waste composition analyses and regulatory controls as a means to identifying material flows into landfills.

Chapter 5 (objective 4) develops a systems dynamics model that utilises available data as a predictor for flows of materials into landfills. The chapter also validates and reviews the proposed model against known depositions into UK landfills.

Chapter 6 (objectives 5 and 6) analyses the chemical evolution of landfill leachate. Leachate data from two new landfills, opened after the implementation of the EC Landfill Directive, is compared to earlier, "classical" analyses. The outcomes were published in Waste Management and Research in 2018.

Chapter 7 summarises the conclusions and provides recommendations for further work.

Chapter 2: Literature Review

2.1 Introduction

The review strategy was centred on four conventional criteria: i) to review research themes and direction ii) to examine historical and theoretical backgrounds iii) to explore connections between published science and this study and iv) justify this study within the context of that research (Ridley, 2012).

Two points arising from the review require explanation. First, publications typically report research from studies having localised, regional or a national foundation. In some respects, the geographic location of a study can be considered as peripheral, the basic science can be applied ubiquitously. Contrary to this, and where some modelling frameworks are reviewed, these utilise localised data as parameters and are therefore applicable to a specific location or locations comprising similar backgrounds. The intention of this review was to both discover and appraise as broad a spectrum of both knowledge and data as was feasible. In so far as data is concerned, this is in the likelihood suitable data is compatible and therefore considered as surrogate data. Second, there exist a number of data sources and articles out of circulation due to age or archiving. It is with regret the uncatalogued, hard copy research undertaken by the Warren Springs Laboratory and stored at the UK National Archive was not accessible.

2.2 Relevance to the aims and objectives

The literature review forms a necessary component of this study as identified by the first of the research objectives. The review process also provides opportunity to uncover data necessary to accomplish research aims one and two.

2.3 Methodology for the literature Search

Research into the management of waste, routes for its disposal, the escalation in waste generation and their negative impact on both the environment and public's health has developed rapidly. For that reason, the review was undertaken at the commencement and repeated towards the completion of the study. In each case the same bibliometric search tools were used with respective scopes based for the first on a mapping review and the second upon a truncated systematic approach identified by Booth et al. (2012). Each review utilized prior scoping and adaptation to meet the specific objectives of the research study. The developed scope identified where to search, what to search for and importantly the time spent in searching. This final point being significant as too little time spent impacts on quality and too much time limits work on other parts of the study.

To identify publications, Web of Science and Scopus were used for the first and second reviews together with websites containing conference proceedings in particular the International Waste Working Group and the International Solid Waste Association platforms. For the second review, Pubmed Reminer was included as an additional search tool. The search strategy utilized the advanced search facility in Web of Science and Scopus using key words and basic Boolean operators. The Systems Dynamics Society, International Waste Working Group and International Solid Waste Association platforms were also searched. Output for the first search was recorded into an Excel workbook, latterly into EndNote.

Specific search criteria concentrated on four areas of research pertinent to this study:

1. Landfills: composition, resource content and potential value. Section 2.4 reviews the scientific literature resulting from landfill mining where resource potential will become inextricably linked to the economics of reclamation.

2. Methodologies for the retrospective identification of landfill mass content. Section 2.5 reviews published literature directed towards the estimation or prediction of materials contained in landfills.

3. MSW generation modelling. Section 2.6 reviews published modelling methodologies as possible data generators. This section includes available system dynamics waste models which are thought to be the only peer reviewed examples of their kind.

4. The evolution of chemical processes within landfills. The second element of this study necessitated data collection. Resulting from the review of waste disposal and landfill data supplied by Viridor Ltd., it was observed that the low pH phase in 2 new MSW landfills had not occurred. Section 2.7 reviews research impacting on the chemical evolution within landfills. This research is developed in Chapter 6.

2.4 Landfills: composition, resource content and value

2.4.1 Introduction

This section's objective is to search and review research and data where landfilled materials (considered as resources) are reported as a result of landfill mining [LFM] projects. One of the motivations for this research study was to provide a useable model for the estimation of resources to advance any LFM decision-making process. LFM projects have provided a series of inventories of the resource content in excavated landfills. Whilst these are not from UK projects, they reflect a range of deposits that provides a useful comparison to that determined by modelling, given similarities in contemporary waste streams. LFM can be undertaken as an ex-situ (excavation) operation or using in-situ technologies (INSPIRE, 2014; Materials World, 2014). In-situ processes, in respect of LFM, minimise the mining impact and do not disturb or expose material content. As such, in-situ LFM specific research is not included in this review.

2.4.2 Search methodology and results

The search generated 295 peer reviewed papers. Further searching by limiting to English language publications, those that had a reference to waste and landfill content or resource content, case studies, evaluation and economics (of landfill mining) reduced this total to 126 papers. In all some 35 papers dealt with the cost effectiveness of LFM and 4 publications are literature reviews examining LFM research dating from 1988: i) Krook et al. (2010), ii) Krook et al. (2012), iii) Krook and Baas (2013) although this review is specific to a special volume of Journal of Cleaner Production and iv) Parrodi et al. (2018) review published research on fine fraction characterization. Both Krook et al. (2010) and Krook et al. (2012) are similar, the second being a journal publication of a conference presentation. Each reports on 39 peer reviewed papers using similar search methods to that used here. This review identified a similar number. The 2013 paper considers LFM and urban mining. Urban mining refers to the release of 'hibernating' resources, that are resources lying within communities and outside of traditional waste management systems. Post-2013, 9 reviews have been published however, these consider specific factors related to landfill mining as distinct from published literature in a more general sense.

It has been estimated over 90% of Europe's landfills pre-date the adoption of sanitary landfilling (Jones, 2016). Extracting value from materials disposed into landfills has received considerable attention either as landfill mining, landfill reclamation or as a means to off-set costs of remediation or long-term aftercare (Krook et al., 2010; Krook et al., 2011). From the resource perspective, landfill mining [LFM] is associated with either waste to energy or waste to materials (Hogland et al., 2010; Danthurebandara et al., 2013; Jones et al., 2013). Jones et al. (2013) propose a combination of waste to energy and waste to materials forming an integrated Enhanced Landfill Mining [EFLM] concept. Inextricably linked to material reclamation is the benefit LFM can bring to greenhouse gas reduction by the removal of recalcitrant organic materials which account for longer-term decomposition (Laner et al. 2016). Reclamation can either be as a means to extend the life of a working landfill or to treat the landfill as contaminated land and remediate for development (U S Environmental Protection Agency, 1997 [EPA]). Remediation seeks to prevent the content from continuing to be a source of pollution whilst salvaging materials to ameliorate remediation costs.

The review undertaken by Krook et al. (2012) identifies five distinct categories in which each of 39 papers can be allocated (Figure 2.1). For this review similar categories were selected however, realization is included with economics and technology and conceptual papers have been incorporated as Other.

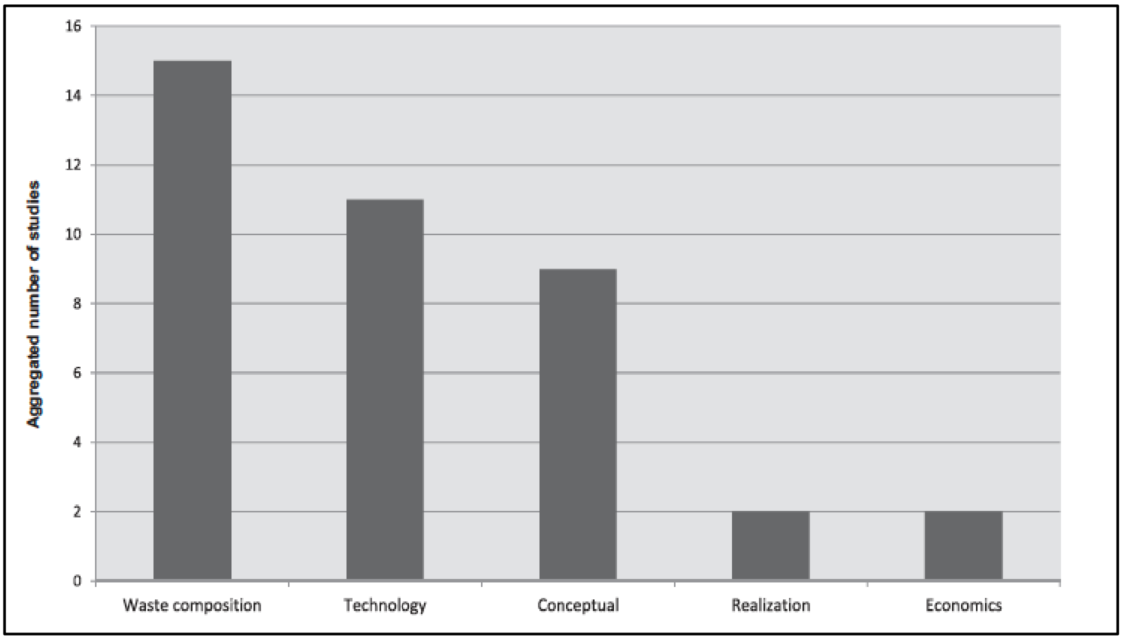


Figure 2.1: Categorization of thirty-nine papers reviewed by Krook et al. (2012). Figure reproduced from Krook et al. (2012 p.515).

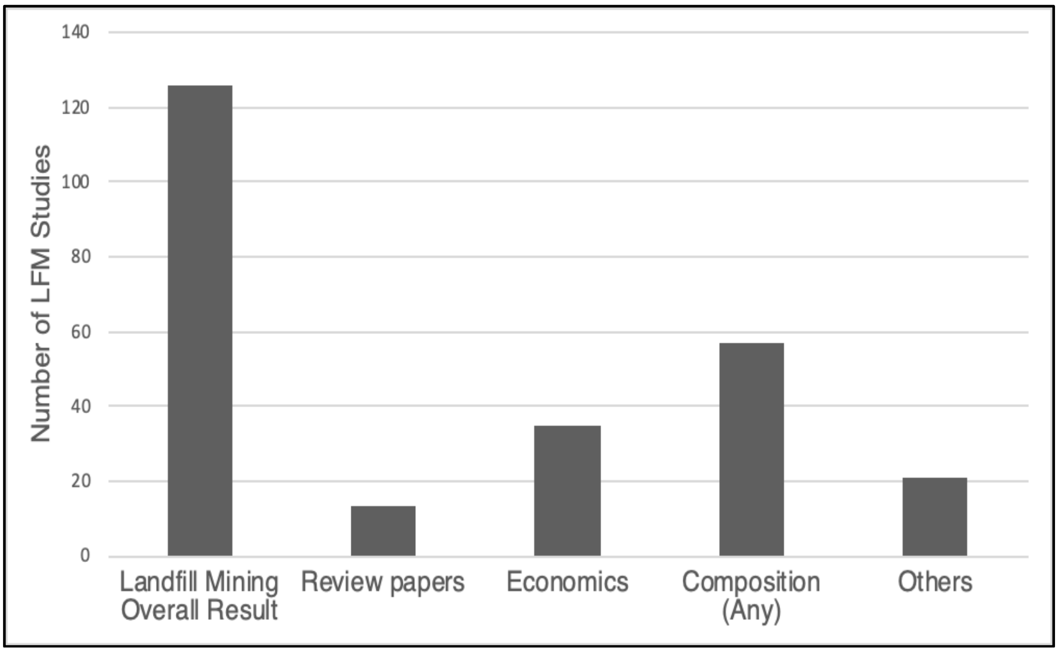


Figure 2.2: Categorization of 126 papers reviewed by this study.

Figure 2.2 includes those papers included by Krook et al. and reflects the increase in research activity in respect of landfill mining generally.

2.4.3 LFM material analysis

Some 57 publications were identified as containing either physical or chemical characterization analyses of exhumed materials. Potential resources are mainly characterized into nine categories. These are summarized in Figure 2.3.

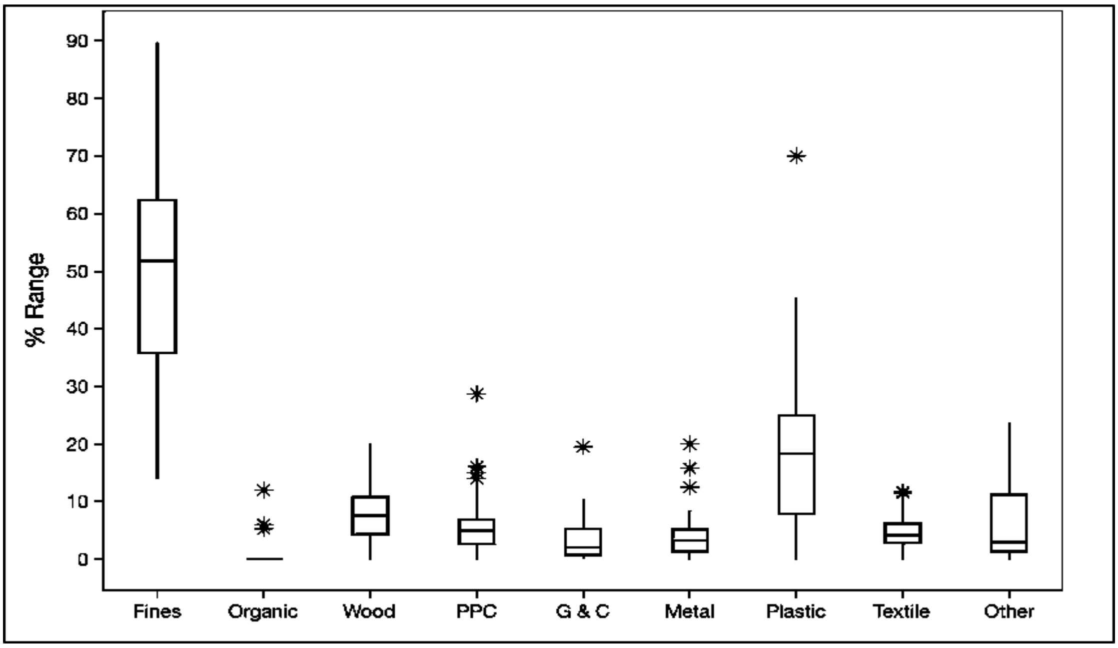


Figure 2.3: Physical characterization of landfilled materials. For LFM projects, fines range 28-65%. Minimum values arise from a Thailand study by Prechthai et al. (2008). PPC is paper packaging and card. G&C is glass and cullet.

Minitab was used to analyse the data where outliers are defined as those observed values that exceed one and a half the interquartile range, with these being denoted with a star (*). For each variable, the median is indicated by the horizontal line within each interquartile range. The data range (excluding outliers) extends to the end of each whisker. A detailed table is included as Appendix 2.1.

From the papers reviewed, MSW landfills contain 36 – 62% by weight of cover materials combined with heavily degraded organic waste or humus within their interquartile range. Degraded wastes originate from biodegradable wastes that occur in MSW as food and garden wastes. These rapidly decomposing organic wastes have decayed, most likely, to a first order decay function with a value for k similar to that used in methane generation (Quaghebeur et al., 2013). From the collected data, recalcitrant organics from paper, wood and plastic form twenty to thirty percent by weight of material content. Approximately ten percent of inorganic materials are from building products, concrete glass and the like and a small percentage of metal; more often than not, ferrous metal. In addition, two further points were observed: i) this configuration appears near to ubiquitous for those sites and materials included and ii) the presence of hazardous waste, for the most part, was recorded at below one percent (Jennings et al., 2007; Krook et al., 2012; Frändegård et al., 2013). However, and by comparison, data from excavations into the Queen's Road Landfill, Manchester for early 1960s refuse exhumed one-year after deposition are: i) weight of cover materials combined with heavily degraded waste a range of 62 – 82 percent (4 samples), ii) organics from paper, wood and plastic 8 – 20.2 percent (4 samples), iii) metals 2.3 – 6.8 percent (4 samples) and iv) glass 2.5 – 6.2 percent (4 samples) (Bevan 1967). These data reflect a specific range determined by a range of MSW generators.

This study's focus was the review of potential resources identified within LFM publications. LFM research has employed trial pitting and boreholes to remove materials where the extent of sampling is shown to be extremely limited when compared to landfill site areas, depths and volumes imported. Prechthai et al. (2008) identify a site area of 108,000m^2 with 4 trial pits excavated using a backhoe excavator resulting in 12 number, 150 kg samples. This review has shown this is typical with Quaghebeur et al. (2013) confirming such sampling is inadequate. A similar opinion results from this study where it is clear significant sampling will be necessary to determine a landfill's content. Both borehole drilling and trial pitting are expensive and comprise practical limitations for sampling. Additionally, there is the cost of analysis. A data-based assessment, as proposed by this study, presents an opportunity to avoid unnecessary expenditure by identifying resource potential before undertaken intrusive investigation.

2.4.4 LFM economics

Krook et al.'s review considered that 40% of papers reviewed made an economic statement but this was subsidiary to the research. However only two publications Fisher and Findlay (1995) and Van der Zee et al. (2004) centred directly on the economics of LFM (Krook et al., 2012). Krook et al. consider this was because no common evaluation framework was available due, for the most part, to the unique issues offered by each project.

Subsequent publications have focussed on two areas. First, the viability of LFM as a standalone project and second, more by way of a corollary, the value, if any, contained in the sizable fines fragment. Van Vossen and Prent (2011) and Münnich et al. (2013) considered the ability of LFM to be economically sustainable in respect of stand-alone projects. However, the likely out-turn of refuse derived fuel [RDF] or waste to energy potential available within respective landfill matrices led to the proposal of ELFM (Chiemchaisri C., 2010; Hogland et al., 2010; van Vossen and Prent, 2011; Spooren et al., 2013; Jones et al., 2013; Quaghebeur et al., 2013; Jones et al., 2016; Wagland et al., 2017; Särkkä et al., 2017; Küppers et al. 2019; Wagland et al. 2019). In respect of the value contained within the fines content (material < 20mm) arising from LFM projects was replaced into the landfill (Münnich et al., 2013; Bhatnaga et al., 2017). Research attention has extended to identifying value content within the fine fractions (Zanetti M. and Godio A., 2006; Jani et al., 2016; Mönkäre et al., 2016; Kaczala et al., 2017; Parrodi et al., 2018; Parrodi et al., 2019a; Parrodi et al., 2019b) in particular their elemental metal content (Wolfsberger et al., 2015b; Wagner and Raymond, 2015; Kaczala et al., 2017; Parrodi et al., 2018; Parrodi et al., 2019b; Faitli et al., 2019; Lucas et al., 2019; Wagland et al., 2019). Overall, however, conclusions were similar in that resource reclamation was not economically viable. Published reviews assessing the viability of LFM offered two conflicting opinions depending upon particular site metrics where it was possible to generate a positive or negative net present value (Laner et al., 2016; Esguerra et al., 2019; Laner et al., 2019). Esguerra et al. (2019) recommend future projects should be learning-orientated so as to determine routes to profitable projects.

Economics and material reclamation either from landfill mining or urban mining are inextricably linked. Rosendal et al. (2017) reported results from an experiment where 2,049 tonnes of material was excavated from imports totalling 45,000m^3. To date this presents the most comprehensive sampling and, furthermore, concluded the costs of LFM would exceed likely revenues, which is also the current opinion resulting from this study. To allow the emerging circular economy paradigm to develop, resource reclamation as a principal component, whether from kerbside collections or materials buried in landfills, will need to overcome the fundamental economic law of supply and demand. With the very large quantities of materials deposited in landfills potential resource supply might appear abundant however, material separation is a significant and costly issue. In urban recycling technologies are being developed to resolve this problem (Messenger, 2018). However, the implementation of the true circular economy will require not only the physical removal of a specified material from a particular waste stream but will develop to include the extraction of specific chemical components from the material matrix in which target materials are bound (Bailly and Tayart de Borms, 1977, Bazargan et al., 2015). Twenty-two chemical elements are identified as critical where known reserves will be depleted as a result of extraction or political control during the next five to fifty years (Hunt et al., 2015).

2.5 Methodologies for the identification of landfill content

2.5.1 Introduction

Whilst previous sections of this review have provided the context and a detailed appraisal of existing research, this section identifies a specific gap in the knowledge base where different approaches are utilized to estimate landfill content. The review includes a report prepared by MEL Research describing commercial and industrial waste flows into UK landfills, Section 2.5.4. The MEL report identifies similar problems in respect of data shortage together with strategies to overcome these.

2.5.2 Search results

Only 6 studies are directed towards proposals for estimating landfill content using non-invasive methods (Leach et al., 1995: Yokoyama et al., 2006; Lyons et al., 2010; Frändegård, 2011; Frändegård et al., 2013; Wolfsberger et al., 2015a). Leach et al. (1995) were commissioned by the UK Department of the Environment to review the availability of data so as to estimate the composition and quantity of industrial and commercial wastes entering English landfills. This study is discussed in Section 2.5.4.

Yokoyama et al., (2006) propose an adapted input/output model for waste to energy and carbon-dioxide emission comparison that result from resources generated from LFM. Japan has limited landfill resources, and the excavation and reuse of existing landfills presents an opportunity to resolve this issue. The authors make reference to the heterogeneity of landfilled waste resulting from many factors including local consumer preference, industrial background and disposal strategy. Data for the possible waste material input is estimated from: (i) an input-output equation based upon economic activity and (ii) a material analysis, acquired during an interview and based upon 4 samples (although these are not identified as such), taken at one site in Tokyo. Each sample indicates the waste composition to be predominantly stones, ceramics and other non-flammable content varying between 50.5 and 77% in content. The paper's concentration is the calorific value of the likely landfill content and how to best incinerate the arisings.

Frändegård (2011) and Frändegård et al. (2013) develop a decision support tool applying Life Cycle Assessment [LCA] combined with Monte Carlo Simulation to overcome uncertainties arising from the application of LCA when applied to landfill mining. The authors propose that any LFM project is burdened with significant risk particularly in respect of content and potential decision makers may be unaware of the full extent of uncertainties. To mitigate uncertainty in respect of content, Frändegård et al. proposed decision makers or actors to input either their own data in respect of potential landfill composition or rely upon a specific default composition based upon data obtained from reviewing sixteen landfill mining pilot projects, the 'hypothetical landfill'. For own data, the author's identify an intrusive process, but this led to significant uncertainty in respect of the landfill's waste mass.

2.5.3 MSW estimation in landfills

Two papers (Lyons et al., 2010; Wolfsberger et al., 2015a) propose modelling methodologies for the determination of landfill content. Their approach is based upon waste generation and the waste disposed into landfills. Lyons et al. (2010) challenge accepted methane emissions from Irish landfills using a consumption-based model. Wolfsberger et al. (2015a) propose a theoretical approach to estimate the secondary raw material potential [RMP] in landfills by determining material composition.

Lyons et al. (2010) describe a model for modelling methane emissions from Irish landfills using estimated data. The Irish Environmental Protection Agency [IEPA] do not measure (directly) the methane emitted from Irish landfills but use historical MSW data to estimate these releases. Lyons et al. propose the IEPA overestimate emissions because the MSW inputs into Irish landfills are overestimated hence greater than actual emissions result. Decomposition of MSW in landfills is slow therefore for an estimate to be credible historical data extending back to 1965 is required. Like the UK, this data was never collected in Ireland. Actual data is only available for the years 1995, 1998 and then consecutively from 2001. To overcome this problem the authors estimated the biodegradable waste [BW] content in household and commercial MSW dating back to 1965. Data for model parameters was obtained from published behavioural research and incorporated with the data for 1995, 1998, 2001 and thereafter in a constant elasticity demand model. The models parameters are based upon income, number of households, service sector production levels and commercial price levels. Their model output simulated different emissions from those obtained by the IEPA, using the values of biodegradable wastes generated over their specified period. Biodegradable wastes comprise a number of changing components. A constant elasticity-based consumption approach is limited where the waste generation variable is determined by household numbers and does not include location or type (Parfitt and Flowerdew, 1997). Furthermore, the model is limited when the waste generation response, as determined by changes in income, cannot include the dynamics of prevailing consumer preferences that result in changes to individual waste components. However, whilst the model is specifically concerned with the degradable carbon in the waste stream as the researcher's concentration are the gaseous emissions produced by the carbon content in MSW the use of estimated MSW is an approach adopted by this study.

The model presented by Wolfsberger et al. (2015) is proposed as an economic alternative to invasive site investigation. The model comprises two elements: i) an analysis of historical MSW composition data and ii) theoretical calculations in respect of MSW degradation in landfills. Wolfsberger et al. proposed waste composition data could be incorporated into a general methodology to determine the raw material potential within a landfill. From a particular landfill's location waste imports, over a specific period, can be determined from respective waste component analyses. Where no data exists, then complementary data from a similar region, it is proposed, can be substituted. Waste content at individual, local landfills can also be accounted for by discounting wastes existing in local waste streams not meeting a particular landfills acceptance criteria. Additional historical data can be found from the landfill operator's records, business registrations and witness testimonies. This model is compared to this study's proposed model in Chapter 5 (Section 7.6) where a similar use of MSW component studies is utilized along with data substitution. However, in respect of published landfill imports, data availability varies significantly from country to country and in quality (Lyons et al., 2010; Rosendal et al., 2017). For the UK, recorded imports into landfills do not exist until the mid-1990s.

2.5.4 Industrial and commercial waste in landfills

Leach et al., (1995) working on behalf of MEL Research quantified industrial and commercial wastes entering English landfills. The study, funded by the Department for the Environment to, primarily, assess the level of industrial and commercial data available relied on secondary data taken from local authority waste management plans. Waste management plans were adopted as a result of Section 2 of the Control of Pollution Act 1974. Using data from these plans, waste output for specific industrial sectors was divided by employee numbers to achieve a figure for waste generated per employee for that sector. The research team were required to update existing waste plan data as significant gaps were found in the data together with inconsistencies in the detail many contained.

As a second approach, and as a route to verification, data already in the possession of MEL Research was incorporated to obtain a second series of estimates for each classification. Identified in the report as Survey Data, this was considered more reliable, due to these data being considered as primary data, rather than the secondary data obtained from the waste disposal plans. From these projections the quantities of wastes from different industrial sectors disposed into landfills was estimated to be 73.216 million tonnes using the Survey Data and 83.265 million tonnes using the waste disposal plan data. However, the analysis referred only to waste going to landfill with no specific landfills identified.

2.6 MSW generation modelling

2.6.1 Introduction

The desire to understand waste generation by sampling and scrutinizing existing waste streams is a well-established practice leading into World War II (Higginson, 1966; Coggins 2009). MSW modelling is seen as a predictive process not a retrospective one. However, as a predictor of future waste streams, the collection, sampling and separation of household MSW confronted two problems. The first is that it does not, sufficiently, incorporate temporal fluctuations or the many socio-economic and geodemographic variables contributing to waste generation (Boyd et al., 1971; Albert et al., 1974; Rufford, 1985; Parfitt et al., 1994). The second is that sampling does not provide the necessary ‘insight’ to enable prediction of either future waste composition or quantity (Boyd et al., 1971; Albert et al., 1974). This section reviews MSW generation modelling particularly the application of specific model types in the determination of missing historical MSW generation data, thus a retrospective application.

This study’s search of waste generation modelling literature located 7 review papers (Beigl et al., 2003; Morrissey and Browne, 2004, Dahlén and Lagerkvist, 2007; Beigl et al., 2007; Laurent et al., 2014; Kolekar et al., 2016; Ruiz et al., 2019) that had, for the most part, undertaken systematic reviews of specific waste modelling approaches. This has resulted in some overlap with those reviews, due in part, to the retrospective nature of this study, its search for data and the keywords used to search publications however, this study’s search objective is fundamentally different. As a result, the framework of this section of the review is to present the search results alongside those of previous reviews where they have relevance (Section 2.6.2). Following which, specific modelling approaches are considered in respect of this study's objectives (Sections 2.6.3, 2.6.4 and 2.6.5).

2.6.2 Search results

A keyword search for waste generation and waste generation modelling publications using Scopus, Web of Science and both the IWWG and ISWA proceedings websites generated 103 peer reviewed articles. Six publications are reviewed in the previous section. Eleven publications employed systems dynamics (Thirmurthy, 1992; Karavezyris et al., 2002; Dyson and Chang, 2005; Chaerul et al., 2008; Gönenç et al., 2008; Kollikkathara et al., 2010; Inghels and Dullaert, 2011; Chen et al., 2012; Dace et al., 2014; Golroudbary et al., 2015; Pinha and Sagawa 2020). Additional systems dynamics publications were obtained from the Systems Dynamic Society Conferences (Kum et al., 2004; Westbrook et al., 2012), two publications in the journal Systems Dynamics Review (Mashayekhi, 1993; Sudhir et al., 1997).

2.6.3 Existing reviews

Seven publications are review papers with Beigl et al. (2003) a conference presentation. Morrissey and Browne (2004) reviewed decision support tools based on cost benefit analysis, life cycle inventory and multicriteria models. Dahlén and Lagerkvist (2007) reviewed methods of analysing household waste composition and is referenced further in Chapter 4. Beigl et al. (2007) reviewed 45 modelling approaches characterising each model using 5 criteria: (i) by region (household, region, country), (ii) time series length (day, week year(s)), (iii) waste stream (household, collection, material), (iv) independent variable type (consumption, disposal, production, trade) and (v) modelling methodology (group comparison, statistical, input-output, time series, system dynamics). Laurent et al. (2014) critically reviewed 222 life cycle assessments of waste management. Kolekar et al. (2016) extended the 2007 review completed by Beigl et al. and included 2 further systems dynamics publications Kollikkathara et al. (2010) and Chen et al. (2012). Ruiz et al. (2019) reviewed construction and demolition waste.

The critical review undertaken by Laurent et al. (2014) centred on life cycle assessments and how MSW management systems for 4 waste types (plastic, paper, organic materials and mixed wastes) impacted on environmental sustainability. Of the 222 publications 8 are germane to the UK where 4 consider waste management scenarios for MSW (Craighill and Powell, 1996; Welsh Assembly Government, 2003; Emery et al., 2007; Tunesi, 20110 and 4 material specific: (i) glass recycling and the environmental cost of transporting glass to a collection point (Edwards and Schelling,1999), (ii) the collection and recycling of batteries (Fisher et al., 2006), (iii) the options for mixed waste plastic (Shonfield, 2008) and (iv) a comparison of three waste management options for wastepaper including bioethanol production, recycling and energy from waste incineration (Wang et al. 2012). Their benefit to this study was the data contained in the publications by the Welsh Assembly Government (2003) and Emery et al. (2007).

Beigl et al. (2007) include 28 publications located by this study (17 are published in German). Ten are pertinent to MSW or household waste with 2 publications, Karavezyris et al. (2002) and Dyson and Chang (2005) employing systems dynamics solutions for waste collection streams where data was incomplete. System dynamics publications are reviewed in Section 2.7.4. Kolekar et al. (2016) is offered as an extension to the review by Beigl et al. (2007). However, the search methodology failed to identify a number of publications from influential journals and does not offer any valuable or significant insight.

2.6.4 Modelling MSW generation

Reviewers are generally critical of many modelling approaches identifying the heterogeneity of both socio-economic factors and the waste itself as significant issues to overcome in a proposed model (Rufford, 1984; Parfitt et al., 1994; Parfitt and Flowerdew, 1997; Morrissey and Browne, 2004; Beigl et al., 2007; Laurent et al., 2014). As such, identifying a model to generate retrospective waste data is problematic. Beigl et al. (2007) state that:

> "*Unfortunately, the majority of these models are often unusable due to the lack of underlying data for the model parameters.*"

Researchers are aware of the issues associated with the paucity of data and have developed approaches to overcome the problem. These can be categorised as:

- Mathematical (Rufford, 1984; Ojeda-Benitez et al., 2008)
- Grey fuzzy dynamic modelling (Chen and Chang,1999; Orsoni and Karadimas, 2006)
- Neural network models (Antanasijević et al., 2013; Abbasi and El Hanandeh, 2016; Oliveira et al., 2019)
- Systems dynamic modelling (Dyson and Chang, 2005; Kollikkathara et al., 2010; Chen et al., 2012; Inghels and Dullaert, 2011).

2.6.5 Mathematical modelling methods to determine MSW generation

Mathematical models, similar to that proposed by Ojeda-Benitez et al. (2008) identify specific variables which are formulated from an empirical context:

$$\Upsilon = \alpha + \beta X_1 + \varepsilon \qquad (2.1)$$

where Υ is the daily per capita quantity of residential MSW generated**,** α is an intercept and indicates the mean of the response variable when equal to zero**.** β represents the average change**,** ε is the error term and X_1 is the variable matrix for the following: (i) X_{EDU} = the average education per household, (ii) X_{HAB} = the number of residents per household and (iii) X_{INC} = the income per household.

Rufford (1984) analysed household refuse composition to derive an empirically based model formulated from MSW sampling:

$$W_{ijk} = \sum_a (W_a)_{ijk} \cdot n_a \qquad (2.2)$$

where W_{ijk} is the quantity of waste generated in a given area in a region i, season j and point in trend k, $(W_a)_{ijk}$ is the waste generation coefficient associated with households of type a in a given area in a region i, season j and point in trend k.

To utilise these relationships (or similar empirically based models) whether as equations and/or matrices both formulations depend upon data, either retrospective for Ojeda-Benitez et al. or sampled MSW data for the Rufford model with each, equally difficult to acquire (Joosten et al., 1999).

2.6.6 Fuzzy grey dynamic modelling and neural networks.

Fuzzy grey dynamic modelling and artificial neural network modelling are highly specialised approaches. However, the results obtained from each are presented.

Chen and Chang (1999) predicted MSW generation for Tainan in Taiwan using fuzzy grey dynamic modelling with an error range - 9.2% to 1.2% of the output. Orsoni and Karadimas (2006) present a model but no output data or validation.

Antanasijević et al., (2013) propose and compare 2 neural network models, identifiable as BP and GRNN, designed primarily for developing economies where data is missing or absent. The model's output represents data at the national level. Input parameters are gross domestic product [GDP] and domestic material consumption [DMC] normalised as per capita with GDP also including the EU (27) average. Output (municipal waste generation) is kg/capita. Figure 2.4 presents output data for Bulgaria and Serbia using the GRNN model. The relative errors obtained for Bulgaria and Serbia from the GRNN model were generally below 10% with the exception of Bulgaria, 2006 which was 17%, when compared to reported output. A value below 10% was the author's assumed success rate. However, the relative errors obtained for 2005 for 9 countries, ranging from Greece 51.09%, Slovakia 48.83%, Latvia 43.33%, Romania 32.62%, Sweden 31.33% to Norway 16.36% were explained by issues with input data. Generally, for 2003 – 2005 for the 27 countries and 2 groups of EU nations modelled the relative errors were below the 10% success value. The authors identify that for

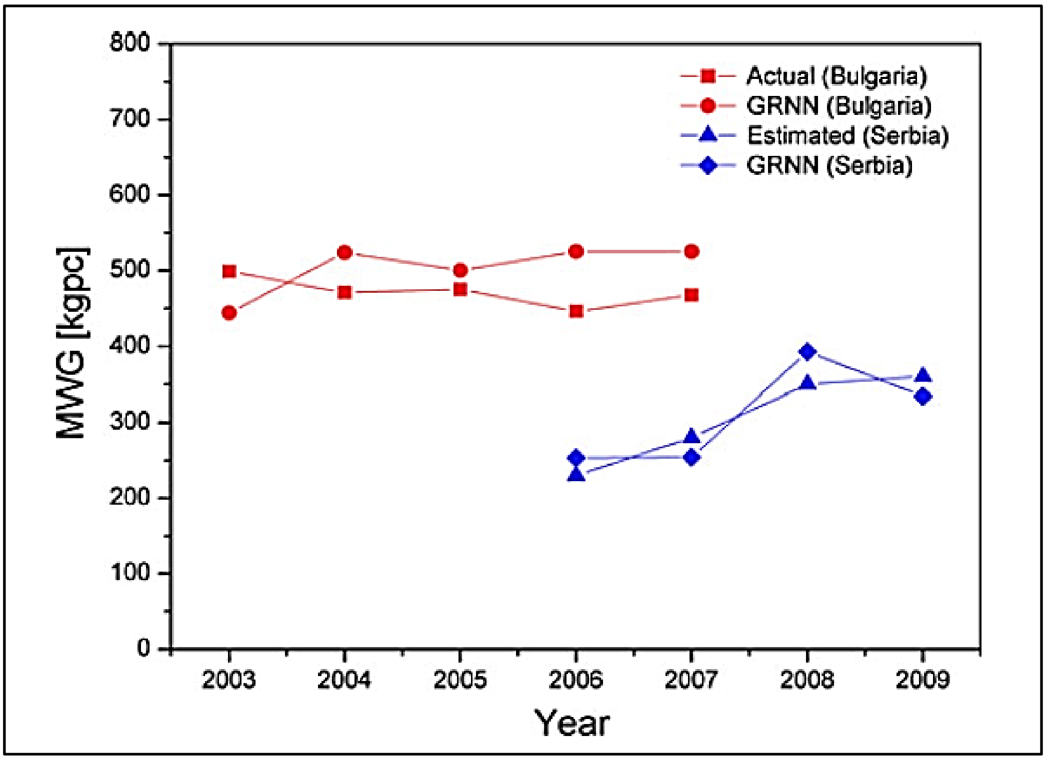


Figure 2.4: Actual and predicted waste generation quantities for Bulgaria and Serbia using the GRNN model. Figure reproduced from Antanasijević et al. (2013 p.45).

Serbia only estimated data was available, but the relative errors were small which they suggest further enhances the GRNN's functionality. Whilst designed for emerging economies their situation reflects that of the UK pre-1995.

Abbasi and El Hanandeh (2016) compared 4 models in a case study to simulate the monthly MSW generated in the Logan City Council area. Logan is located in Queensland, Australia. The models were identified as a support vector machine [SVM], an artificial neural network [ANN], an adaptive neuro-fuzzy inference system [ANFIS] and k-nearest neighbours [kNN]. The authors report that, for the training stage R^2 values exceed 0.8 but for the test simulations there were noticeable differences between the R^2 values with the ANFIS model (R^2 = 0.98) outperforming the others and showing good agreement with actual data supplied by the city authority. The ANN model produced the worst results (R^2 = 0.46). The ANFIS model was able to produce the most accurate monthly forecast the kNN model was better suited to producing average values of waste generated.

Oliveira et al. (2019) developed an artificial neural network to predict the quantity of packaging waste where the models variables are related to population education levels, the size and degree of urbanization of a community and a range of background socio-economic factors. The authors propose their neural network model achieves an R^2 value equal to 0.98. The better performing multiple non-linear regression models, using the same input data, achieved R^2 values of 0.65. With input data outliers removed this increased to 0.73.

Generally, MSW generation ANN models can be classified as long, medium and short-term (Oliveira et al., 2019). Waste generation ANNs are reliant upon data as inputs whether obtained from socio-economic and demographic variables or specific MSW related variables. As such, as with any modelling approach, each model's time-step whether monthly, annual or longer is reliant on those inputs reflecting changes that may occur either seasonally or to some other time-based influence. Results published by Oliveira et al. (2019) demonstrate seasonality in the generation of packaging waste however, the time-step included for Antanasijević et al. (2013) is annual with no detail provided for assessment of longer-term fluctuations other than to assume there is an increasing trend. This study rationalises significant changes to occurring within 4 defined epochs.

2.6.7 System dynamics modelling

This literature search located 15 system dynamics publications designed to resolve dynamical problems associated with a range of waste problems. These publications are presented in Table 2.2. Four models from Table 2.2 are relevant to this study (Dyson and Chang, 2005; Kollikkathara et al., 2010; Chen et al., 2012; Inghels and Dullaert, 2011). Dyson and Chang (2005) proposed 5 individual systems dynamics models for the city of San Antonio, Texas to represent waste flows into 4 recycle facilities to estimate each facility's recycling output.

Table 2.1: Overview of peer reviewed systems dynamics publications.

Reference	Modelling application	Geographic region
Thirmurthy (1992)	Investment in waste services	Madras, India
Mashayekhi (1993)[1]	Transition from landfilling	New York
Sudhir et al. (1997)[1]	Waste planning	Developing economies
Karavezyris et al. (2002)[2]	Development of a waste management system[3]	Berlin, Germany
Kum et al. (2004)[4]	Recovery through composting and informal recycling	Chey Landfill, Phnom Penh
Dyson and Chang (2005)[5]	Forecasting MSW generation[6]	San Antonio, Texas
Chaerul et al. (2008)[5]	Hospital waste management	Jakarta, Indonesia
Gönenç et al. (2008)[7]	Transition from landfilling	Holland
Kollikkathara et al. (2010)[5]	MSW forecasting	Newark, New Jersey
Inghels and Dullaert (2011)[8]	MSW management policy	Flanders, Belgium
Chen et al. (2012)[9]	MSW generation/landfill capacity	Singapore
Westbrook et al. (2012)[4]	Comparison: waste to landfill, waste to energy or waste biofuel	California
Dace et al. (2014)[10]	Packaging waste management	Latvia
Golroudbary and Zahraee (2015)[11]	Modelling recycling	Seremban, Malaysia
Fan et al. (2018)[10]	Modelling computer recycling	Taiwan
Pinha and Sagawa (2020)	Waste management and financial analysis	Brazil

Notes: 1) Systems Dynamics Review. 2) Mathematics and Computers in Simulation. 3) Includes fuzzy modelling. 4) Systems Dynamics Conference. 5) Waste Management. 6) Includes fuzzy modelling. 7) Computational and Mathematical Organization Theory 8) WMR. 9) The Macrotheme Review. 10) Resources, Conservation and Recycling. 11) Simulation Modelling Practice and Theory. 12) Journal of Cleaner Production.

Each model simulated a different input scenario: (i) Model 1, income received from each facility, (ii) Model 2, people per household, (iii) Model 3, historical amount of waste generated, (iv) Model 4, income per household and (v) Model 5 general population level. For verification, each model is compared to a base case obtained from an established regression method using historical US census data for population and income data. The modelling objective was to better plan the siting of future facilities. The authors identify 5 complex issues that are necessary to include within the basic model structure: (i) the relationship between income/wealth and waste generation, (ii) the relationship between income/wealth and increased recycling commitment (iii) current environmental and regulatory obligations (iv) the manner in which variables interact with one and other and (v) the dynamical element. The authors report a disparity in overall outturn of waste generated for 2010 equating to 46,260 tonnes per annum against a population increasing from 786,023 in 1980 to 1,144,646 in 2000 and conclude the structure of Model 1 provides the best representation for planning similar facilities in the future. This publication provides a working demonstration of the performance of systems dynamics where the supply of data is poor. One point to note is that the authors accept the regression model as reflecting an accurate assessment. However, there is no ‘test’ result included for the regression neither is the methodology identified.

Kollikkathara et al. (2010) propose a system dynamics model based on LCA_IWM (Beigl et al. 2003) as applied to Newark, New Jersey for the period 2003 - 2013. The modelled variables include the total waste output for the period, per capita waste output and the available landfill space both with and without waste prevention measures. The modelled waste comprises paper and card, organics, metals, plastic and hazardous waste with initial data supplied by the waste authority. For the 10-year simulation period the model generated weights for the individual materials, available landfill capacity based upon the estimated waste and financial data in respect of the cost advantage obtained from recycling. The authors conclude that using data, “*real and plausible . . . . . and statistical estimations of past behaviour*” the model can be applied to similar applications as it presents *“a practical and realistic picture*” however, they accept that further development is necessary particularly the introduction of additional sub-components (Kollikkathara et al., 2010, p. 2202). As a general rule, systems dynamics models are constructed as a series of sub-components. This study's model is constructed in the same manner and would form elements within an integrated waste system whether estimating historical or future outturn.

Chen et al. (2012) was published in the Macrotheme Review which appears from its website to no longer publish. There are mistakes within the model construction which also contains a number of unnecessary variables. The equations driving the model need further qualification in particular how the 'look-ups' are formulated, whether they result from specific formulae or have been generated from the graphing function and if so, on what basis. Total waste generated is equated to Total waste disposed + Total waste recycle. It would be better to sum Total waste generated from Industrial waste generation + Domestic waste generation. Whilst there are many issues of a similar nature the basic concept, however, is good in that the overall model incorporates a series of sub-models that reflect waste generation and management that result from Singapore's population growth and developing and expanding economy.

Inghels and Dullaert (2011) is a waste management and policy model however it is included because it has a retrospective element in that the researchers collected historical data ranging back to 1991 along with interviewing and reviewing literature to augment missing data. This study has followed a similar blueprint.

2.7 The evolution of chemical processes within landfills

Employing the search methodology identified in Section 2.3, seventy-nine journal papers alongside sixteen hard-copy publications were selected for review. The range of publications is large and exhibit the ubiquitous biochemical phases occurring during the decomposition of MSW from the onset of what are basic fermentation reactions resulting from the decomposition of carbohydrate and further developed in Chapter 6, through to techniques to increase reaction kinetics by use of better operational practice and the development of specific bacteria to increase the breakdown of more recalcitrant carbon materials present in waste streams (Bevan, 1967, Rees, 1980; Stegmann, 1983; Bugg et al., 2011a; Bugg et al., 2011b; Muaaz-Us-Salam et al., 2020). Whilst MSW is heterogeneous, materials disposed into landfills are subject to similar physical, chemical and biological processes (Rees, 1980; Ehrig, 1983; Hoeks, 1983; Stegmann, 1983; Christensen and Kjeldsen, 1989; Belevi and Baccini, 1989; Kromann and Christensen, 1998; Bozkurt et al., 2000; White et al., 2004; Vavilin et al., 2006; Haarstrick and Völkerding, 2007; Barlaz et al., 2010; De la Cruz and Barlaz, 2010; White and Beavan, 2013; Reinhart and Stegmann, 2019; Cossu et al. 2019). Christensen and Kjeldsen (1989) proposed a five-phase (idealized) degradation sequence for materials disposed into landfill: i) a short aerobic phase which, in most cases, is measurable in days, ii) phase 2, the acetogenic (or sometimes, fermentative) phase is characterised for the production of weak organic acids and carbon dioxide, iii) in phase 3 methanogenic bacteria become dominant and the production of methane commences as the pH increases and the concentration of sulphates decrease, iv) phase 4 establishes a period of stable methane generation and v) the 5th phase results from the gradual penetration of air into the surface layers of the landfill. These phases can be thought of as the 'classical' understanding which has continued to develop in particular where regulation has resulted in changes to established decomposition processes (Warwick et al., 2018). This research is developed and presented as Chapter 6 of this thesis.

Historically, the composition of materials within landfills was thought to be 58% inorganic and 42% organic (Jones and Owen, 1932 and reported in Bevan, 1967). It was also known chemical and biological processes progressed decomposition of materials within landfills resulting in the generation of gaseous emissions, heat and water. Bevan (1967) attributes the birth of science-based investigations to Jones and Owen at Manchester in 1932. The detail of their work although published is unavailable with their (summarised) results reproduced by Bevan. Bevan himself states they could not be (generally) obtained. The micro-biology and chemistry of organic degradation, as it occurs in landfills, is clearly identified and can be found referenced to others in more recent publications, no doubt due, and understandably so, to the unavailability of their findings. The paucity of research until the 1960s is confirmed by Campbell (2011) who cites the work Jones and Owen along with that of Eliassen et al. (1957) as the only research particular to landfill gas [LFG] release and its generation as a result of decomposing waste. The then current view, now accepted as false, is exemplified by Paragraph 179 of the 1971 report of the Working Party on Refuse Disposal proposes that filling over laid landfill should be delayed allowing LFG to escape.

Landfills are now labelled and modelled as bioreactors with organic fractions decomposing both aerobically and anaerobically (Rees, 1980; Ehrig, 1983; Hoeks, 1983; Stegmann, 1983; Christensen and Kjeldsen, 1989; Belevi and Baccini, 1989; Kromann and Christensen, 1998; Bozkurt et al., 2000; White et al., 2004; Vavilin et al., 2006; Haarstrick and Völkerding, 2007; Barlaz et al., 2010; De la Cruz and Barlaz, 2010; White and Beavan, 2013; Reinhart and Stegmann, 2019; Cossu et al. 2019). Modern landfill design allows for the inclusion of specific features to enhance organic decomposition as this is the dominant process and determines the chemical environment (Reinhart et al., 2002; Barlaz and Reinhart, 2004; Christensen et al., 2011; Bolyard and Reinhart, 2016; Rashid et al., 2017). Landfills considered in this study are conventional bioreactors that is, it is assumed such sites are without leachate recirculation or flushing technologies as these are recent developments and their employment by site operators suggests suitable importation records will have been maintained.

A review of research by Stegmann (1983) proposed changes in established site operating practices as routes to increasing landfill reaction kinetics resulting in faster methane output over a shorter generation period. The review examined experimental results from laboratory scale testing, the use of lysimeters and field trials in Austria, Germany and the US/Switzerland, respectively. Field work undertaken and reported by Rees (1980) and Ehrig (1983) identified the existence of different bio-chemical environments. Christensen and Kjeldsen (1989) proposed a five-phase (idealized) degradation sequence with no time units other than the first of the phases, the short aerobic phase which, in most cases, is measurable in days. Phases 2 – 4 are anaerobic with phases 2 and 3, forming the 1st and 2nd intermedial phases. Resulting from laboratory experiments theoretical inference led to an eight-phase model. This extension to the original 5-phase decomposition model occurs very slowly taking decades or even centuries (Belevi & Baccini, 1989; Christensen et al., 1996; Manfredi & Christensen, 2009).

Cossu et al. (2019) provide a series of generalized biocatalyzed reactions with greater detail of those occurring during the acetogenic phase. Three series of linear reactions: hydrolysis, acidogenesis fermentation and acetogenesis fermentation create suitable conditions for the development of methanogenesis. Reinhart and Stegmann (2011) identify physiochemical factors influencing decomposition. These include waste settlement, temperature, waste permeability, pH, material content – although the authors accept this list is not exhaustive. These factors impact upon and can both enhance or inhibit biochemical decomposition processes that exist within differing and dynamic micro-environments found within an MSW landfill matrix. The MSW matrix is heterogeneous, porous and complex (Barlaz et al., 1990; Komilis et al., 1999; Bozkurt et al., 2000). Numerous models represent different physical processes however, these cannot be treated in a holistic way due to the matrix's inherent complexity (Muaaz-Us-Salam et al., 2019; Reinhart and Stegmann, 2019).

References to the heterogeneity of MSW are commonplace. Regulation controlling MSW into landfills is included under paragraph 17 of the Landfill Directive (1999/31/EC) "whereas the measures taken to reduce the landfill of biodegradable waste should also aim to encourage the separate collection of biodegradable waste, sorting in general, recovery and recycling" (European Union 1999, P.L 182/2). Biodegradable wastes defined by the Directive are rapidly degrading wastes and include kitchen and garden residues. Evidence of slow degradability or to some extent the non-biodegradability of certain organic wastes has been demonstrated in landfill trial pitting of historic sites (Rathje and Murphy, 2001; Jennings et al., 2007). Such materials comprise organic polymers (plastics), treated wrappings and wood (Rathje and Murphy, 2001; Jones, 2008; Muaaz-Us-Salam et al., 2020). Evidence supporting the diversion of rapidly decomposing biodegradable wastes away from landfills could not be found. However, with the use of data sets obtained for this study it was demonstrated that the objective was being achieved (Warwick et al., 2018).

2.8 Discussion and conclusion

This chapter has reviewed four areas of research into the use of landfilling as a method for the disposal of waste. It is perhaps worth noting where these areas are expanded upon in subsequent chapters:

1. Landfill material content and potential. Chapter 5 proposes a dynamic model for the prediction of resources in landfills. To test the model's

validity, published data from landfill mining projects are used as a comparison to the model's output.

2. Methodologies for the retrospective identification of landfill content. The literature search identified 2 proposals and confirms the requirement for additional research where a series of input variables are presented and reviewed in Chapters 3 and 4 and applied to a system dynamics model in Chapter 5.
3. MSW generation modelling where proposals were considered for their potential to fill missing data which is considered in Chapter 5.
4. The evolution of chemical processes within landfills. This research is developed in Chapter 6.

As such, a significant number of publications have been reviewed and a large quantity of data collected. For this reason, Chapters 3, 4, 5 and 6 build on the reviews included here. The legacy of waste to landfill and the developing political dynamic regarding the location and ownership of material resources has increased attention on the resource potential contained in landfills. The removal of waste from landfills by landfill mining, it was proposed, would not only serve to remediate the developing environmental issues but would source supplies for energy generation and reclaim lost materials along with being economically sustainable. In principle this seemed logical however, the costs and uncertainty resulting from individual pilot projects are negative and reinforce the view that LFM projects are high-risk, in no small part, due to the uncertainty surrounding a landfill's content. However, pilot projects did create opportunities for invasive investigation whether this was pre-LFM site investigation or significant open-cutting of the landfill mass.

LFM is not a new endeavour however, an enhanced concept where greater material valorisation offers improvement to the financial return is proposed. LFM remains a high-risk operation due to: (i) the lack of recording of the wastes landfills content and (ii) the high degree of heterogeneity found in municipal waste streams. Any proposal for LFM will require significant pre-project investigation to mitigate these risks (Jennings et al., 2007; Frändegård et al., 2013; Rosendal et al., 2017). One alternative is to model the content of landfills.

Limited research has been undertaken to propose models that identify the resource content in landfills. Two publications proposed different approaches (i) a mathematical method (Lyons et al., 2010) and (ii) a data model based on historical site records and decomposition modelling (Wolfsberger et al., 2015a). Part of this study's aim is to contribute to that research. However, whilst each approach is peer reviewed each model contains limitations that impact upon either its accuracy or usefulness. The mathematical approach is further considered in respect of waste generation. Fundamentally, whilst the approach captures facets of MSW generation, the heterogeneity of waste and the factors driving that heterogeneity are numerous, complex and are never completely modelled (Rufford, 1984; Parfitt et al., 1994; Parfitt and Flowerdew, 1997; Beigl et al., 2007). In addition, many models rely on data to populate their parameters, this data is simply not available.

To overcome the issues associated with complex variables and data, 3 computer-based solutions are reviewed. The use of: (i) 'fuzzy' mathematics substitute for data, (ii) artificial neural networks and (iii) systems dynamics. The published results for each approach indicate any of the approaches should be beneficial in the solution of the landfill content problem.

Decomposing MSW creates a series of identifiable biogeochemical environments which create harmful gaseous and liquid emissions. The gaseous phase is a function of the breakdown of organic compounds existing within waste streams and develops from an intensive reactor phase early in an MSW landfills evolution. After one to two decades gaseous emissions reduce to become negligible by comparison. The liquid emissions continue for a period only established by modelling and this is proposed to exceed one thousand years (Rees, 1980; Ehrig, 1983; Hoeks, 1983; Stegmann, 1983; Christensen T. and Kjeldsen P., 1989; Belevi and Baccini, 1989). Active research is underway to concentrate the so-called gas phase by increasing the decay rates of recalcitrant organic materials. Regulation regarding waste imports and improvements to landfill operations (Reinhart et al., 2002; Barlaz and Reinhart, 2004; Christensen et al., 2011; Bolyard and Reinhart, 2016; Rashid et al., 2017) have provided protection for the shorter-term and evolved the way landfills decompose (Warwick et al., 2018). However, the long-term legacy of landfills as pollution hotspots remains.

Chapter 3: MSW generation and disposal – history, management and evaluation

3.1 Introduction

The literature search and review uncovered numerous references to the paucity of data also comment as to its veracity. General opinion of MSW data is exemplified by Professor Chris Coggins' (2014) witness statement given to an enquiry led by Queen Mary University:

> "*In those days nearly all waste - and, again, back in the 1980s you're talking about probably about 85 per cent, 90 per cent of waste, household waste - went to landfill. Landfill was cheap, it was widely available, there was no real control . . .. [Professor Coggins continues by explaining why the waste data is of poor quality]. We were collecting whatever data that came out. I illustrated the fact that local authorities often weighed their dustbin one week in a year and multiplied it by 52. . . there was an organization called, it still exists, the Chartered Institute of Public Finance and Accountancy and they did surveys of local authorities for waste. We became involved with them back in the early 1980s, and I have a very complete record of what they were doing, and it was evident that, if you looked at the data, it was primarily based on estimation.*"

Unfortunately, Professor Coggins died in February 2017. Having discussed these points with Tina Benfield, FCIWM, Chartered Waste Manager, CEnv, CChem, MRSC, a former professional associate of Professor Coggins the data referred to was that published by the Chartered Institute of Public Finance and Accountancy [CIPFA]. His testimony was based on his experience as an academic and advisor to CIPFA. However, this study found no formal analysis has been undertaken in respect of MSW data. This is confirmed in a PhD study, Watanabe (2003). The Watanabe study reviews CIPFA recycling data and comments on waste sampling completed by the Department of the Environment, Food and Rural Affairs and reports by the UK Audit Commission.

The data collected by this study is extensive and requires a more thorough evaluation before either accepting or contradicting the generally accepted sentiment. As such, it is necessary to present a considerable quantity of data therefore, this Chapter, together with Chapter 4 presents and evaluates 3 different elements of contemporaneous MSW data and considers its use as potential model inputs. The approach is to consider historical data's effectiveness as a series of representative samples able to establish a functional sequence of waste disposal and generation activity so as to replicate the dynamics occurring over the sixty-year period to 2007. As a route to achieving this, Chapter 3 commences with a proposal for evaluating MSW data before reviewing disposal and generation data. The chapter is structured as follows:

- Section 3.3 identifies and considers why there are issues with MSW data:
    - Background to the problem with MSW data.
    - Why data is necessary to model landfills.
    - MSW data sources
    - Data management
- Section 3.4 proposes a methodology for evaluation and considers:
    - Sampling and bias
    - Sample size
    - Representative data
- Section 3.5 presents and reviews MSW disposal and generation data and considers:
    - MSW disposal strategies
    - MSW generation
- Section 3.6 discusses the data and presents a series of outcomes arising from using actual and estimated MSW generation data and considers:
    - Congruous and incongruous MSW generation data
    - Consistency of MSW generation data
    - Recorded MSW compared to calculated
    - Reviewing the differences

How these data fit into the proposed model is presented in Figure 3.1. Each data set presented below is imperfect and exists from the completion of voluntary surveys. These were not sampling exercises and were not designed for that purpose. However, consideration is given to the criteria that constitute an acceptable, sampling framework. In particular the number of elements necessary to form a sample and what is required for a sample to be representative of its population.

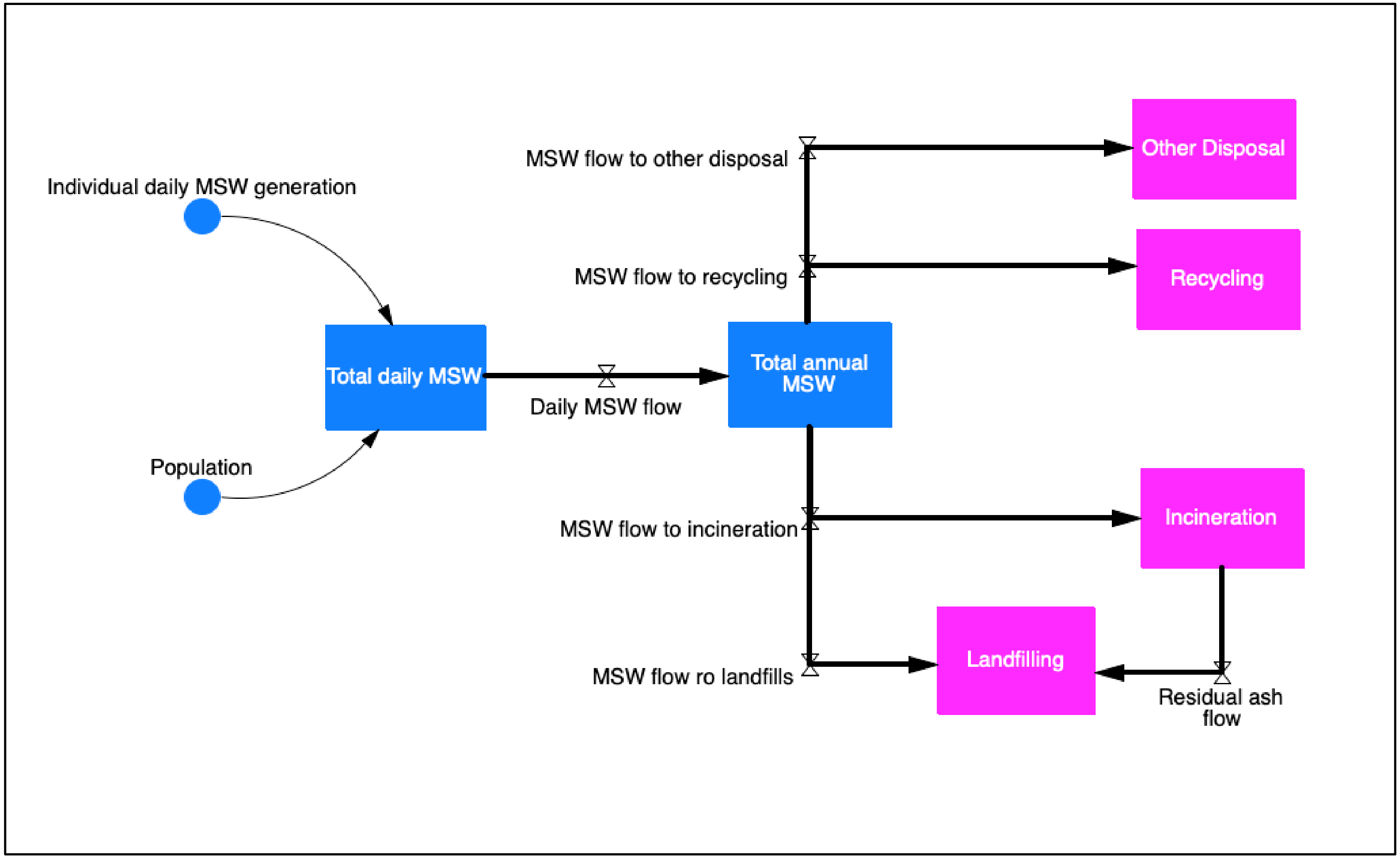


Figure 3.1: Abridged system dynamics sketch of the waste generation and disposal sub-units of the proposed model. By convention, boxes represent stocks with blue boxes stocks of material flowing to adopted disposal strategies, the pink boxes, which can be considered as stocks of value materials. Attached to a stock is a flow, a broad black arrow. Variables (in this study) are represented by circles which can be connected to stocks or flows by connectors (narrow black arrows). Arrow heads identify the direction of flow and identify where a variable has an impact. Other Disposal includes composting and pig swill collections.

3.2 Relevance to the aims and the objectives

This chapter evaluates MSW generation and disposal data as a contribution to answering this project's first and second research questions in respect of the suitability of historical and geographic data in predicting landfill content. As such, it proposes to demonstrate these data are suitable in providing parameters and surrogate data as the basis for a predictive model as encompassed by objectives three and four. These objectives entail the development of a model to determine materials contained in landfills.

3.3 MSW data: background, necessity, sources and management

3.3.1 Background to the problem with MSW data

From the late 1950s MSW started to change in composition and volume (Higginson, 1960; Stirrup, 1965; Higginson, 1966; Bevan, 1967; Flintoff and Millard, 1969; Wilson, 1981; Bridgewater, 1986). Higginson (1966) sought to provide sufficient data through MSW sampling so as to allow local authorities to formulate best decisions for MSW disposal. Ferguson (2015), when describing the mid-1960s, attested to the essential need for reliable composition data. This personal reflection resulted from the London boroughs becoming increasingly aware of a developing land shortage for landfilling and began looking to transport MSW out of the metropolis.

MSW data has been criticized for being incomplete and subject to not only over-estimation, possibly deliberately so (Ministry of Housing and Local Government, 1956: Note 6, 1961 Note 7, 1967 Note 7; Coggins and Brown, 1995) but also under-estimation (Cooper, 1996). There is general agreement that waste data before 2007 has a number of issues (Wilson, 1981; Rufford, 1985; Parfitt et al., 1994; Parfitt and Flowerdew, 1997; Parfitt et al., 2001; House of Commons Environment, Food and Rural Affairs Committee, 2005; Parfitt and Bridgewater, 2009; Stokes et al., 2013; Vinogradova et al., 2013; Jones and Tansey, 2014; Coggins, 2016). Whilst the paucity of MSW data is accepted, the major criticism stems from the fact that many local authorities estimated MSW weights from only a small number of samples quite literally 1 – 5% of generated MSW with some authorities sample weighing only a small number of waste-bins. Clearly, this could lead to questions regarding any parameter or surrogate value derived from such data as this may bias their value. A preliminary examination of respective data sets indicates that this would be the case. However, on average, at least 130 English local authorities weighed their waste and, for the purposes of this study, any local authority weighing 50% or more of generated MSW has been included within the analysis. The range 50% - 100% has been selected with 50% as the lower bound as this provided the widest sample frame of those selected by the Ministry of Housing Working Party on Refuse Collection (1967), with 80% - 100% defining the upper level. Using 80% tends to bias a sample towards the London Boroughs as most of the London authorities transported their waste which necessitated knowing its weight.

3.3.2 Why data is necessary to model landfills

At the basic level (identified as Tier 1 in this study), what is required to estimate the quantity of materials disposed into a landfill are 4 variables: (i) the number of people in a settlement or collection of settlements, (ii) the rate at which MSW is generated. Daily, weekly or whatever time period is chosen. (iii) the percentage of MSW disposed into landfill for that settlement or settlements or, alternatively, knowledge of the disposal strategies adopted by the waste authority and iv) how do other disposal strategies impact on others. To move to the next level (Tier 2) so as to model the material content of the landfill mass then it is necessary: i) to be able to identify the components of MSW, ii) estimate how different components degrade and iii) determine the quantity of landfill cover material, was it imported or site-won? Furthermore, and pertinent to each tier, are data specific to a particular landfill or landfills or is a general, non-specific landfill being modelled. This becomes a complex modelling process as additional variables are included. Whilst the end use is in respect of landfills and not a new antibiotic, nevertheless there is a requirement for rigour in the analysis. “Data quality and data quantity are important” or as this quotation is frequently applied, “rubbish in, rubbish out”.

MSW data form the foundation of this study and what follows is to understand what is meant by good, bad and, from a practical perspective, useful data. Defining good or bad data my appear obvious with bad data not meeting the criteria set for good. What of useful data? Is the data functional in that it describes a situation or provides indicators to better understand? Good quality data is provided by well-designed sampling methods which take into account the data end use (Sapsford and Judd, 2006; Cox and Donnelly, 2011). However, as has been identified much of the data did not result from sampling experiments although in respect to the many MSW composition studies reviewed (Chapter 4) these were planned, well executed and, over time, developed into scientific studies. Section 3.4 proposes a methodology for the evaluation of MSW data, Section 3.5 presents and reviews MSW disposal and generation data, and its application is discussed in Section 3.6.

3.3.3 MSW data sources

This study has identified a number of potential data sources however, much of the historical quantitative data, excluding MSW composition survey data, originate from the Ministry of Housing and Local Government publications or those published by the Chartered Institute of Public Finance and Accountancy [CIPFA]. These sources refer directly to the collection and disposal of household and commercial wastes by local authorities and individual data entered for each local authority should be considered as primary sources. A schedule of these sources is included as Appendix 3.1 with MSW composition sources listed in Appendix 3.2. Some data has been used by the publishers to produce estimates, where these are included as part of this study, they are identified. The Ministry of Housing and Local Government Public Cleansing Returns existed before World War II but were suspended at its outbreak. However, following the war, publication resumed in 1952/53. Investigation suggests publications in this format ended with 1966/67. Additional collection and disposal data for the mid/late 1960s are provided by 2 UK government working parties. The 1967 report for the Ministry of Housing and the 1971 report for the Department of the Environment with each initiating 2 surveys however, response to both remained voluntary.

MSW data, generally, for the years between 1968 and 1978 is very limited. A further report from the Department of the Environment (1978) in association with the Society of County Treasurers and County Surveyors' Society provides limited, averaged data partly filling the void left by the Ministry of Housing publications. The societies published separately with some minor differences in the data. From this point data is provided by CIPFA. CIPFA was formed from the amalgamation of the County Treasurer's and County Surveyors' societies. The publication of

MSW data remained limited until the mid-1990s. Despite the issues identified with CIPFA publications they remained the most comprehensive source (Coggins, 1995). Annual waste data was collected from the mid-1990s by the Department of the Environment, Transport and Regions (1995/96) and (following) the Department of Environment, Food and Rural Affairs (1996/97 – 2002/03). The response to these waste surveys, whilst not obligatory, improved markedly and achieved 97 - 99% response rates. Unlike the CIPFA data, which presents data for individual authorities, data was aggregated and presented for England, Wales, the regions and for each category of local authority.

3.3.4 Data management

The basic formatting of published data in the UK follows local government structure (Table 3.1). Every English local authority [LA] belongs to a specific sub-group of authorities: county borough, borough, metropolitan borough, district etc. However, over the study period, 3 major changes to the structure were instituted.

Table 3.1: English local authority structure 1888 to the current time.

Authority Class	1888 - 1974	1974 - 1997	1997 - 2021
Super Tier	GLC from 1965	GLC until 1986	London
First Tier	County County Borough	County (either metropolitan or non-metropolitan)	County (administrative)
Joint			Unitary Authority Metropolitan Borough
Second Tier	Metropolitan Borough Municipal Borough Urban District Rural District	District Metropolitan Borough	District
Third Tier[1]	Civil Parish	Civil Parish	Civil Parish

Note: 1) A Civil Parish's authority is limited to small, local functions only (usually allotments footpaths, village greens and the like).

To manage the data, initially LAs were separated into these sub-groups, that is by local authority category and the (financial) year the waste was generated and disposed. This allows the data to be aggregated nationally (for England as a whole), at the regional level, as county councils or the larger non-metropolitan authorities and treated as individual authorities responsible for specific settlements. The intention is that where data does not exist for individual LAs then suitable surrogate data can be used in model simulations. Further separation into two categories follows: i) authorities weighing 50% or more of generated MSW and ii) all reporting authorities. An abridged graphic identifying the MSW data management process is included as Figure 3.2.

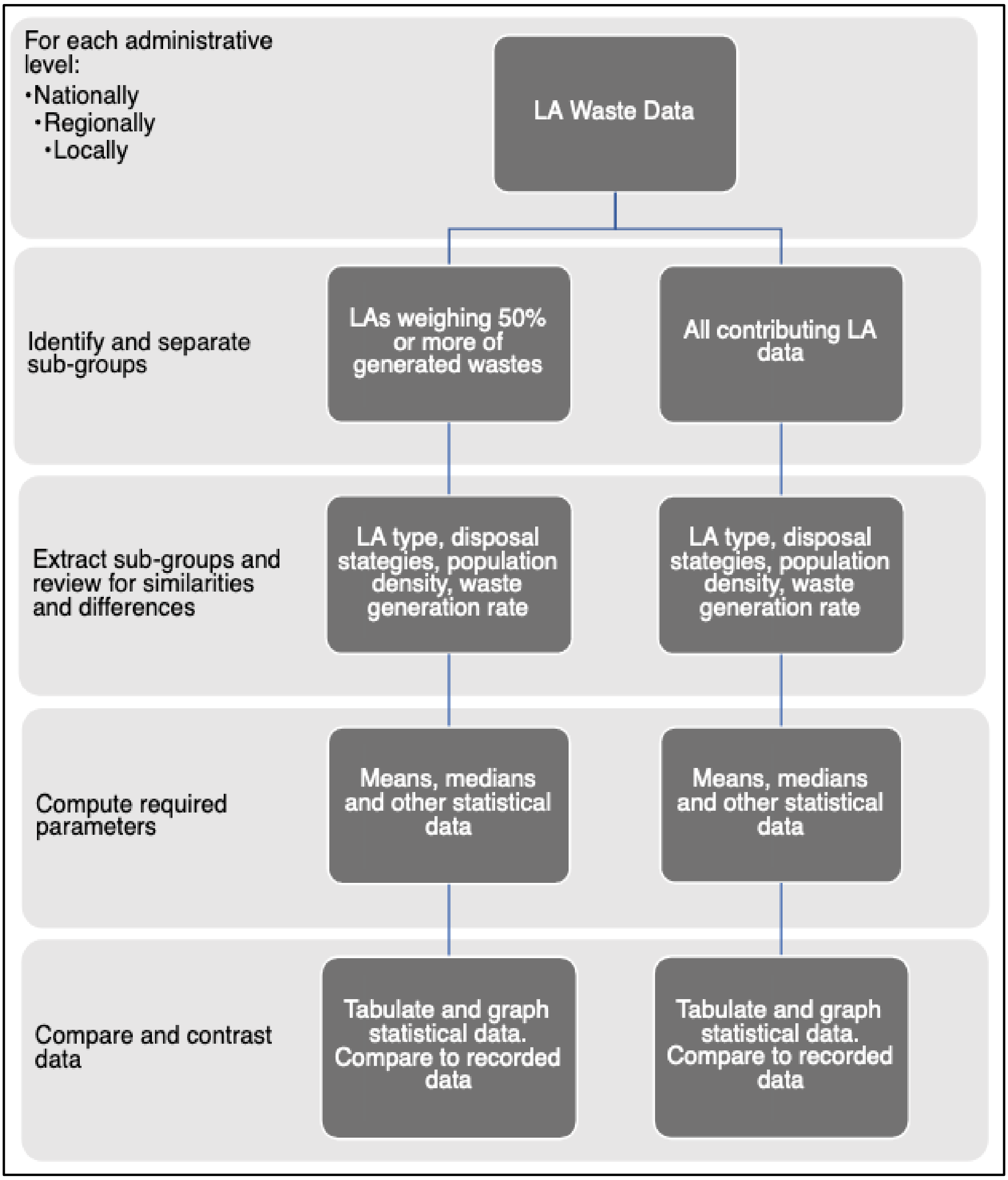


Figure 3.2: Abridged flow chart identifying the stages of the data handling process.

From the managed data, Minitab provided descriptive statistical functions (mean, median, standard deviation, range, skewness, etc) for a series of daily MSW generation rates. Minitab statistical formulae are defined and presented in Appendix 3.3.

3.4 Data evaluation

3.4.1 Sampling and bias

A common issue and concern in statistics is bias. Bias can be introduced into a data set as a result of the way the data is sampled either in the design of the sampling framework or the methods of data collection. Poor sampling design can impact on parameters determined from data sets and adversely influence any estimator derived from a particular data set (Poll, 1988; Gy, 1992; Piegorsch and Bailer, 2005; Cox and Donnelly, 2011; Webster and Lark, 2013). Researchers attempt to eliminate (or at least reduce to a negligible level) sampling bias. Gy (1992) proposes effective sampling bias can be suppressed with the adoption of three basic sampling criteria:

1. The sample group should be of sufficient size to reflect the characteristics of the sample frame.
2. A sample is required to be representative and reflect the characteristic of the sampling frame being studied.
3. Elements of the sample set should be chosen randomly.

The size of a sample is considered in the next section. Cox and Donnelly (2011) state that where data is collected for comparative analysis point two may not be so critical and is dependent upon the context of the final study. The context of this study is the material content of municipal landfills. Point three is somewhat immaterial as the data are historical.

3.4.2 Sample size

Data reported to the Ministry of Housing and Local Government include approximately four hundred local authorities from one-thousand five hundred English authorities in existence during the period. Three key respondent variables from three randomly selected years are presented as Table 3.2: (i) population represented, (ii) waste generated and (iii) local authority disposal option.

Table 3.2: Summary of data for all English LAs making household and trade waste collection and disposal returns to the Ministry of Housing and Local Government.

	1954/55[1]	1959/60[2]	1965/66[3]
Population (England)	41,941,000	43,175,000	45,071,400
Number of English LAs making return	371	447	407
Population represented	28,445,272	31,422,807	32,005,100
Total waste generated (tonnes)	9,016,107	10,235,952	10,520,870
Total waste to landfill (tonnes)	7,170,454	8,573,542	9,177,406
Other disposal routes (tonnes)	1,845,653	1,662,410	1,343,464

Notes: Data from: Ministry of Housing and Local Government 1) 1956. 2) 1961. 3) 1967

From some four hundred local authorities reporting annually, a sample of approximately 27% arises. Is this likely to be considered a satisfactory sample? If the three years are considered against the population they represent compared to the national population of England at that time, the sample size increases to a range of 68% - 73% representation. A similar range for waste generation occurs, based on 14 million tonnes generated annually in England.

Sample size in respect of numbers of LAs can be calculated. Where the standard deviation can be evaluated (or estimated) a sample size is determined by the desired confidence level. Webster and Lark (2013) calculate sample size using:

$$N = (zs)^2 / L^2 \qquad (3.1)$$

where N is the required sample size, s is the standard deviation, z is the is the standard normal deviation for 95% interval = 1.96 and L is the tolerance around the estimated mean value. One of the variables used in this study is the weight of waste generated each day by a single person. Calculating this variable using the same data set used to compile the 1959/60 data for all local authorities presented in Table 3.3, the standard deviation calculated by Minitab for this variable equates to 0.32. The tolerance (L) about the mean, is usually selected at an acceptable level, in most cases between 5 – 10% and determined by expected sampling costs. A lower tolerance increasing the number of samples.

In this case 10% is taken. The sample mean, again calculated by Minitab, equates to 0.924. Using equation 3.1 the required sample size N equates to approximately 46 where N for this data set is 447.

Data sets for generated wastes over the course of six decades have been collected and reviewed by this study. The above example uses, by comparison, a large standard deviation. Taking data for metropolitan districts for the financial year 1984/85 (Appendix 3.4) the mean for daily waste generation is 0.793 kg per person with a standard deviation equating to 0.109. Using these in equation 3.1, the required sample size reduces to 7 authorities. Standard deviations for waste generation in the following tables, with the exception of Inner London generally lie in the range of 0.100 to 0.200. This study treats the Cities of London and Westminster independently following the formation of the GLC. Each has a significant work-day population increase which impacts on respective waste generation rates. For the City of London, the resident population varies between 4,800 – 5,200 with its work-day population increasing this number to more than 300,000. However, from the asymmetric perspective, it provides an indication of the quantities of waste generated by a busy commercial centre.

3.4.3 Representative data

An adequate sample in respect of size does not itself determine whether that sample is representative of the population (Piegorsch and Bailer, 2005). Data reported by LAs are composed from a series of subgroups (or strata) determined by LA category. A representative sample should reflect the character of the study population which entails elements from each subgroup. Further questions arise in respect of both the complete data set and individual strata. First, the criteria used to determine representation, population size or local authority category or numbers. Second, the factors that determine a subset's representativeness of the overall set. Tables 3.3 and 3.4 identify how representative the data are for each sub-group by i) comparing the respective populations of those responding local authorities to those having weighed 50% or more of generated MSW and ii) highlighting the differences within sub-groups. This point may be considered distracting as subgroups composed of larger urban centres (Table 3.3) contain more people elements than smaller urban or rural centres however, their population densities may be similar. In addition, what characteristics of waste generation are particular to each subgroup, is each subgroup homogeneous? What of the inter subgroup characteristics? Does each subgroup contain elements that reflect a series of basic characteristics?

Table 3.3: Numbers of responding English local authorities to the Ministry of Housing 1954/55, 1959/60, 1965/66 and the Department of the Environment (1971) and the populations each category of authority represent. Mid 1960s data present numbers of local authority category and the populations they represent.

	Local Authority	County boroughs	London councils	Boroughs	Urban districts	Rural Districts
1954/55[1]	Number	76	83	124	88	-
	Population	12,591,630	7,875,531	5,225,691	2,752,420	-
1959/60[1]	Number	76	86	125	103	57
	Population	12,486,170	7,008,100	5,425,927	3,471,820	1,971,720
1965/66[1]	Number	76	28	123	119	61
	Population	12,634,200	7,008,100	5,644,000	4,225,100	2,493,700
Mid[2] 1960s	Number	78	GLC[3]	233	453	409
	Population	13,227,190	7,913,600	6,977,100	6,977,100	9,308,600

Notes: 1) Ministry of Housing and Local Government Public Cleansing. Refuse collection and disposal, Street cleansing costing returns. 2) Data taken from Department of the Environment. 1971. 3) The GLC was created in 1965 and became responsible for waste collection and disposal.

Table 3.4: As Table 3.3 but only includes those authorities weighing 50% or more of generated waste.

	Local Authority	County boroughs	London councils	Boroughs	Urban districts	Rural Districts
1954/55	Number	22	59	38	6	-
	Population	4,846,060	5,896,631	1,795,231	252,900	-
1959/60	Number	19	65	30	9	-
	Population	3,452,690	6,312,440	1,514,710	322,970	-
1965/66	Number	20	24	30	8	2
	Population	4,015,400	5,975,399	1,511,000	301,700	95,100

Notes: Population data taken from Department of the Environment. 1971. All other data as Table 3.3. Mid 1960s data is same in Tables 3.3 and 3.4. and is included for comparison.

Table 3.3 compares local authority subgroups taken from all reporting authorities to the situation occurring in England during the mid-1960s. Authorities well represented in terms of participation and inhabitant population are the county boroughs, the London councils and non-county boroughs with those larger authorities appearing, from inspection of the data, to respond to the Ministry of Housing request for data. For comparison, those authorities weighing 50% or more of waste generated are presented in Table 3.4. The London councils retain a high proportion of authorities weighing generated waste however the county boroughs and boroughs reduce significantly to a quarter of the number presented in Table 3.4 with urban and rural authorities poorly represented if at all. The concentration of data is more pronounced across sub-groups with larger population centres as a result of excluding authorities weighing less than 50% of waste generated. However, this has less impact if it can be demonstrated that an acceptable degree of homogeneity exists across or within the sub-groups for the desired key variables, namely disposal options and waste generation such they can be applied generally.

3.5 MSW disposal and generation data

3.5.1 MSW disposal strategies

One approach to establishing local authority waste disposal strategies is to accept the often-quoted figure that 90% of municipal wastes were disposed into landfills. This then implies 10% of MSW was disposed into alternative strategies. For 1987/88 10% represents some 2.5 million tonnes of MSW. A review of published disposal data suggests that while this may satisfy the general case for England it would lead to imprecision if applied across the period 1945 – 2007 and to inaccuracies when considering the resource content of individual landfills. A further complication arises when considering individual landfills, where changes implemented by local authorities, impact on material inputs into landfills either as pre-treatments or salvage.

To identify how this impacts on waste disposal, Table 3.5 presents data for local authorities reporting to the Ministry of Housing and Local Government. MSW reported as being disposed to alternative strategies, 15.8% for 1954/55 and 12.8% for 1959/60 (reported other disposal as % of generated), exceed estimates based upon the generally accepted percentage disposed to landfill that is around 90%. Furthermore, alternative disposal strategies were more common before 1965/66 which suggests pre-1954/55 figures may have exceeded 16%.

Table 3.5: Summary of data for all English LAs reporting MSW to the Ministry of Housing and Local Government. Estimated total MSW: 1954/55 from JC Wylie, 1959/60 figure taken from Ministry of Housing and Local Government (1967), 1965/66 figure based on DofE (1971). Estimated disposal weights based on DofE (1971) percentages: landfill 90.4%, other 9.6%.

	1954/55	1959/60	1965/66
Generated MSW (estimated - tonnes)	11,700,000	13,000,000	14,200,000
90.4% MSW direct to landfill (tonnes)	10,580,000	11,750,000	12,840,000
9.6% MSW to other disposal (tonnes)	1,120,000	1,250,000	1,360,000
Total waste reported (tonnes)	9,016,107	10,235,952	10,520,870
Actual reported to landfill (tonnes)	7,170,454	8,573,542	9,177,406
Actual reported to other disposal (tonnes)	1,845,653	1,662,410	1,343,464

The Ministry of Housing and Local Government data sets provide useful detail in respect of elected disposal strategy when identifying landfill imports. For example, individual local authority MSW flows to landfill can be illustrated from the 1963/64 data set where 65 local authorities incinerated some 900,000 tonnes of waste thus generating between 30 – 40% of that weight of incineration residues (Wilson, 1981). Incineration residues had a use as a construction aggregate and landfill cover (Bevan, 1967; Wilson, 1981) where Wilson identified 60% was utilised however data included below does not support this.

A similar situation existed in the 1970s. Table 3.6 presents a summary of disposal data published by the Department of the Environment (1978). Although reproduced directly from the report, the data identify a large increase in the quantity of MSW generated, the increasing use of private contractors and the re-emergence, during the 1970s, of alternative disposal strategies when 36 incineration plants were constructed and operated during that period (Wilson, 1981). The report identifies that MSW disposed by contractors were likely to have been landfilled but does not provide evidence to support this. Taking this to be a reasonable assumption then for the period covered by Table 3.6 landfilled MSW equates to 89 – 91% however the quantity of MSW incinerated exceeds 2 million tonnes annually, which is significant. Wilson's incineration data and data from the Department of the Environment provide sufficient detail to determine incinerator location and approximate quantification of incinerated MSW and residues arising for local authorities adopting this strategy.

Table 3.6: Weights of MSW disposed to major options 1974/75 to 1977/78. Data reproduced from Table 5, Department of the Environment (1978 p.8).

All weights in tonnes	1974/75	1975/76	1976/77	1977/78
Landfilled untreated	18,165,000	17,412,000	16,343,000	15,876,000
Direct incineration	1,613,000	1,940,000	1,935,000	2,084,000
Separation/incineration	418,000	346,000	294,000	209,000
Disposal by contractor	2,635,000	2,796,000	2,681,000	3,212,000
Total MSW generated	23,743,000	23,270,000	22,216,000	22,277,000

Local government reorganisation resulted in urban and rural local authorities responsible for waste collection and disposal being merged with borough councils to form non-metropolitan district councils. As such, each became a waste collection authority [WCA]. Responsibility for disposal passed to the forty-six English County Councils which became waste disposal authorities [WDA]. This reorganisation into a dual waste collection and disposal system was reflected in the publication by CIPFA of two data sets, MSW collected, and MSW disposed. However, this separation produced contradictory data between the two data sets noticeably in the quantities of MSW submitted by WCAs and waste received by WDAs from the WCAs. These contradictions have been reported previously (Stokes et al., 2013) and are discussed further in Section 3.6.3. In addition, the preface and notes accompanying each publication draw attention to inconsistencies in respect of double counting in the way waste was treated.

These inconsistencies lead to difficulty when quantifying weights attributable to the different methods of disposal. Table 3.7 presents data reproduced from CIPFA's summary tables for England. There appears to be a significant shortfall in the MSW disposed to landfills whilst there is a weight of waste not included in any disposal option. It must be assumed that transferred MSW was landfilled.

Table 3.7: Waste Summary data taken from CIPFA Waste Disposal Actuals: 1) total MSW disposed by English WDAs, 2) MSW disposed into landfill by English WDAs without further treatment, 3) MSW incinerated, 4) MSW undergoing pre-treatment with final disposal not identified, 5) materials reclaimed from MSW 6) other (unspecified) disposal routes.

	1979/80[1]	1984/85[2]	1986/87[3]
1. Total MSW for disposal (tonnes)	26,787,000	24,637,000	25,132,700
2. Disposed to landfill (tonnes)	19,344,000	15,620,000	16,076,100
3. Waste incinerated (tonnes)	2,526,000	2,371,000	1,705,100
4. Transferred waste (tonnes)	-	5,818,000	6,467,600
5. Reclamation (tonnes)	-	186,000	205,300
6. Other methods(tonnes)	212,000	622,000	1,147,800

Notes: 1) CIPFA included aggregated data for England in the published summaries. 2) Excludes Greater London and Merseyside. 3) Includes London authorities representing some 3.7 million people following abolition of the GLC but excludes South Yorkshire, Tyne and Wear, West Yorkshire, Wigan and 3 London WDAs.

CIPFA identify transferred MSW as including compacted crude waste, shredded and baled waste and civic amenity waste totalling 5.8 million tonnes for 1984/85 and 6.5 million tonnes for 1986/857.

Table 3.8 presents aggregated data compiled from individual WDAs annual data submission. These data include the final disposal methods following compaction, baling or shredding which, along with civic amenity wastes are disposed to landfill. In both presentations, neither data set maintains the 89 – 91% disposal rate to landfill. However, individual WDA data presented in Table 3.7, identifies a reduced percentage disposed to landfill, 1984/85 equates to 80.4% with 1986/87 further reducing to 72.5%. This shortfall is corrected if MSW disposed by external contractors is included. However, the data do not balance. More MSW was handled (rows 2,3 and 4) than disposed (row 1). One explanation is that some WDAs used contractors to dispose into WDA landfills and others used contractor's own sites. CIPFA suggest shortfalls arise through losses during pre-treatment, but this would require loses equating to some 4 – 5% of all wastes and not just those wastes undergoing pre-treatment. Further consideration is given to the quantities of MSW generated, hence disposed in Section 3.5.2.

Table 3.8: Aggregated individual WDA data taken from CIPFA Waste Disposal Actuals: 1) total MSW disposed by English WDAs includes pre-treated MSW and incineration residues, 2) MSW disposed into landfill by English WDAs including pre-treated MSW, 3) MSW incinerated, 4) contractor disposed MSW however no detail as to final disposal is provided, 5) incineration residues disposed to landfills.

	1979/80	1984/85	1986/87
1. Total MSW for disposal (tonnes)	26,984,855	23,817,059	25,132,775
2. Disposed to landfill (tonnes)	20,194,500	19,149,773	18,232,423
3. MSW incinerated (tonnes)	2,483,512	2,191,839	1,705,564
4. Contractor disposal (tonnes)	5,080,405	3,989,909	6,171,141
5. Incineration residues landfilled (tonnes)	869,229	749,522	622,385

Note: 1) CIPFA included aggregated data for England. Data extracted from the same sources included in Table 3.7.

However, these publications reflected a serious attempt by CIPFA to introduce greater detail into the annual waste surveys with the inclusion of source data together with contractor data, although this did not include specifics on how contractors disposed of MSW. Greater contractor employment by local authorities developed after 1984 along with the construction and increase in popularity of household waste amenity centres. For this study and at this stage it is assumed external contractors disposed of MSW to landfills as this would be the cheapest option. Landfill tax was not introduced until 1996 and the regulatory framework for landfills was limited. Incineration residues are included to give support to the generation rate quoted by Wilson (1981) and to identify that, for the most part, residues were landfilled as opposed to 60% assumed recycled.

The Department of the Environment, Transport and Regions [DETR] and (later) The Department of Environment, Food and Rural Affairs [DEFRA] undertook a series of surveys of local authorities in England and Wales commencing from 1995/96. The response rate for this first survey was some 90%. By the second and third reports this had increased to 415 authorities of the 416 contacted. Landfill accounted for 83.5% of the 20.9 million tonnes disposed in 1995/96 with 1.6 million tonnes (7.1%) recycled (Department of the Environment, Transport and Regions, 1997). By 1998/99 the quantity of MSW generated had increased to 23.76 million tonnes. Disposal statistics indicated a reduction in landfill disposal (82.4%) with MSW being diverted into recycling (9.5%) and incineration (7.5%) (Department for Environment, Food and Rural Affairs, 2000a). Reported statistics are quoted for England and regionally. Some regional variation in disposal methods exist over England exemplified by Merseyside landfilling 96% of waste generated. In contrast the West Midlands landfilled 71%.

3.5.2 MSW generation

Where the previous section considered data in respect of MSW disposal, this section introduces data that leads to the establishment of individual, daily MSW generation rates. MSW generation can be considered a key variable in the model developed in Chapter 5. For this study, MSW generation data is obtained directly from published data or is derived from annual, total MSW disposed divided by the population generating that waste which is the approach adopted for the Ministry of Housing data (1952 – 1966) and early publications from CIPFA (1978 – 2006).

Averaged MSW generation data for the period 1965/66 – 1977/78 are taken directly from two Department of the Environment publications following surveys of all local authorities in England and Wales. As in the previous section, data for Wales has been removed. For the 1971 report, Initial respondents represent almost 43 million people from a population taken from the 1966 census total for England of 45 million however, the data set contained a large number of unweighed estimates (identified by this study as All-LAs and All-data) for comparison with reporting authorities weighing 50% or more of generated MSW. A truncated record of these data is presented in Figure 3.3 with supporting data in Appendix 3.4.

3.6 Results and Discussion

3.6.1 Congruous and incongruous MSW generation data

The introduction to this chapter quoted Kazuhito Hasimoto in that data quality and quantity are important. Appendix 4 of Defra WR0119, A Review of Municipal Waste Component Analysis (2008) acknowledges that:

> “*Above all, ‘some data is better than none’: (i) even a misleading study of a ‘minor’ municipal waste stream is a better basis than assuming composition on a spurious basis (i.e., composition of the ‘minor’ stream assumed to be the same as overall waste composition); (ii) this project has demonstrated that outputs are evidenced based wherever possible*” (Resource Futures Appendix 4, 2008 p.13.)

This chapter has presented data to reflect Hasimoto’s perception whilst anticipating it conforms to Resource Future’s statement. The reviewed local authority disposal data readily demonstrate landfilling as the route for almost 90% of MSW until the mid-1990s. This is particularly the case for England until a sharp decline resulted in a radical shift in disposal policy following the 1999 Landfill Directive (Ministry of Housing, 1956, 1961, 1967; Department of the Environment, 1971; EUROSTAT, 2017). To achieve a more detailed picture, that is to be able to determine a particular local authority’s disposal strategy the combination of data provided in publications and documented evidence help provide the necessary detail so as to piece together respective landfill’s content.

In respect of waste data generally, paragraphs 665 and 667 of the Summary, Conclusions and Recommendations (Chapter 20) of the 1971 Department of the Environment Report underline the perplexing position of the 1970s:

> “*No comprehensive figures are available for trade, commercial and industrial waste, but the quantities (excluding power station ash and mining wastes) probably total more than 20 million tons (20.32 M tonnes) a year*.” (Department of the Environment, 1971 p.125.)

Given that it is proposed many waste generators, including local authorities, estimated, with varying degrees of uncertainty, the weight of waste generated, such a view is not surprising. Paragraph 667 continues:

> “*Indications of the changes in output and character of house refuse (excluding ‘bulky’ refuse and waste from business and commercial premises) were obtained by studying analyses made in authorities over a period of years. In 1968 the weight averaged 1.5 lb (0.68 kg) per head per day. Based on a forecast of the probable composition of house refuse in 1980 . . . . this is estimated to increase by then to 1.7 lb (0.77*

kg), an average yearly increase of about 1 per cent.” (Department of the Environment, 1971 p.125.)

The identified daily MSW generation rate of 0.68 kg per person appears low. Mid census population figures vary, but using that cited in the report, 45,374,090, multiplying by 365 days for a year the MSW generated equates to 11,261,850 tonnes. This does not compare favourably with the report compiled by the Working Party on Refuse Collection (1967) whose 1962/63 survey resulted in 13,835,553 tonnes (13,617,000 tons) or 0.85 kg per person per day. Tables 11 and 12 of the 1971 report (page 182) identify 12,788,249 tonnes (12,586,240 tons) as the quantity collected for disposal by non-mechanical means. Tables 14 and 15 identify an additional 1,499,925 tonnes (1,476,231 tons) collected and disposed by mechanical means. The sum of these quantities, 14.3 million tonnes, is comparable to the 1967 report and not the quantity cited in Paragraph 667 of the same report. Perhaps more significant is Paragraph 668:

“*Reliable figures obtained from a number of local authorities show that over a period of 4 years to 1969 the weight of . . [collected refuse] . . showed a similar increase and averaged 1.75 lb (0.79 kg) per head per day. In Greater London it was 1.95 lb (0.89 kg). During the next 10 years it is not expected that the weight of house and trade refuse per head as collected by local authorities will increase by more than 1 per cent per year, though there may be local variations.*” (Department of the Environment, 1971 p.126.)

MSW generated outside London is 10,801,732 tonnes and for Greater London, 2,570,733 tonnes resulting in a total of 13,372,465 tonnes. A 1976 Department of the Environment publication for 1974/75, identified collected household and commercial MSW of 17.14 million tonnes generated by a population of 46,436,000. That is an average generation rate of 1.01 kg per person per day.

Department of the Environment statisticians were concerned with authorities reporting greater than expected collected waste arisings. However, paragraph 3.1.2 comments on the data received and states:

With the exception of the GLC, Waste Disposal Authorities have made only sample weighings of the total disposed of. In the past it was felt *that WDAs that weighed a small proportion of their waste tended to overestimate the amount of waste per head, but this view was not supported by the statistical tests carried out on the data in this report* (Department of the Environment, 1978 p. 4)

To this point, the MSW generation rates identified in this section sit in isolation to the Ministry of Housing, CIPFA and DEFRA data. Figure 3.3 compares the per capita daily waste generation rates from 1954/55 – 2005/06. There is clearly discrepancy between authorities weighing 50% or more of MSW and those that do not with the former recording lower generation rates. The 1990s witnessed significant improvements in data reporting and in quantities weighed. Defra data reflects response rates ranging between 97 – 99% of all local authorities with all MSW weighed. However, the daily generation rate increases to 1.38 kg equating to some 26.3 million tonnes of generated household and commercial MSW.

The data set for 1974/75 provides two values for the mean with the lesser valued mean having that data set's outliers removed. The report does not define what constitutes an outlier but figure 3 of the Department's report identifies 5 in total (labelled A – D) and these appear to exclude those authorities whose reported data exceed 475 tonnes of waste per 1000 people annually (1.3 kg/person/day). The Department's data for 1968/69 includes only LAs weighing 80% or more of MSW generated. In each of the data sets presented in Figure 3.3, the increasing trend in the per capita generation rate is common to all authorities with the exception of 2005/06. The range for the 50%+ data local authorities are smaller compared to those for the All-Data authorities.

Further analysis of the data used to compile Figure 3.3 was undertaken using a series of boxplots to compare each local authority category. These boxplots are presented in Appendix 3.5 (a – d). Each series of boxplots compares MSW generation rates taken from local authorities submitting data where less than 50% of MSW was weighed (identified as All-Data) to those weighing 50% or more of generated MSW (identified as 50%+). For each series of 4 boxplots, the format follows contemporary local government structure. Within each series, Boxplot (1) compares London generation rates and (2) all authorities data taken together. For Appendix 3.5(a), boxplot (2) presents the county boroughs with (3) the boroughs. Urban and rural districts were omitted because the sample data for 50%+ is too

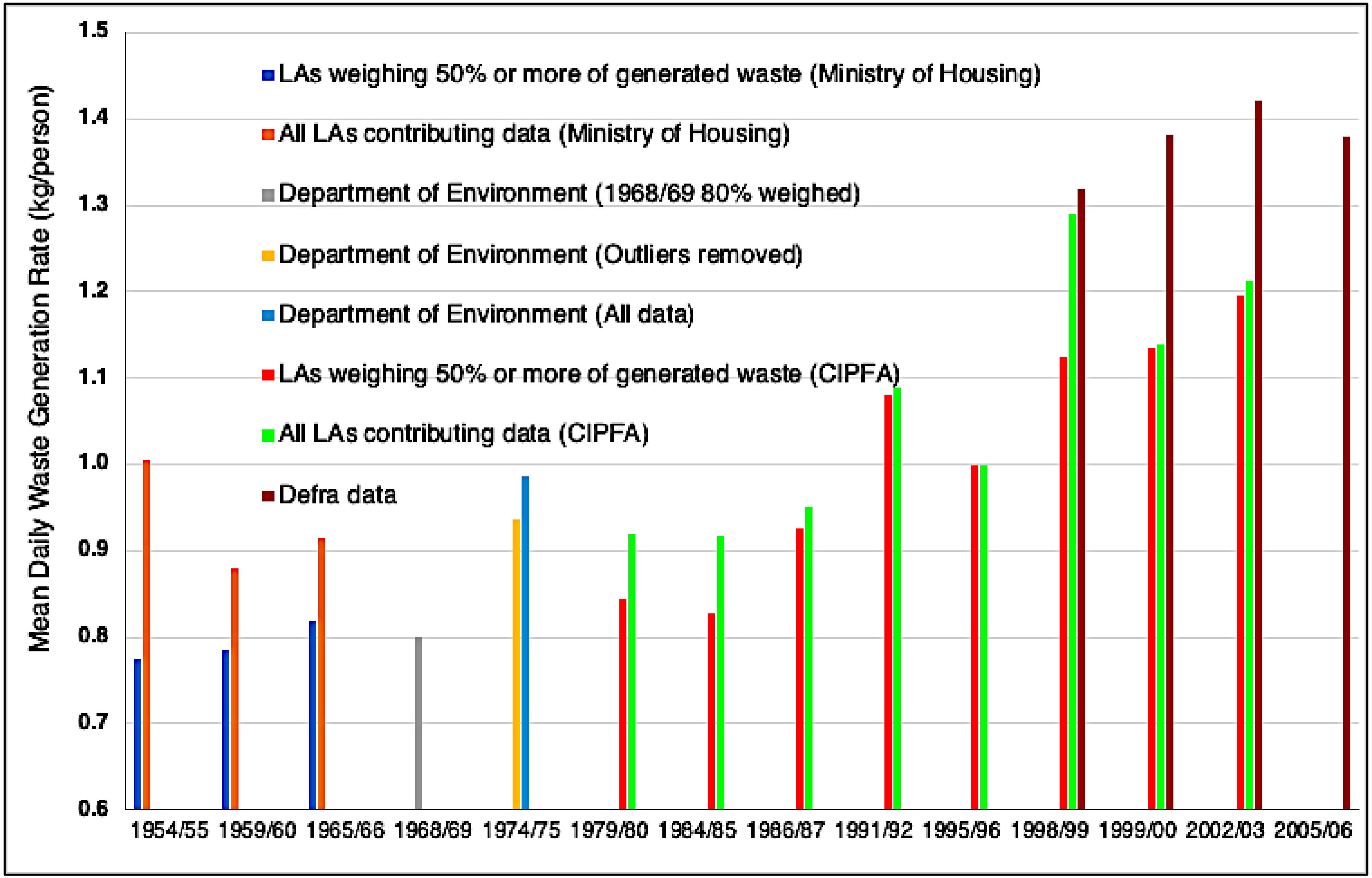


Figure 3.3: Per capita daily waste generation (kg) comparison between authorities weighing 50% or more of generated waste to all authorities contributing data

small. For Appendix 3.5 (b – d) (2) are metropolitan counties (latterly unitary authorities) and (3) non-metropolitan districts. Following the formation of the GLC in 1965 weighing of waste was near universal for London authorities hence Boxplots (1) present the All-Data as this does not differ from the 50%+ set. London data is split into Inner London and Outer London for these purposes because of the increased generation rate occurring within Inner London. In 1986/87 waste generated for the City of London was 54,300 tonnes. This equates to a generation rate of 30.99 kg/person/day (Westminster is 3.27 kg per person in 1986/87). The Cities of London and Westminster are excluded from Appendix 3.5 (b – d) as they skew the data.

Generally, the mean as a parameter is sensitive to outliers and the means arising from the All-Data values are influenced by these values. The data sets contain two low-value outliers with one included in the 50%+ weight sets. Although this value is treated by Minitab as an outlier it is the 1991/92 generation rate for Bradford. Bradford weighed 78% of its MSW producing 117,064 tonnes with a per capita generation rate of 0.68 kg/day and remains a credible value within that data set. There is a strong likelihood the number of outliers and the increased data range occurring with the All-Data sets arise from the estimation methodologies employed. However, many values from the All-Data sets equate to similar values in the 50%+ data sets. The age of the data prevents access to the detail of the test weighings undertaken or information to regarding employed estimation methodologies. Therefore, it is proposed to use only values lying within the 50%+ data sets unless these data are non-existent. With regard to the values equating to similar values in the 50%+ data sets these will provide a comparison, when surrogate estimators are required.

It is clear a spread of values exists for elements within data sets however, the range of values for each element is small and is further limited by employing (i) the 50%+ data sets and (ii) their further separation into 4 epochs. Furthermore, these data sets present original information and have no control data from which to make a comparison. The supporting descriptive statistics for Figure 3.3 (Appendix 3.4) identify the means, medians, standard deviations and coefficient of variation (a.k.a. the relative standard deviation). The means and medians for 37 of 40 data sets are very similar with the average difference amounting to ±0.04

kg per person per day which indicates a near to normal distribution for those data sets. For this study the mean values have been selected as estimators for daily MSW generation however, in respect of modelled MSW generation and disposal (Chapter 5) and for those years not identified in Figure 3.3 the ranges identified in the figure are similarly presented.

The sources for the MSW composition sampling data presented in Chapter 4 do not provide detail in respect of either sampling error or confidence limits. In addition, sampling undertaken by individual LAs is treated in this study as a series of snapshots that present a contemporary description of what was collected and then disposed as MSW. Section 1.5 of Chapter 1 distinguishes between specific and non-specific data where specific data is obtained directly from and refers to local authorities reporting that data. What is clear from the data is that MSW components vary in type and quantity. Use of the mean values from each data set or combination of sets to derive a single modelled value would be too limiting where non-specific data is the only option for England. Hence, the maximum and minimum values for each waste component are included to represent a range of possible outcome scenarios.

3.6.2 Consistency of MSW generation data

Section 3.6.1 has identified MSW generation rates to be generally increasing for each local authority category since the mid-1960s and confirmed by Figure 3.3 and the boxplot analyses (included as Appendix 3.5). However, gaps are present across these data in respect of missing years and LA categories. This section examines homogeneousness (discussed in Section 3.5.3) in respect of waste generation rates and considers their suitability as a replacement for missing MSW generation data.

Data substitution can be considered from two perspectives. First, as inter-relational connections that is, demonstrating one local authority category’s data set for MSW generation is similar to that of another. Second, as intra-relational connections that are able to establish a value or range of values for other set members. However, this is not straightforward as the use of a single parameter value may prove too restrictive. Factors connecting possible surrogates are subject to change. As such, this study restricts surrogates to those from the same epoch, whether inter or intra-relational connections. In respect of Inter-relational data sets, Figures 3.4(a) – 3.4(c) compare means for MSW generation for local authority categories for the periods 1954 – 1969, 1975 – 1992 and 1995 – 2006.

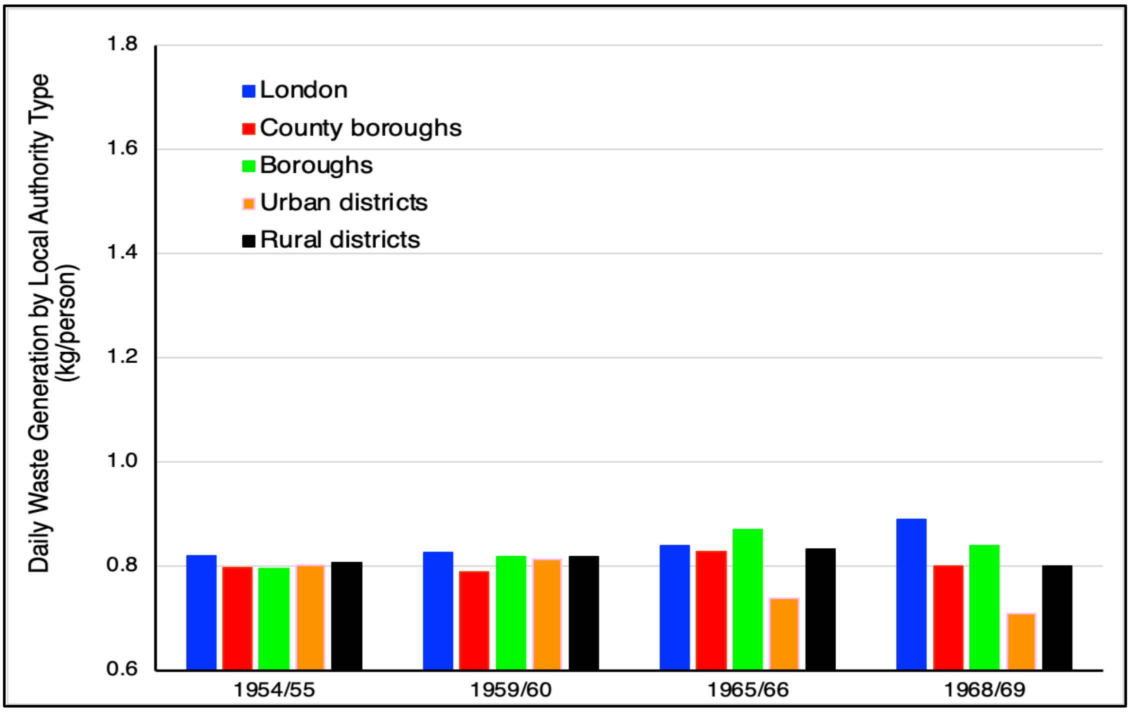


Figure 3.4(a): Means of MSW generation 1954/55 – 1968/69. Values for urban and rural authorities are taken from the All-Data sets with outliers excluded as 50%+ sets do not constitute a sample (std dev for urban = 0.26, N = 9. N for rural = 0).

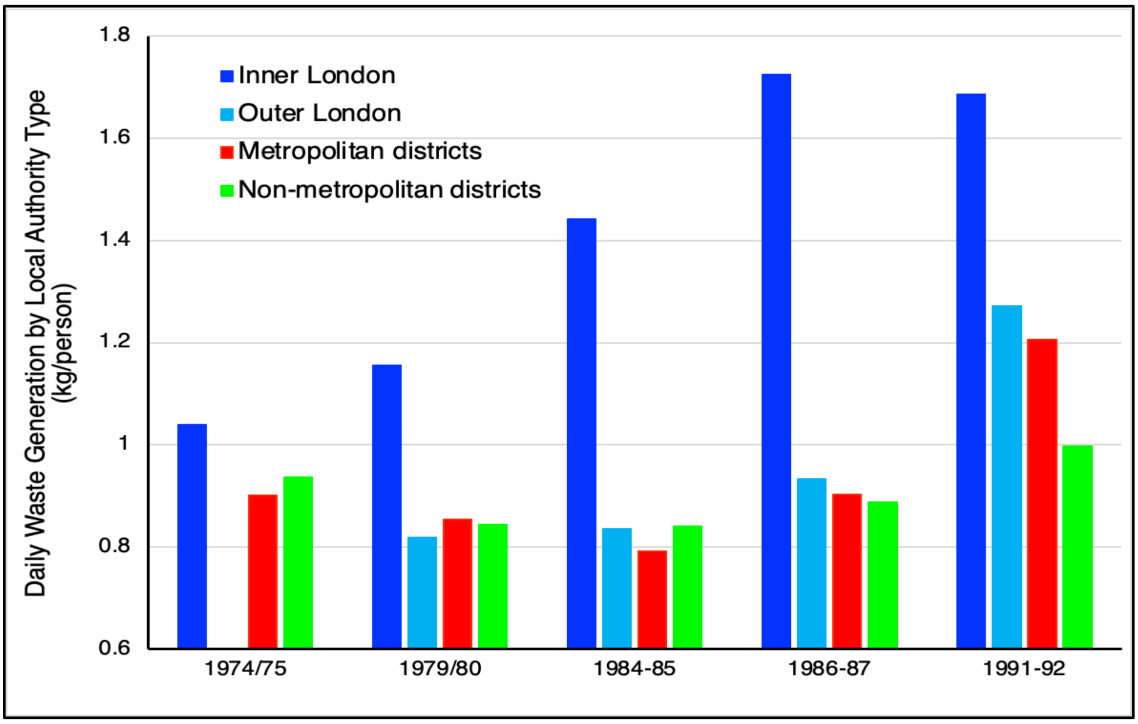


Figure 3.4(b): Daily MSW generation mean values by local authority category 1974/75 – 1991/92. All data taken from 50%+ data sets. MSW generation rates start to diverge from 1991/92

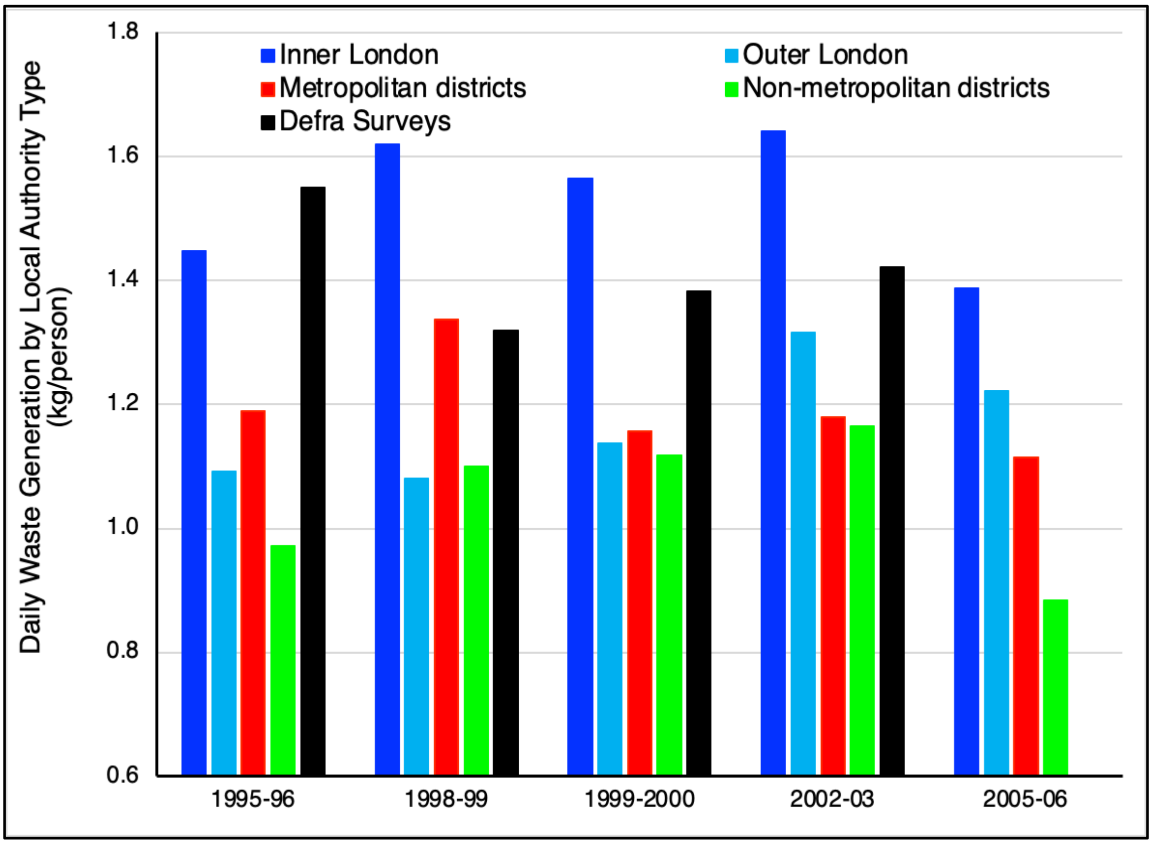


Figure 3.4(c): Daily MSW generation mean values by local authority category 1995/96 – 2005/06. CIPFA reporting format changed for 2005/06 and includes only household MSW.

For each of the data sets, outliers are excluded. In Figure 3.4(a) values for urban and rural districts are taken from the All-Data sets as the 50%+ sets do not constitute samples. London, from 1979/80 has an increased generation rate which continues through subsequent publications as Inner and Outer London with the former, from that point, treated as being unique due to the significant day-time population increase.

However, excluding urban district and the London values from 1965/66 and 1968/69 reduces these ranges to: 0.83 – 0.87 and 0.80 – 0.84 kg per person per day. Selecting the widest range which is 0.04 kg per person per day and taking 1968/69 as an example gives 0.82 ± 0.02 kg per person per day. This equates to 2.379 million tonnes of waste generated for urban districts for the lower of the values with 2.498 million tonnes for the higher. These are acceptable as surrogates because the waste generation rate for each local authority category is homogeneous as defined by their similar mean values. For later years most sets show pronounced differences that require surrogates to come from within the same sub-group. Missing data occurs where LAs did not complete surveys. To overcome missing data, it is proposed to classify missing MSW generation rates, within the same sub-group, as lying between the upper and lower limits of respective interquartile ranges. This will provide an upper and lower bound for total MSW and individual material components across the model. Using only the mean resolves every non-responding local authority, belonging to a particular sub-group, to generate waste at the same rate. A better solution would be to examine each part or non-submission within a sub-group for similarities (intra-relational connections) and assess its potential output based upon the known output of an authority with similar characteristics that submitted weighed data. However, identifying possible characteristics is an undertaking that necessitates access to further data beyond this project's scope. An approach employing population densities is summarized in the suggestions for further work.

3.6.3 Recorded MSW compared to calculated MSW

This section compares reported annual MSW generation for England to annual quantities arising from MSW generation rates calculated from 50%+ local authority data sets presented to this point and population data available from the UK Office for National Statistics. Table 3.9 identifies the reported quantities and sources for the annual MSW produced in England over the period 1954 – 2007 however, there are 41 years for which these totals are unavailable. The estimation methods are presented in Tables 3.10 and 3.11 together with their data sources. The calculated MSW quantities are presented as those for respective local authority categories based upon population levels for each local authority category and presented as a series of graphs (Figure 3.5 (a – c)).

Population data for individual local authorities are not available for much of the study period. As such, these missing data are required to be estimated. Furthermore, and historically, aggregated data for England and Wales were published as a single statistic therefore population levels for England require the deduction of those for Wales until 1974. B R Mitchell's British Historical Statistics includes 10-year census data to enable this. For the years between the census data then addition and deduction of births/deaths and migration statistics are necessary. A similar addition/deduction approach must be employed in respect of local authority category commencing from known populations.

Table 3.9: Recorded total household MSW quantities for England, 1954/55 – 2005/06 and their respective sources.

	MSW (M tonnes)	Source
1954/55	11.77	Quoted by JC Wylie (1957)
1959/60	14.41	Ministry of Housing Report (1967) but is the recorded value for 1961/62 as no figure for 1959/60 could be located
1965-1969	14.06 14.72	Department of the Environment (1971)
1974/75	14.69	Department of the Environment Appendix 4 (1978) adjusted for outliers.
1979/80	14.58	CIPFA collected waste 14.58 million tonnes
1984-1987	24.64 25.57	CIPFA Waste Disposal Statistics
1991/92	23.30	OECD Statistics: Municipal Waste, Generation and Treatment (2020)
1995/96	24.40	Department for Environment, Transport and Regions. 1997. Municipal Waste Management 1995/96
1999-2003	27.40 29.31	Department for Environment, Food and Rural Affairs. Municipal Waste Management Statistics 1999/200
2005/06	28.73	EUROSTAT 2016

Table 3.10: Waste generation rate and population estimation sources.

Data Range	Calculation Method	Data Source
1953 - 2007	To obtain the annual waste output for each local authority: $Local\ authority\ population \times daily\ waste\ generated \times 365$ For each local authority category calculate the mean and apply to those not meeting the 50% data reporting criterion. Use aggregated population data for each local authority category to establish total waste outputs for each then sum.	Ministry of Housing and Local Government (1952 – 1967) Department of the Environment. 1978. *English Local Authority Waste Disposal Statistics* (1974/75 to 1977/78) Chartered Institute of Public Finance and Accountancy Department for Environment, Transport and Regions (1997). Department for Environment, Food and Rural Affairs (1997 – 2005)

Table 3.11: Waste generation rate and population estimation methodology.

Required output	Date range	Method	Data source
Population estimates for Wales Population estimates for England	1950 – 1971 1950 - 2007	Identify population for England by deducting that for Wales. Establish annual population change for England.	Mitchell B. 2011. British Historical Statistics. https://gov.wales/sites/default/files/statistics-and-research/2018-12/980827-historical-1974-1996-chapter-1-en.pdf https://statswales.gov.wales/Catalogue/Population-and-Migration/Population/Estimates/nationallevelpopulationestimates-by-year-age-ukcountry
Commencing population levels for local authority category	1965/66 1974/75	Base years – data taken from publications.	*Refuse Storage and Collection. Report of the Working Party on Refuse Collection 1964-65.* London: Her Majesty's Stationary Office. Department of the Environment. 1978. *English Local Authority Waste Disposal Statistics 1974/75 to 1977/78*
Population levels for different local authority categories	1950 – 1964 1967 – 1973 1976 - 1990	Systematically amend each year using ONS births, deaths and migration data, I year at a time. For each completed year and before starting the next year check against (i) total annual change (ii) London population data which are available. Record the difference and if under 100,000 move the following or preceding year	https://www.ons.gov.uk/timeseriestool?topic=/peoplepopulationandcommunity/populationandmigration/populationestimates https://www.ons.gov.uk/search?q=migration+statistics https://www.ons.gov.uk/search?q=births+and+death+statistics
	1991 – 2007	Data available	

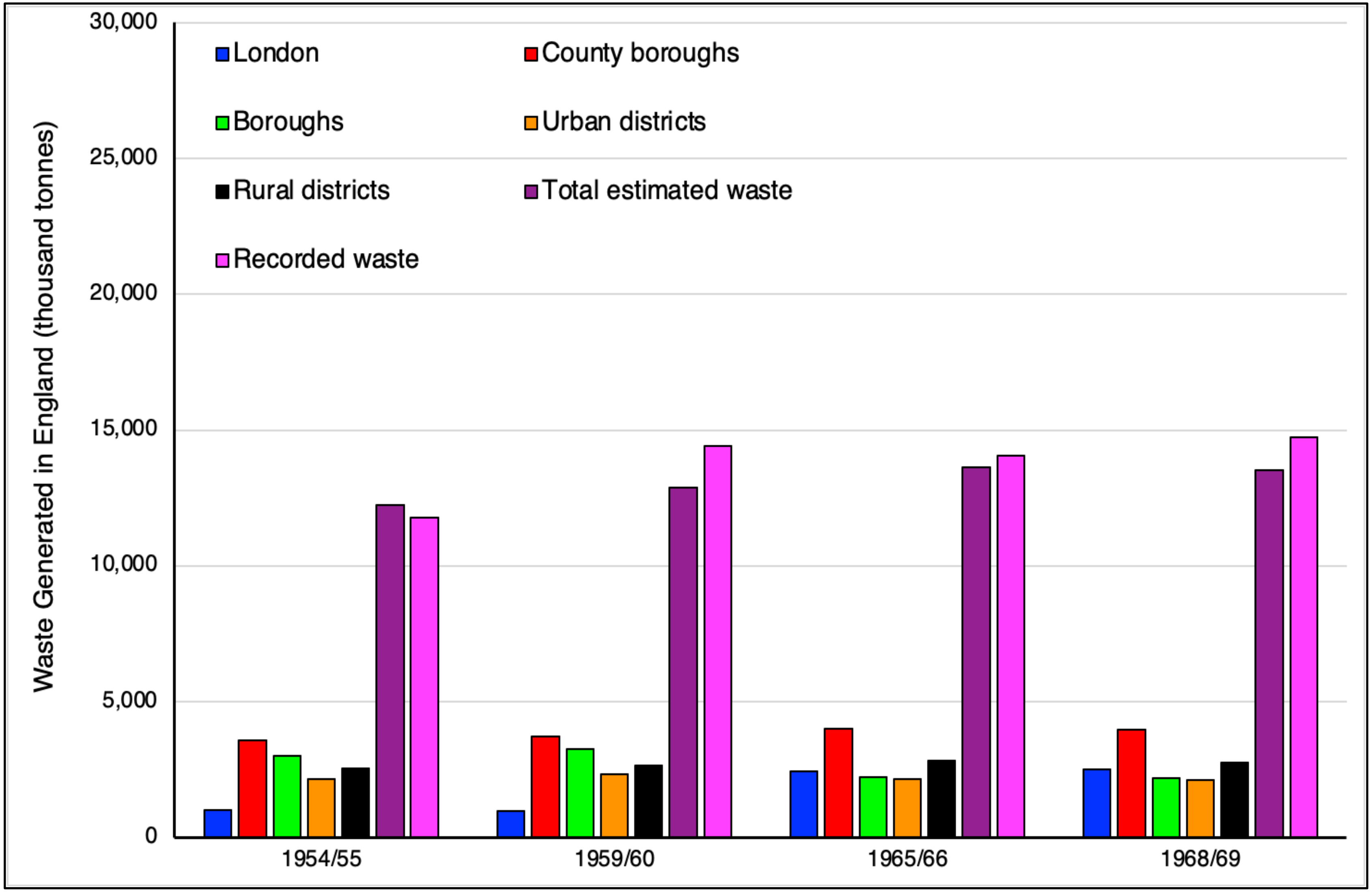


Figure 3.5(a): Estimated waste quantities using generated means for local authority categories against actual recorded waste 1954/55 – 1968/69

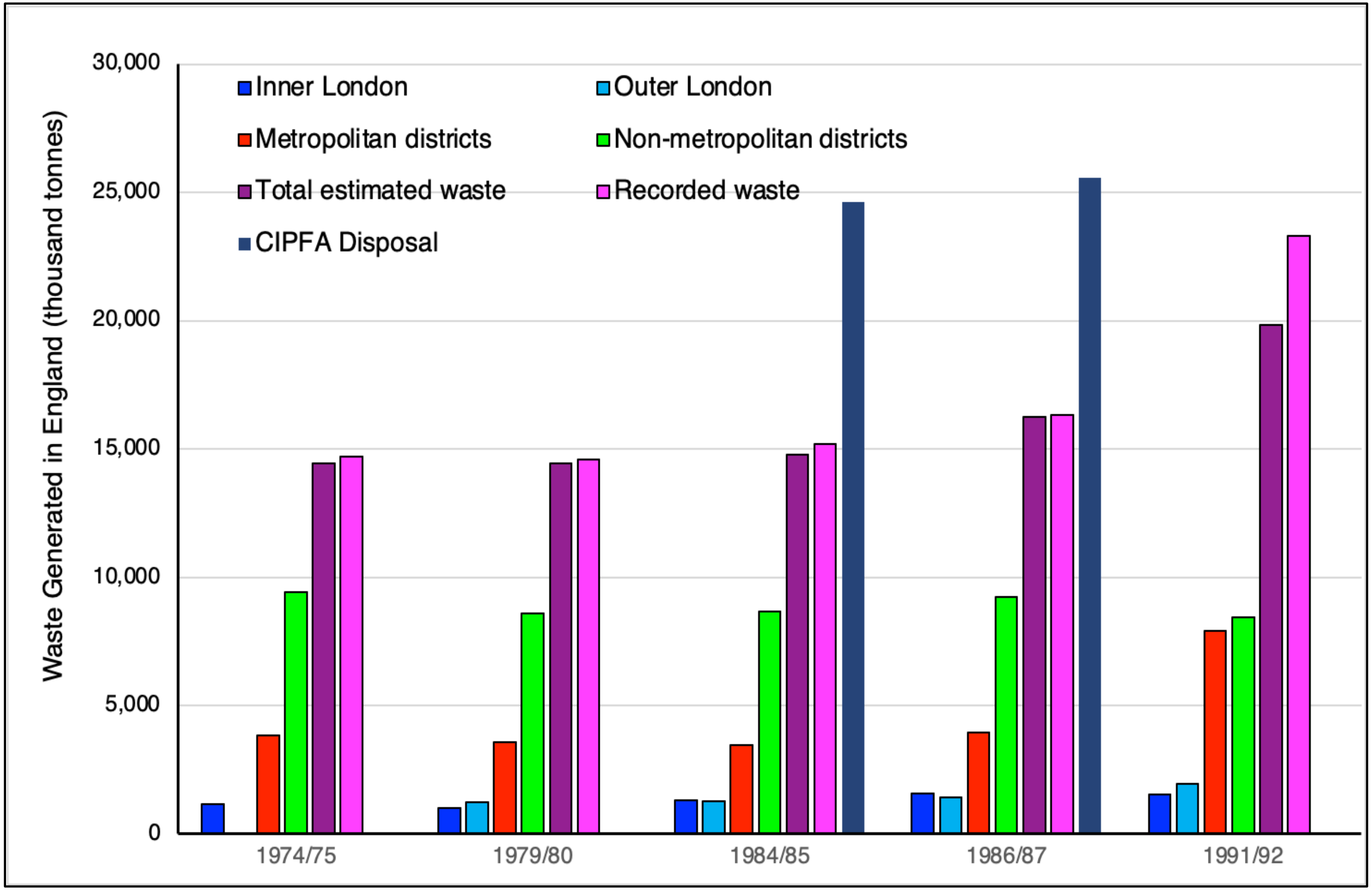


Figure 3.5(b): Estimated waste quantities for local authority categories against actual recorded waste 1974/75 – 1991/92. Total estimated waste is calculated using this study's generation rate. Recorded waste from DofE and CIPFA collection statistics (WCAs). CIPFA disposal reported by WDAs.

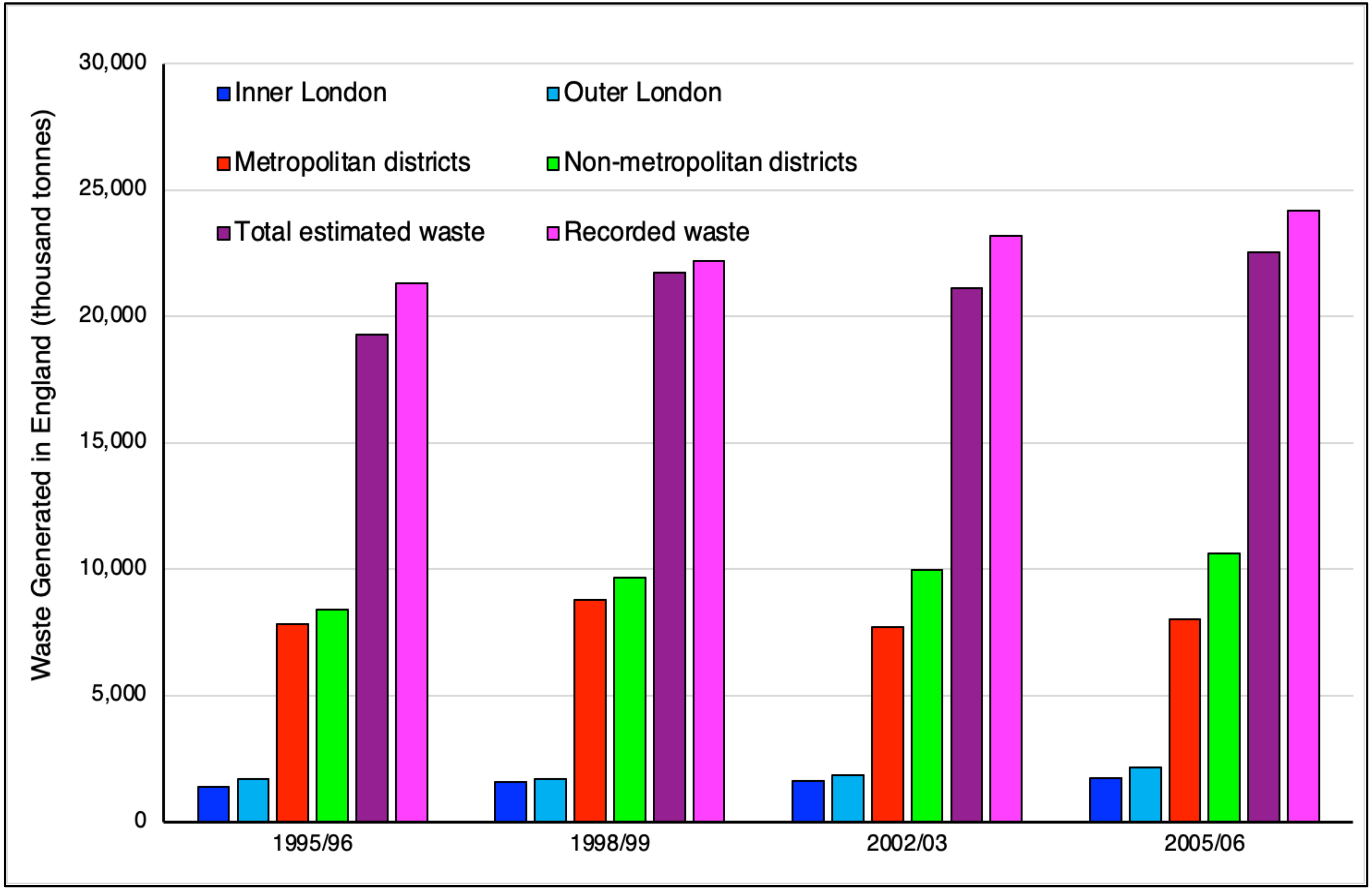


Figure 3.5(c): Estimated waste quantities using generated means for local authority categories against actual recorded waste 1995/96 – 2005/06.

Populations for each local authority category are available from 2 Government reports for Department of the Environment (1971) and Department of the Environment (1978). The population of London is available from a number of sources however, in respect of waste generation and management London had 3 separate eras. These being the London Boroughs with an approximate population of 3.3 million, the GLC with a population of 7.8 million and Inner and Outer London with populations of 2.7 and 4.3 million, respectively. Data published by the Department of Environment (1978) for the years 1974/75 through to 1977/78 and waste totals published by the Organisation for Economic Co-operation and Development [OECD] suggest the CIPFA disposal estimates for 1984/85 and 1986/87 are excessive, possibly by 9 to 10 million tonnes.

3.6.4 Reviewing the differences

Table 3.12 presents the percentage difference between this study's estimated generated waste outcomes and comparable recorded totals presented in Table 3.17.

Table 3.12: Percentage differences between estimated and recorded values for generated wastes presented in Figures 3.5 (a – c).

% Difference	1950 -1969	1970 - 1989	1990 - 2007
0 – 5%	1954/55 (4.0%) 1965/66 (3.2%)	1974/75 (1.8%) 1979/80 (0.9%)	1989/99 (2.2%)
5 – 10%	1968/69 (8.1%)	-	2002/03 (9.0%) 2005/06 (6.9%) 1995/96 (9.5%)
> 10%	1959/60 (10.7%)	1984/85 (40.0%) 1986/87 (36.5%)	1991/92 (14.8%)

Table 3.12 groups similar outcomes and notable discrepancies such that similar outcomes are considered as lying in the range 0 – 10% with discrepancies exceeding 10%. These differences arise because both figures have been estimated and published statistics, population statistics amongst others, are continually amended. For example, the population for England in 1962/63 used for this study is recorded by the Office for National Statistics [ONS] (2020a,

2020b) as 44,001,676 however the number used by the Ministry of Housing Working Party was 44,792,000. Aggregated population statistics for different local authority categories are only available for limited years as these are subject to review hence change. Sources for population statistics are included in Table 3.11.

No record for the total quantity of MSW for 1959/60 could be found. Therefore, the quantity is an estimated figure based upon 1961/62 for generated MSW reduced by the decrease in population for 1959/60. The same approach is adopted for any year where recorded waste totals were unavailable. The CIPFA disposal data included for 1984/85 and 1986/87 computes to daily generation rates of 1.44 kg and 1.48 kg respectively. The stated daily generation rate from waste collection authorities are 0.92 kg and 0.97 kg. This contradiction is that referred to by Stokes et al. (2013). Using these generation rates MSW generation results in 15.81 million tonnes and 16.79 million tonnes of household MSW. These compare more favourably with this study's estimates of 14.78 million tonnes and 16.24 million tonnes.

This study's MSW quantity for 1991/92 compares the estimated output to that figure cited by the OECD. The OECD waste quantity includes Northern Ireland, Wales and Scotland. These have been estimated and deducted from the overall total. These estimates are based upon similar, published data for that period.

3.7 Conclusion

This chapter presents established views in respect of waste generation and disposal data whereby reported quantities were based on limited test weighing and waste weights were inflated. The approach taken is to question those objections by assessing historical data's effectiveness as a series of samples able to establish a functional sequence of waste generation and disposal activity. Statistically, a sample is required to comprise an acceptable number of elements and to be representative of the sampled population. Waste generation and disposal form 2 sub-units of a more complex model that includes waste composition data (presented in Chapter 4) to estimate the material content in landfills. The model generates a series of outputs that reflects contemporary MSW management from the materials present in MSW streams (identifiable from

MSW composition sampling), to the quantities generated and how MSW was disposed.

For computed MSW generation rates to have substance it is necessary that the original data has an acceptable provenance. In this instance this is provided by knowledge of the total weight of MSW generated by a given population. Population records are available and can be verified. To achieve the requirement in respect of weighed MSW, local authorities weighing 50% or more of generated waste were used to establish a series of generation rates. A linear series of daily per capita MSW generation rates based on mean values which extend from 0.80 kg for the 1950s to 1.55 kg for 2005/06 is established. When used to compute recorded total MSW for England, similar quantities to those recorded are achieved. For purposes of providing data for use in the proposed model a series of lower and upper bounds are presented as a means to accommodate missing data. Available data would also allow regional or more localised generation rates to be utilised.

Waste disposal data is presented to confirm that 85 – 91% of waste was disposed to MSW landfills from the early 1950s through to the mid/late 1990s. Although generally accepted to form the disposal option for many local authorities, it does not apply to all. Three further issues are considered in regard to these data. These are necessary for the accurate application of the proposed model at the local or individual authority level: (i) how the remaining percentage of waste was disposed and by which local authorities (ii) how different local authorities waste strategies evolved over time and iii) how different data sets confirmed or contradicted others in respect of MSW disposal.

Chapter 4: MSW composition and flows into landfills

4.1 Introduction

The previous chapter reviewed data with which to quantify the generation of MSW and its subsequent disposal through the 4 strategies adopted by English local authorities. Data presented in this chapter replicate the flows of waste, or potential resource, into any landfill and allow these to be quantified. Landfills represent a stock of these resources. This framework is presented as an abridged system dynamics sketch (Figure 4.1).

MSW composition in the UK has, historically, been classified into a series of eight to eleven broad categories that each encompass numerous constituents (Coggins, 2009). Household and light commercial waste was stored in bins or plastic sacks, loaded into trucks with approximately 90% quickly discharged to landfill. There was little or no regard to whatever was contained in the bins or sacks. In either case, bin or sack these are the ‘black boxes’ of the refuse world (Chappells and Shove, 1999).

This chapter presents MSW composition data so as to, retrospectively, reconstruct the materials likely to be found in those ‘black boxes’. As a route to realizing this, the pertinent sections in this chapter are structured as follows:

- Section 4.3 identifies the materials found in MSW and considers how these are classified:
  - MSW composition and its characterization
  - MSW sampling
- Section 4.4 identifies the methods used to collect historical MSW data by considering:
  - The methods of history
  - Data sources and their management
- Section 4.5 presents, reviews and considers historical and recent composition data within the framework of this study’s 4 epochs, defined in Chapter 1, (Section 4.2):
  - Post-war Austerity: 1945 – 1955/60
  - The New Consumerism: 1955/60 – 1988/92
  - Directives and WEEE: 1988/92 – 1998/02
  - MSW Becomes Resource: 1998/00 – 2007

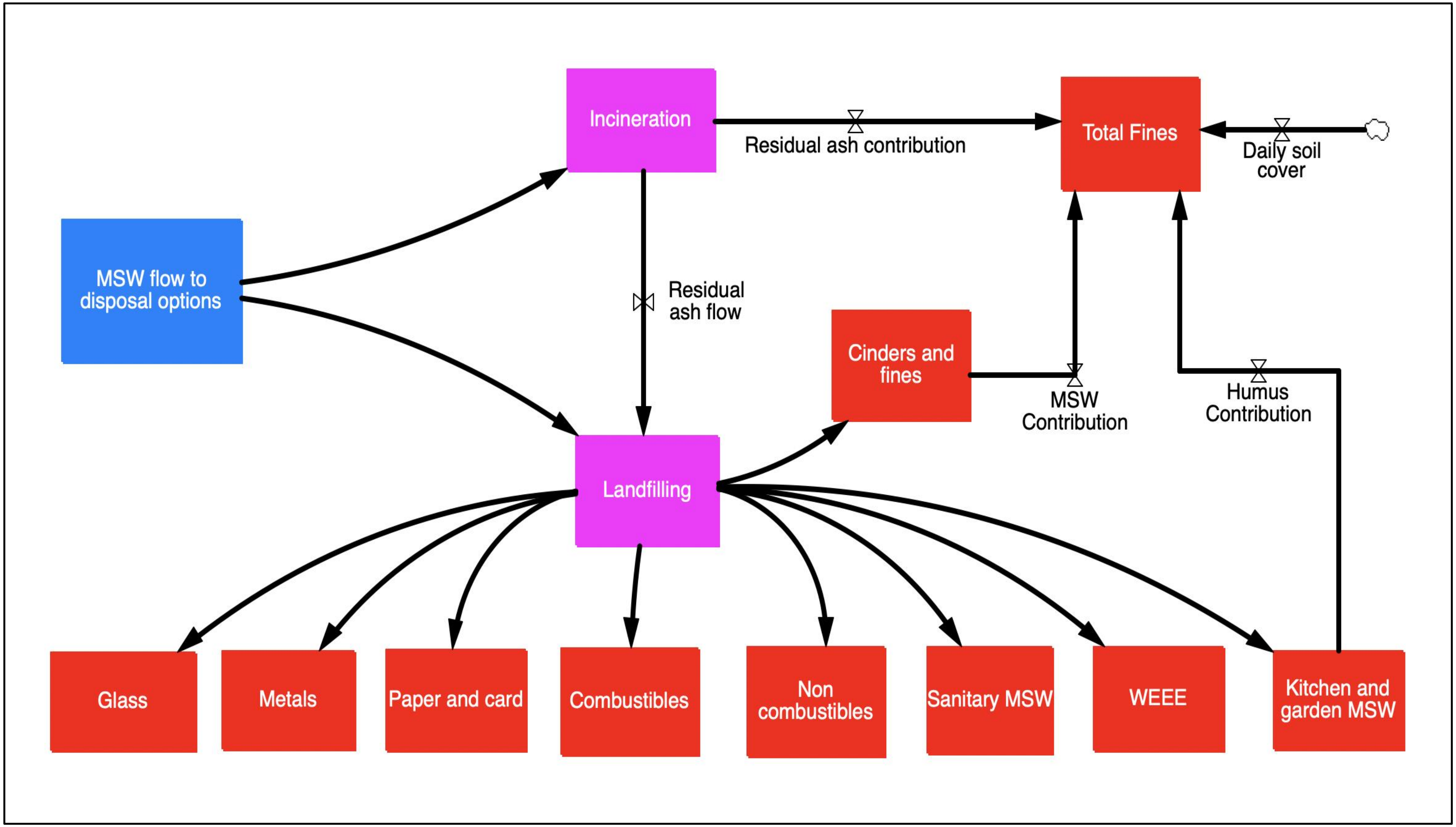


Figure 4.1: Abridged system dynamics sketch of major MSW materials flowing into landfills 1945 – 2007. Chapter 4 presents and reviews data for this sub-unit of the proposed model. The rectangular boxes represent waste flows which become stocks of a particular resource contained in a landfill or a group of landfills. The black arrows are flows. Cover material use is dealt with in Chapter 5 (Section 5.6.5).

Data for wastes other than MSW is more limited. The Department of the Environment (1971) reported 90% of industrial wastes (some 20 million tonnes) was disposed into private landfills. Leach et al. (1995) estimated the quantities generated from 2 data sources to be 86 million tonnes or 121.5 million tonnes annually but provided no detail on disposal location or ownership.

- Section 4.6 reviews and considers waste from other sources:
  - waste from civic amenity sites
  - industrial sources entering municipal landfills

Salvage and recycling removed specific materials from the household MSW stream before disposal into landfill. Quantitative data collected as a result of local authority recycling provides data from the late 1990s. However, prior to this only limited data exist. The impact of MSW recycling on material flows into landfills is reviewed in Section 4.7 and discussed further in Section 4.8.

Following the introduction of WASTDATFLOW in 2006, MSW entering landfills is considered to be recorded. However, MSW sampling analyses to 2010 are included as they are indicative of composition occurring in preceding years.

4.2 Relevance to the aims and the objectives

Identifying and locating MSW composition studies and the data these studies collected was necessitated by the first research question and objectives 2, 3, 4 and 5. In all respects it is complementary to the previous chapter. However, the collection of MSW composition data rapidly developed into a quasi-stand-alone project due to, for the most part, the age and location of publications. The methodology used to collect and catalogue these data are discussed in Section 4.4 with the collected data presented and reviewed in Section 4.5.

4.3 MSW composition

4.3.1 MSW composition and its categorization

The necessity to identify the composition of MSW was understood before World War II. Coggins (2009) reports on the “pioneering work” of MSW composition samplers during the 1930s from which was developed eleven quite general categories of household MSW (Table 4.1) were established. From 1994, the EU and UK developed (i) the European Waste Catalogue [EWC] and (ii) the UK

Classification where the latter allowed further sub-division to aid local authorities introduce and implement the development of household and commercial recycling. This study adopts those categories set-out in Table 4.1 however, it does not differentiate between different plastics or metals. Non-ferrous metals feature as small percentages in composition studies due to their marketability.

Table 4.1: Established categories for MSW sampling including a brief description of the materials included within each. Table reproduced from Coggins (2009, p.13).

Waste classification	Typical materials included
Paper and card	Newsprint, cardboard and tissue
Plastic film	Food wrappings, carrier bags, refuse sacks
Dense plastic	Beverage + other bottles, toys, food trays
Textiles	Clothing
Miscellaneous combustibles	Shoes, wood, carpets
Miscellaneous non-combustibles	Bricks, stones, ceramics
Glass	Bottles, jars
Putrescibles	Kitchen and garden wastes
Ferrous metals	Beverage and food, cans, batteries
Non-ferrous metals	Beverage cans, foil, food trays
Fines (< 10 mm)	Irrespective of composition, ash and soil

4.3.2 MSW sampling

MSW composition sampling is reliant on accurate sorting and recording (Higginson, 1966). As a practice, it is expensive and unpleasant. Furthermore, the process, for much of the study period, lacked a standardised methodology (Parfitt and Flowerdew, 1997; Dahlén and Lagerkvist, 2008). Not only did sampling lack standardisation but often local authorities failed to recognise the variation in MSW generated by different socio-economic groups. This was less obvious and had little impact during the first epoch where MSW comprised mainly grate ash. However, during the growth of consumerism (2nd and 3rd epochs) where incomes increased and led to: i) a rapid increase in the generation of MSW and ii) hyper consumption (Campbell, 2015b) such imprecise sampling led, in some respects, to the inclusion of diverse MSW generators into a far too simple structure. Appreciation of the issues with waste sampling led the World Health Organisation (1971) to venture to encourage the development of a standardised set of sampling procedures. Two studies that include a thorough and objective analysis of waste sampling in the UK are: (i) Parfitt et al. (1994) which considered the socio-economic and seasonal factors in household MSW sampling and modelling and (ii) Parfitt (2002) the effectiveness of sampling by questioning the effects of methodological differences between seventy studies undertaken by local authorities in the UK during the period 1999 to 2002.

Seasonal variations are mitigated using an annual time step however, care has to be taken to ensure that where specific seasonal data is utilised this is balanced with complementary data to represent the entire year. Identifying the representativeness of historical composition samples across a broad range of socio-economic variables is more problematic. Sampling data confirm there is neither an average sample nor a sample that reflects 'typical' household MSW (Coggins, 2009). Differing socio-economic factors contribute to a range of weights generated. One such factor, affluence, is highlighted by Warren Spring Laboratory [WSL] and Aspinwall and Company (1993) in two MSW composition analyses (Table 4.2).

Table 4.2: Differences in MSW composition of 2 socio-economic groups determined by affluence, revised to percentages. Reproduced from WSL and Aspinwall and Co. (1993).

Waste classification	More-affluent (% by weight)	Less-affluent (% by weight)
Paper and card	32.9 – 45.4	26.2 – 36.1
Plastic	10.1 – 10.5	9 – 11.1
Textiles	1.2 – 2.1	1.4 – 3.5
Miscellaneous combustibles	3.5 – 7.2	6.9 – 8.3
Miscellaneous non-combustibles	1.2 – 6.3	1.4 – 2.8
Glass	7.6 – 9.3	5.6 – 9.7
Putrescibles	18.6 – 19.4	21.4 – 22.2
Metals	5.8 – 6.3	7.6 – 8.3
Fines (< 10 mm)	4.7 – 8.0	6.9 – 11.7

In addition to diverse socio-economic groups being bracketed together, composition samplers reported on a limited range of categories. These categories, whilst being practical for sampling purposes, concentrate wastes composed of similar (not necessarily the same) material. To ameliorate the issue with variation in composition, data will be separated into respective epochs and presented as a range of values. Unfortunately, and where historical data is concerned the categories are pre-determined. However, and as will be demonstrated, the components having a greater resource value tend to have the narrower ranges and are, individually, below 12% of the material content. Furthermore, these materials maintain a greater temporal consistency. These factors are presented and reviewed in Section 4.5 and discussed in Section 4.8. Over the longer-term, defined as the period 1945 – 2006, there has been a non-uniform, increasing trend in the MSW generation rate. However, MSW generation does not conform to a specific pattern. Wilson (1981) identifies the problem with extrapolating MSW trends from data sets. The mid-late 1960s witnessed a change in the composition of household MSW from cinders and ash to MSW with a higher paper content. Expectation was that this would continue however, during the early 1970s paper and card started to reduce.

In identifying the complexities arising from the sampling processes employed, Parfitt (2002) proposes an integrated data set that included seasonal changes in the waste stream. Following a review of 70 MSW composition studies he determined only 27 were acceptable due to sampling inconsistencies. A major concern was the failure to account for household recycling resulting in 'leakage' from analysing only 'bin' or 'sack' MSW.

The impact of recycling and civic amenity waste on flows of materials into landfills are considered in Sections 6.1 and 6.2. However, to avoid the effect of 'leakage' only composition data that include these MSW elements are utilised. Such studies aid the formulation of a MSW modelling framework as they deem it necessary to identify specific issues and establish bounds within which composition data can be incorporated into a proposed model.

4.4 Waste composition data: collection methodology

4.4.1 The methods of history

Due to the nature of published MSW composition literature the search methodology described in the literature review was unsuitable for many publications containing MSW composition data. The adopted approach owes more to the methods employed by historians when searching for their sources. Historians define primary sources as official documents and records, immediate, first-hand accounts, data sets and surveys and original research. For this study these were often found in bound collections of conference papers and contemporary scientific textbooks containing data or references to waste, its collection or disposal. Historical texts and newspapers were also reviewed to achieve a rounded and informed view of what was a dynamic period in British history (Tosh, 2015; The Open University, 2016).

At the academic level, social and economic history is an interpretation and record of human behaviour (Tosh, 2015). Surprisingly, the examination of historical waste extracted from excavations into landfills is an established source for historians and archaeologists (Rathje, 2001). Whilst such studies have provided invaluable perspective, often contradicting anecdotal recollections, their methods are science based and are applied to this study. Rathje's Garbage Project was, primarily, an archaeological investigation into recent human behaviour. However, significant data in respect of waste decomposition resulted together with the (US) Environmental Protection Agency's [EPA] first study of hazardous household waste (Rathje, 2001; Rathje and Murphy, 2001).

4.4.2 Data sources and management

Reviewed sources were categorised into one of the following four groups: i) specific composition studies from any source, ii) conference papers, iii) consultancy reports and iv) textbook and journal publications. Journal publications were sub-divided into a) data and history, b) waste generation modelling and c) material recycling. Some 36 sources were reviewed for waste composition data. From these 165 distinct composition samples were recorded and these are identified in Appendix 3.2. Care was taken that samples were not double counted by first checking the source and then comparing each samples recorded data for individual MSW categories.

4.5 Four epochs of landfilling

4.5.1 Post-war austerity: 1945 – 1955/60

Post-war Britain was a time of reconstruction and change. The extensive use of coal for power generation and as a means of domestic heating resulted in a high cinder content in MSW. Published composition data show cinder content to lie in the range 54 – 80% by weight. Higginson (1960) proposed average post-war household refuse to contain 67% cinders and other fines, 9% paper and card, 4% kitchen and garden waste, 5.3% glass and cullet, 1.3% textiles 0.5% bones, 5% metals with the remainder as miscellaneous combustible or non-combustible materials. However, the use of solid fuels was curtailed following the Great Smog of London in 1952 with the 1956 Clean Air Act prohibiting the emission of 'dark smoke' from any building. Consumption of coal for domestic purposes reduced from 38.1 million tons in 1957 to 23.4 million tonnes in 1968 (Skitt, 1972). Whilst local authorities were provided with means to enforce the provisions of this Act, little change occurred to waste composition until the early 1960s.

Published MSW composition data included a single or mix of seasons with generators defined by property type discounted. Such an approach to sampling was considered to reflect the waste stream generally. Two waste professionals, P D Fairlie and A E Higginson, identified the requirement for providing detailed analyses in respect of seasonal variation and comparative statistics between different property types and local authorities. Data presentations in these 4 sections consist of a chart (here, Figure 4.2) which offers a snapshot of the averaged MSW composition over the epoch a series of boxplots which provide more detailed composition data (Figure 4.3 a - d). The data for 1945 – 1960 (Figure 4.3 (a – c)) confirm Higginson's averaged composition for that period. Figure 4.3 (d) includes two MSW sampling surveys undertaken for modern (1960) terraced properties and high-rise apartments using predominantly electric central heating.

Figure 4.3 (b) provides a comprehensive review of MSW composition for 1953 of eight local authorities. The boxplot tails and outliers reflect the range of property types surveyed together with seasonal variations resulting from increased use of solid fuel during colder periods. Published results for Islington, a London Borough with a combination of centrally-heated multi-storey flats and traditionally heated housing is responsible for the lower quartile tail and outlier exhibited by the dust and cinders boxplot having a range of 14 – 31.5% by weight of these materials with the summer being the lower figure and, in this case, spring being the greater. Islington and Easington, a mining town with a population of approximately 5,000

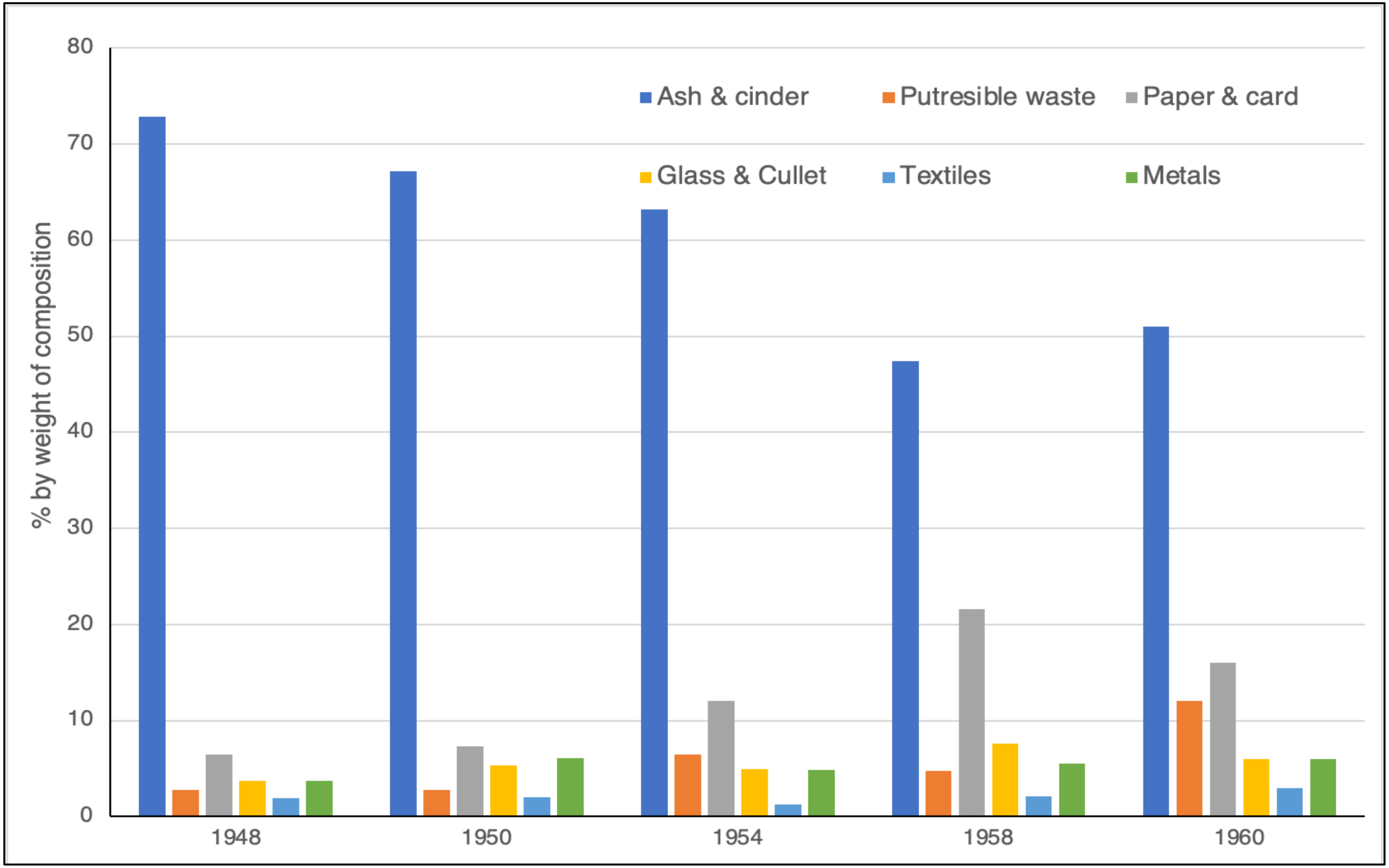


Figure 4.2: Evolution of domestic MSW from individual recorded (averaged) sampling exercises, 1948 – 1960. Data: 1948; AV Bridgewater (1986). 1950; JC Wylie (1955). 1954; AE Higginson (1966). 1958; AV Bridgewater (1986). 1958; Flintoff & Millard (1969).

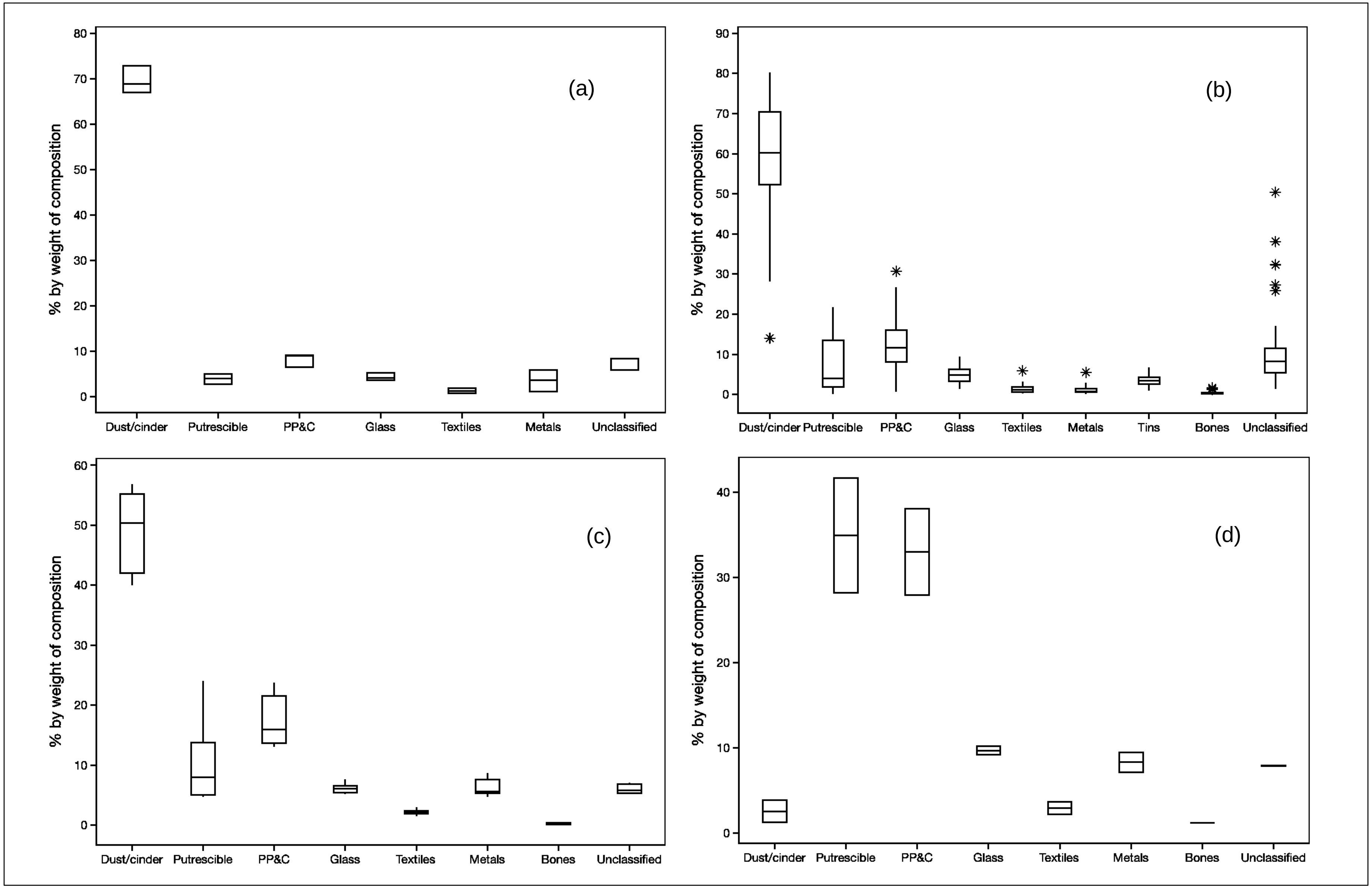


Figure 4.3 (a - d): Composition of domestic MSW (a) 1945 – 1950 (3 sampling surveys). (b) 1950 – 1955 (32 sampling surveys) (c) 1955 – 1960 (7 sampling surveys). (d) 1960 modern house with central heating (2 sampling surveys)

is responsible for the outliers in the unclassified boxplot. Islington contributes the lower two values with Easington the remaining values. These values are outliers with the mean for the remaining values equating to 7.4%. Typically, and throughout each epoch, paper and card, metals, textiles and glass and later plastic and waste electrical and electronic equipment [WEEE] are referred to as value materials due to their resource potential when removed from a landfill.

It would be incorrect to assume regulation had little or no impact on MSW during this period. The aftermath of World War II saw direct regulation of consumption on the entire population of the UK which, following a change in the Government in the middle of the 1950s led to the demise of rationing and the progressive transition towards a consumer driven society (Zweiniger-Bargielowska, 2000). MSW composition more than mirrored this austere lifestyle and was similarly reflected by the materials flowing into landfills.

4.5.2 The new consumerism: 1955/60 – 1988/92

Whilst the 1940s and 1950s saw a degree of consistency in generated MSW, this cannot be applied to this second period where an evolution in MSW composition reflects the closure of the policy attributed to the period of post-war austerity, the cessation of solid fuels, changes in consumption patterns aligned with an increase in average income and the introduction of plastic and packaging into the MSW stream. Published sampling data reinforces this evolution. However, exceptions remained, and Wilson (1972) cites dust and cinder levels in late 1960s MSW remaining at 68% by weight for mining areas where smoke control measures had not been instituted. Micro-data sets will occur and are necessary when considering individual or a collection of localised landfills. Their impact at the macro-scale, that is at the regional or national level is less observable as can be deduced from Figures 4.4, 4.5 and 4.6. However, the relevant issue is not the ash quantity but the quantities of materials of value which, in this and the previous period, are combustible materials, glass and more importantly metals. Putrescible wastes quickly decompose forming landfill gas with the residue becoming humus.

Higginson (1966) produced a summary chart (Figure 4.4) comparing MSW components sampled from 3 English local authorities located in the Midlands, the North and the Southeast. Data was collected throughout 1963, 1964 and 1965.

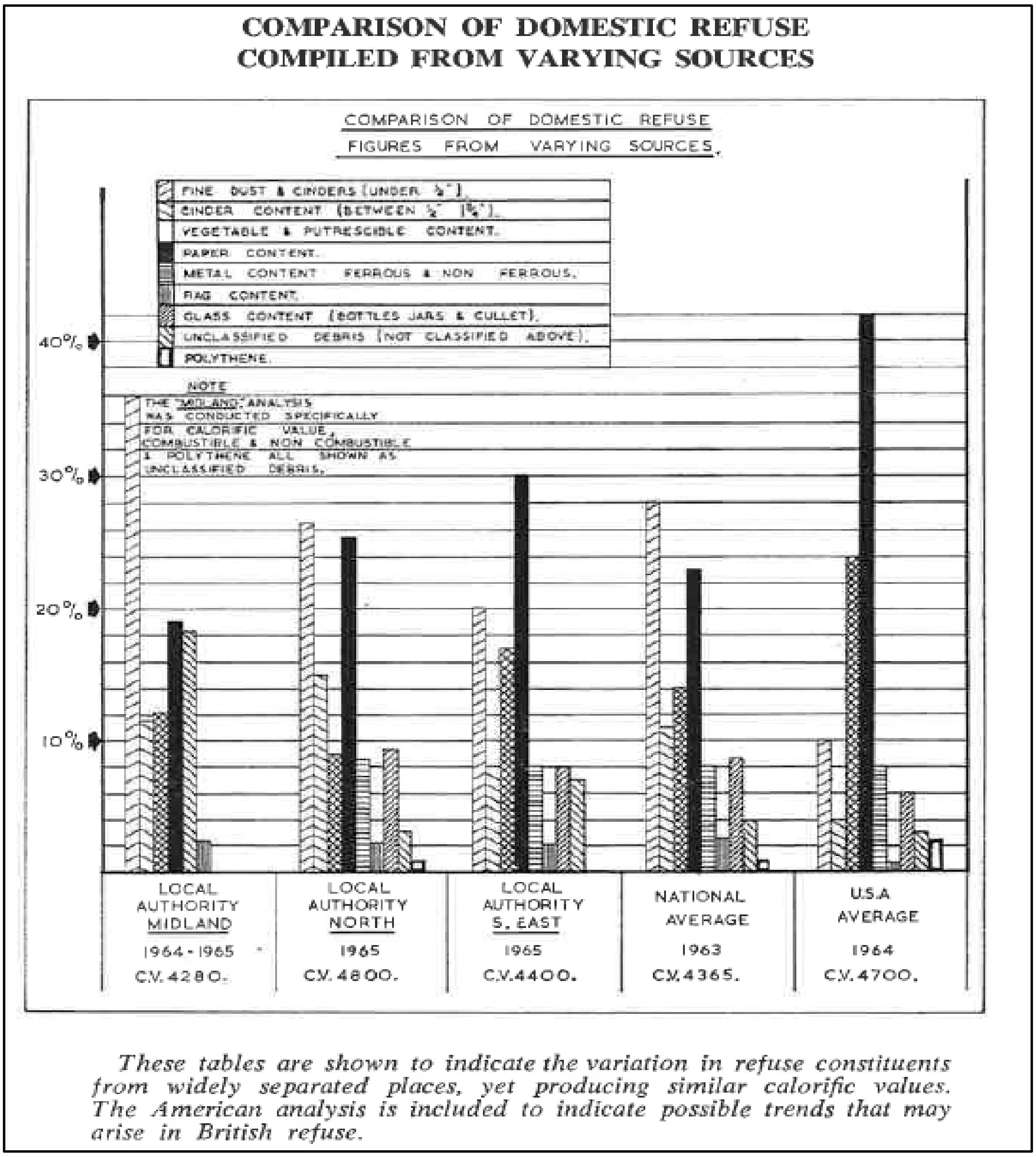


Figure 4.4: Reproduced bar chart from Appendix IX, *The Analysis of Domestic Refuse* published by A E Higginson Assistant Divisional Engineer, GLC (1966). Dust and cinder content is represented by two columns and putrescible is the checked area in this figure.

Furthermore, the chart compares MSW generated in England with that of the USA where dust and cinders are replaced by paper and card as the predominant waste material. Higginson predicts a similar MSW composition profile may develop in England. He was correct in doing so as this was occurring and manifested itself during the early 1970s. MSW composition typified by centrally heated housing becomes the 'average' household waste composition following the installation of the national gas grid. Figure 4.5 expands upon Higginson's survey data presented in Figure 4.4 and includes MSW composition for this entire period.

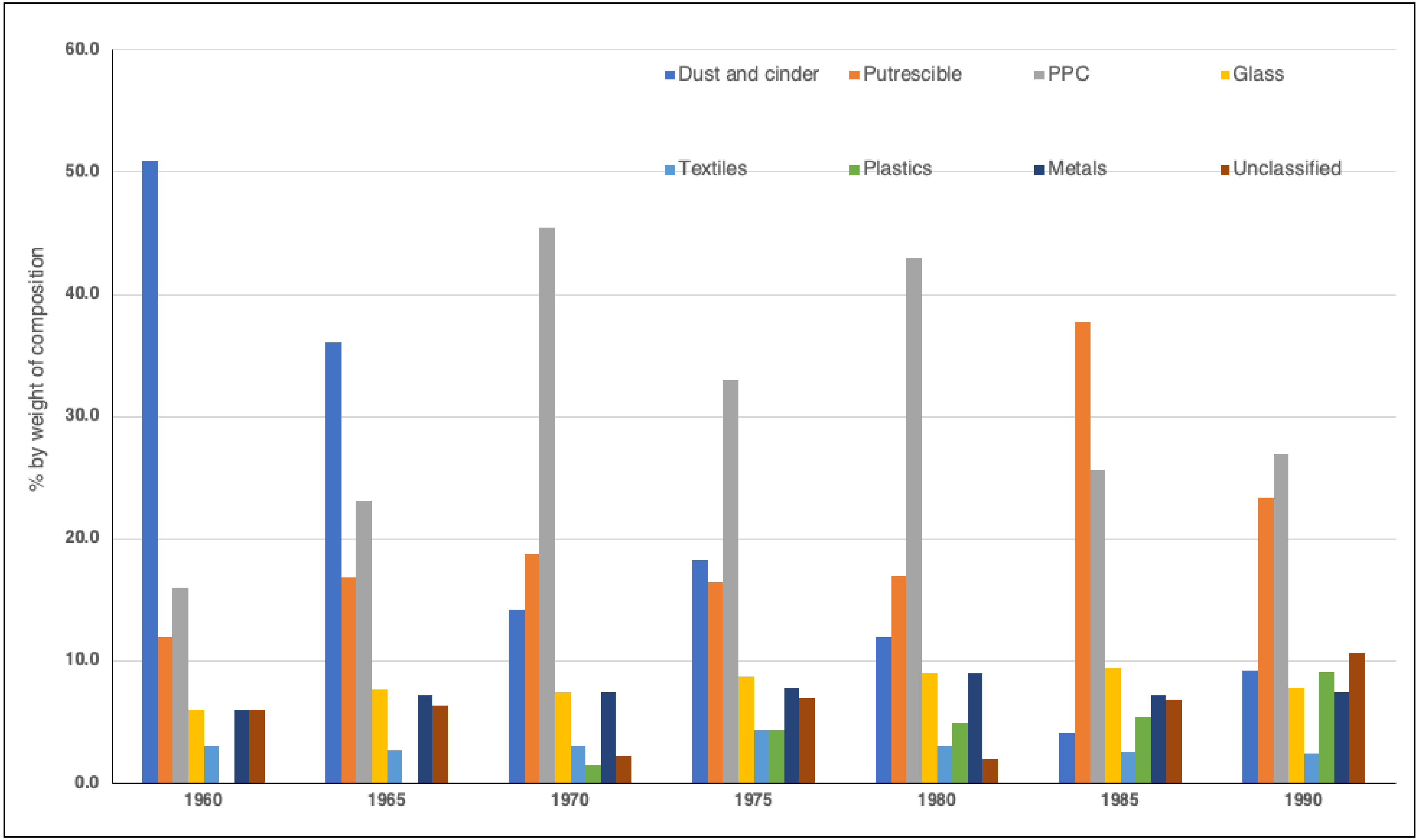


Figure 4.5: Evolution of domestic MSW from individual recorded (averaged) sampling exercises, 1960 – 1990. Data: 1960; Flintoff & Millard (1969). 1965; AE Higginson (1966). 1970 and 1975; AV Bridgewater (1986). 1980; DC Wilson (1981). 1985 N M Rufford (1985). 1990; J Parfit (2002). Plastic was not recorded on the 1960 and 1965 surveys presented here but is recorded from 1965 by Higginson and 1966/67 by the Greater London Council.

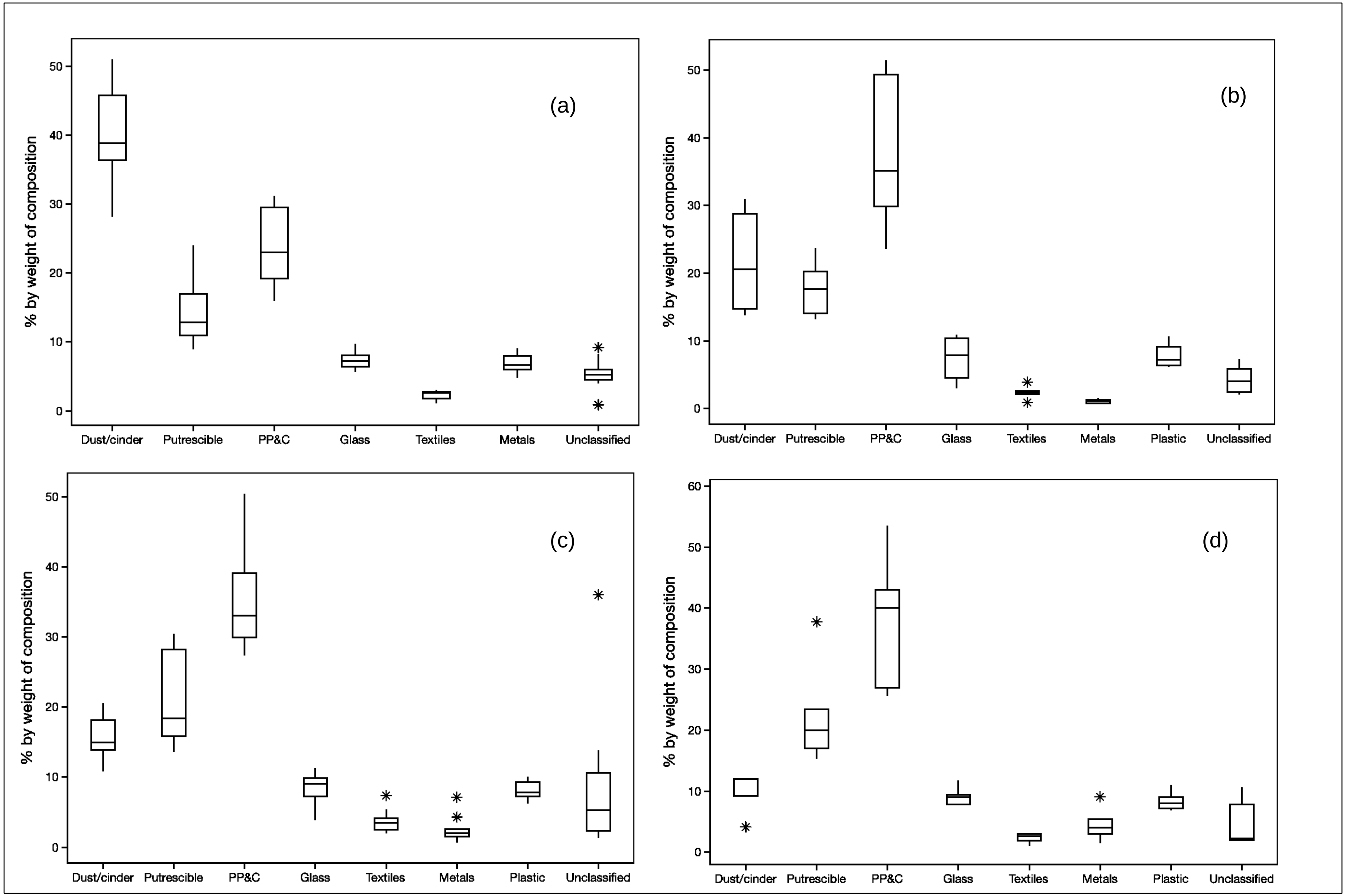


Figure 4.6 (a - d): Composition of domestic MSW (a) 1960 – 1965 (15 sampling surveys). (b) 1965 – 1970 (8 sampling surveys) (c) 1970 – 1979 (13 sampling surveys). (d) 1980 – 1992 (6 sampling surveys).

The figure is presented as a bar chart for comparison with Figure 4.4 and to better illustrate the evolving nature of MSW over this period. For a more complete estimate of landfill content, it is necessary this dynamic is captured in the resulting model. To complement these data, Figure 4.6 (a – d) presents a series of boxplots denoting the data range of individual household MSW components. Data sets, pre-1965, for blocks of flats with central heating and smaller local authority or mining areas are not included in the presented data sets. Mining areas along with seaside towns, the latter having an influx of summer visitors, tended to represent smaller local authorities and, more relevantly, skew data.

Two late 1960s Acts of Parliament would impact on the composition of MSW flowing into landfills. The second of the Clean Air Acts (1968) further reduced grate-ash however, the 1967 Civic Amenities Act which introduced household recycling centres or bring sites did not have a real impact until the 1980s and by the mid-1990s were responsible for 20% MSW generated (Parfitt and Flowerdew, 1997). Furthermore, Parfitt and Flowerdew (1997) reported household MSW analyses tended to not include materials taken to these sites. Limited data for these facilities is available and is presented in Section 4.6 together with clarification of their impact on MSW flows and how these will be modelled.

The UK's membership of the European Economic Community [ECC] commenced on 1st January 1973. The first of the ECC pollution control Directives was issued in 1975 along with the adoption of the 'polluter pays' principle (European Union, 2020). In many respects the impact of regulation on materials flowing into landfills remained indirect however, Her Majesty's Government convened four reports from the Royal Commission on Environmental Pollution which initiated controls over the disposal of poisonous wastes through the Special Waste Regulations (1980).

4.5.3 Directives and WEEE: 1988/92 – 1998/02

Waste composition as reflected in respective studies undertaken during the period 1988/92 – 1998/02 and for that matter, extending up to 2006, did not vary perceptibly other than in the quantities of putrescible waste generated. However, a very real transformation occurred in the materials flowing into landfills as a result of EU directives and their subsequent adoption into law. Figures 4.7 and 4.8

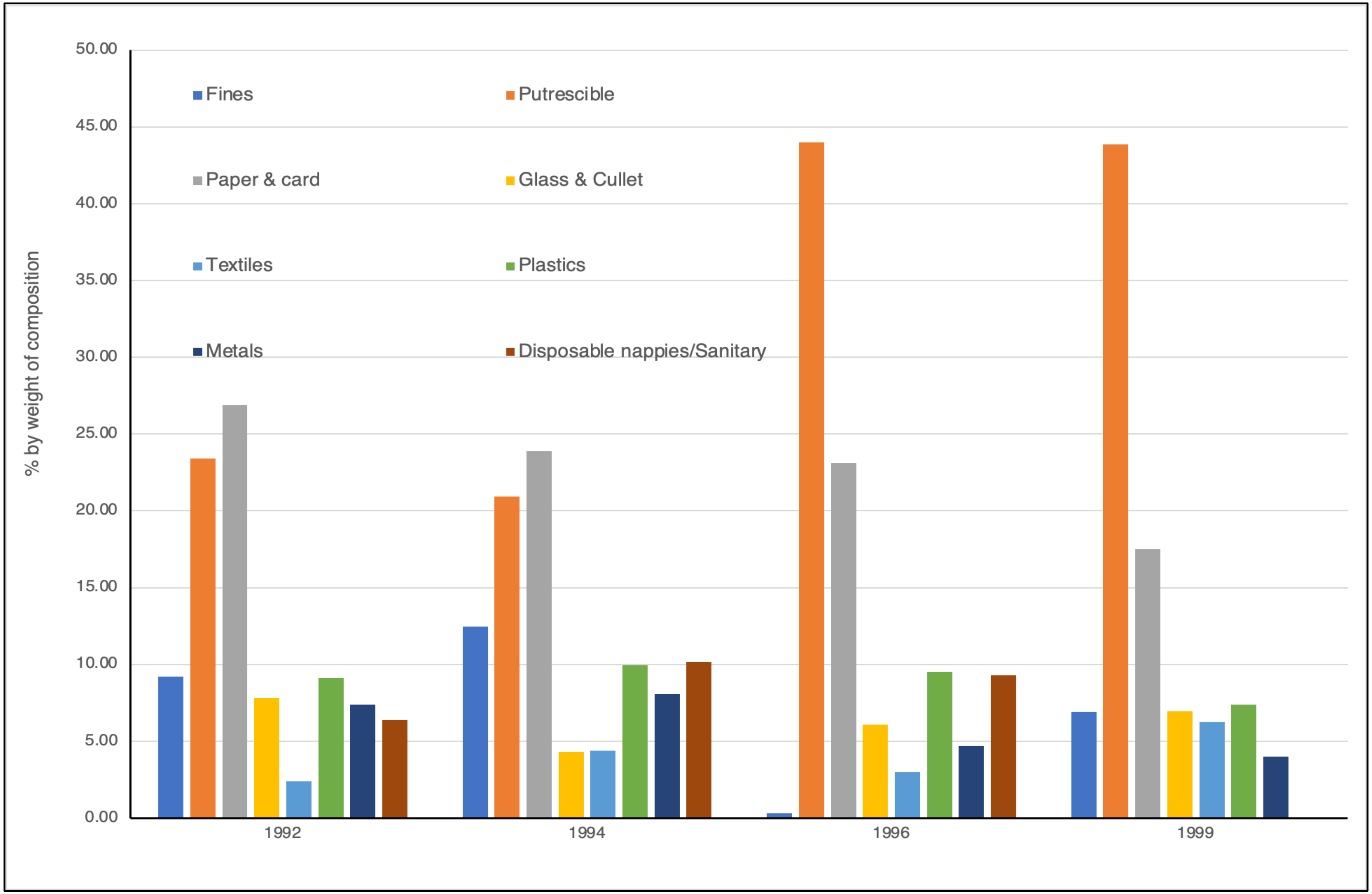


Figure 4.7: Evolution of domestic MSW from individual recorded (averaged) sampling exercises, 1992 – 1999.

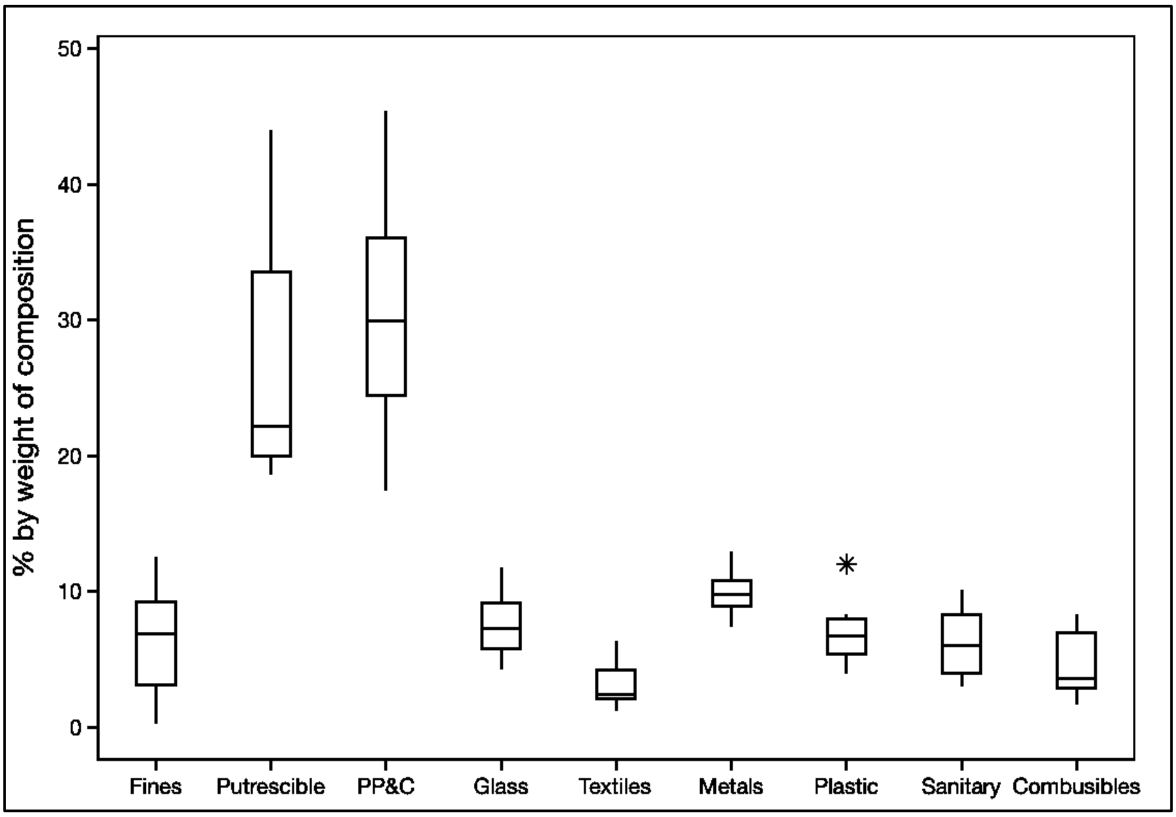


Figure 4.8: Composition of domestic MSW 1992 – 1999 (12 sampling surveys).

present MSW composition data for this period which witnessed both the rapid development and mass selling of the personal computer and the mobile phone.

This period and the next reflect significant regulatory intervention in MSW flowing into landfills not only in respect of their composition but also their quantity. This third period sought to control the composition of MSW flows to landfill with the fourth seeking to divert MSW components. Despite MSW generation in England increasing from 25.2 million tonnes in 1995/96 to 29.6 million tonnes annually in 2005 (35 million tonnes for the UK), where 85% was disposed into landfills in 1995, regulation had reduced this to below 10 million tonnes by 2015 (DEFRA, 2000; DEFRA, 2017; EUROSTAT, 2017b). The emphasis of regulation leading to the implementation of recycling, is the reason for splitting the period 1988/92 – 2006 where recycling removed value materials from the waste flow to landfills.

Not evident in these analyses are waste electrical items although e-waste items were a common feature of the household as the use of transistors in consumer appliances was commonplace from the mid-1950s with more sophisticated semiconductors introduced from the 1970s (Stokes et al., 2013). Often these items were resold or when disposal was necessary, were too large to be taken by LA collection services and were likely to be deposited at the civic amenity sites. However, the 1980s witnessed the mass selling of the home computer, the video recorder and later the mobile phone. Wilson et al. (2017) propose that significant numbers of mobile phones are in dead storage or hibernation. Is it possible this applies to other items of electronic equipment? Latterly, a United Nations University report estimated 41.8 million tonnes of e-waste was generated in 2014 valued at US $52 billion (Kuehr et al., 2015). Waste electrical goods are recorded in the next epoch however, their percentage by weight remains low (Parfitt, 2002).

4.5.4 MSW becomes a resource: 1998/00 - 2007

Article 3 (b) of the EEC Directive 91/156/EEC directed member states to recover materials from the waste stream. Paragraph 8 of Council Directive 1999/31/EC included for recovery to be enhanced with article 13 setting target dates for the removal of biodegradable materials from MSW entering landfills. In England, household MSW was increasing (Department for Environment, Food and Rural Affairs, 2000a) and many local authorities, conscious of this issue and the requirements of each directive reviewed their waste streams with a view to establishing recycling and diverting biodegradable materials away from landfills. In addition, the introduction of the landfill tax would impact on disposal budgets, but this was not immediately the case (Stokes et al., 2013).

Parfitt (2002) examined household MSW compositional data and the drivers leading to the increase in domestic generation. His study provides a useful snapshot of the material content of the English MSW stream whilst providing a somewhat derogatory view of current local authority waste sampling. Reference to this study is included in Section 4.3.2 however his detailed compositional analysis for ‘bin’ and civic amenity waste is included as Appendix 4.1. A generalised composition is presented in Figure 4.9 which also compares English domestic MSW to an average for Europe. Whilst the waste components are similar, these will not reflect the materials flowing into respective landfills due to the differences in member states recycling. This is considered further in Chapter 5 (Section 7.6). Figure 4.7 identified the increase in the generation of putrescible wastes where this has stabilised by 2000/01 with garden waste equating to approximately 15% of the quantity and kitchen and food waste 27%. The value materials: glass, metal, textiles and now paper and card would become the materials targeted for recovery and by 2006/07 8.94 million tonnes (30.6%) of MSW was being recycled (Department for Environment, Food and Rural Affairs, 2018).

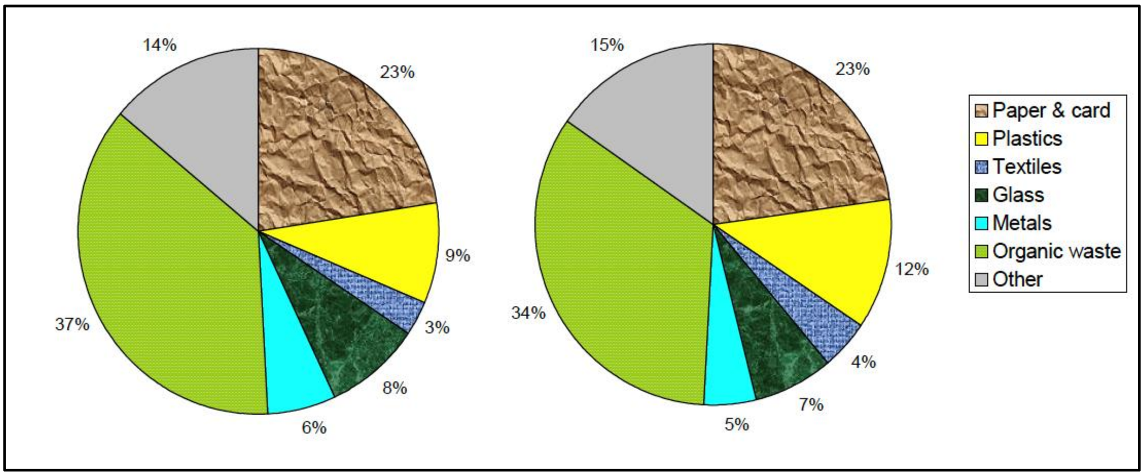


Figure 4.9: Composition of English domestic MSW (left-hand pie chart) compared to an average for Europe. Figure reproduced from Parfitt (2002 p.15).

For this final epoch, data obtained from waste composition studies will require adjustment by an increasing percentage to accommodate the increase in recycling. Salvage and recycling data are presented and reviewed in Section 4.7.

The Department for Environment, Food and Rural Affairs (2003) identified that during 2000/02 60% of MSW was collected at the household, 15% at civic amenity sites, 10% collected for composting, 11% was non-household waste and 4% was labelled as other household collections. The data in Appendix 4.1 identify 23.96 million tonnes of household MSW generated for 2000/01 with 18.44 million tonnes collected from households and 5.52 from civic amenity sites.

4.6 Other waste to landfills

4.6.1 MSW from civic amenity sites

The 1967 Civic Amenity Act required local authorities to provide sites for the acceptance of household waste. Referred to as “bring sites” and now household waste recycling centres [HWRCs], they reduced kerbside collections. However, data published in the Department of the Environment report (Table 4.3 (a)) is the earliest data located for these sites. MSW quantities arising between 1979 to 1995/96 will require estimating as no data could be located. However, after 1995/96 data has been published and these are summarised in Table 4.3 (b).

There is no clear methodology to estimate the missing data. Three of the years 1977/78, 1997/98 and 2000/01 saw increases in the use of civic amenity centres with 1999/00 a reduction. The period from 1979 – 1995 was one where local authorities invested in civic amenity [CA] sites with many opening however, these data are not available.

Table 4.3 (a): CA site MSW collection data. Data reproduced from Department of the Environment (1978 p.5). Total MSW Table 5, Civic amenity MSW Table 2.

	Total household & commercial MSW (tonnes)	CA site MSW (tonnes)	CA MSW as a % of total MSW	Number of authorities providing CA data
1974/75	23,743,000	1,031,000	4.3	29
1975/76	23,270,000	1,273,000	5.5	41
1976/77	22,216,000	1,393,000	6.3	38
1977/78	22,277,000	1,941,000	8.7	42

Table 4.3 (b): CA site MSW collection data. Data reproduced from the identified sources in column 5.

	Total household & commercial MSW (tonnes)	CA site MSW (tonnes)	CA MSW as a % of total MSW	Data source
1995/96	22,500,000	4,000,000	17.8	DETR[1] (1997)
1996/97	22,550,000	4,260,000	18.9	DEFRA[2] (2000)
1997/98	23,340,000	4,900,000	21.0	DEFRA[2] (2000)
1999/00	24,760,000	4,574,000	18.5	DEFRA[2] (2003)
2000/01	25,592,000	5,521,000	21.6	Parfitt (2002)
2006/07	26,045,000	5,403,000	20.7	DEFRA[3] (2008a)

Notes: 1) Department of the Environment, Transport and Regions. 2) Department for Environment, Food and Rural Affairs. 3) WR0119: A Review of Municipal Waste Component Analyses – Appendix 4.1

Accounting for MSW site disposal is problematic particularly for the periods before Parfitt's 2002 publication. The approach taken in this study identifies a series of MSW generation rates from which it calculates flows of materials to landfill (Chapter 3). The composition of these flows is considered in this Chapter with Section 3.2 identifying the impact of CA sites upon MSW composition studies.

Studies specific to CA sites for the early 1990s identify kitchen and garden waste to be a dominant fraction averaging 38% and rising to 64% during the Spring. Paper and card to be 2 – 5%, with glass, textiles, plastic 0.3 - 2.8% and metals 2 – 9.8% (Coggins et al., 1990; WSL and Atkinson, 1993; Parfitt 2002). Excluding the Spring increase, kitchen and garden MSW reflect the percentages identified in most household composition studies. The value materials, glass and textiles are understated by 50% of their values when compared to household composition sampling namely 2.5 – 9.1%. Metals are very similar 2.5 – 10%, whilst household plastic is 5 – 10% with paper and card 20 – 30%. This study has located 3 CA site composition studies. Whilst it is accepted each will have been conducted thoroughly they represent only a very limited number of catchment areas. It is proposed to use household composition rates whist accepting these studies overstate glass and textiles. Metals and garden waste are similar with further consideration given to plastic in Chapter 5 (Section 7).

4.6.2 Construction and industrial waste

The 1971 Department of the Environment report published by the working party contained a thorough examination of contemporary waste issues. At this point in time (and until the mid-1990s), municipal waste included only those wastes collected by local authorities. Latterly this definition is superseded by the European Commission definition of MSW that includes all wastes of a similar composition, wherever generated. In addition to household MSW generation data, the report identified the large quantities of industrial and commercial wastes arising from a recovering economy. Industrial waste, including mining and power station wastes was estimated to exceed 20.32 million tonnes (Department of the Environment, 1971). The Confederation of British Industry reported that approximately 11.27 million tonnes of wastes were produced by 1000 premises surveyed. The report caveats that specific industrial waste data and large bulky wastes are not included and should be treated at the local level.

For the most part, approximately 90%, of industrial waste is reported to be disposed at privately owned landfills. Local authorities were concerned with preserving volume and tended to refuse such wastes. However, trade and commercial wastes collected, disposed or received at LA disposal operations are a factor in the composition of many landfills (Department of the Environment, 1971). Figure 4.10 and Table 4.4 summarize the principal data published in this report, although very limited, is an indication of the contemporaneous situation. To this end, the Working Party requested data and information from the National Association of Waste Disposal Contractors, but no reply was forthcoming (Department of the Environment, 1971). The largest recorded waste stream disposing into local authority landfills was from the construction sector.

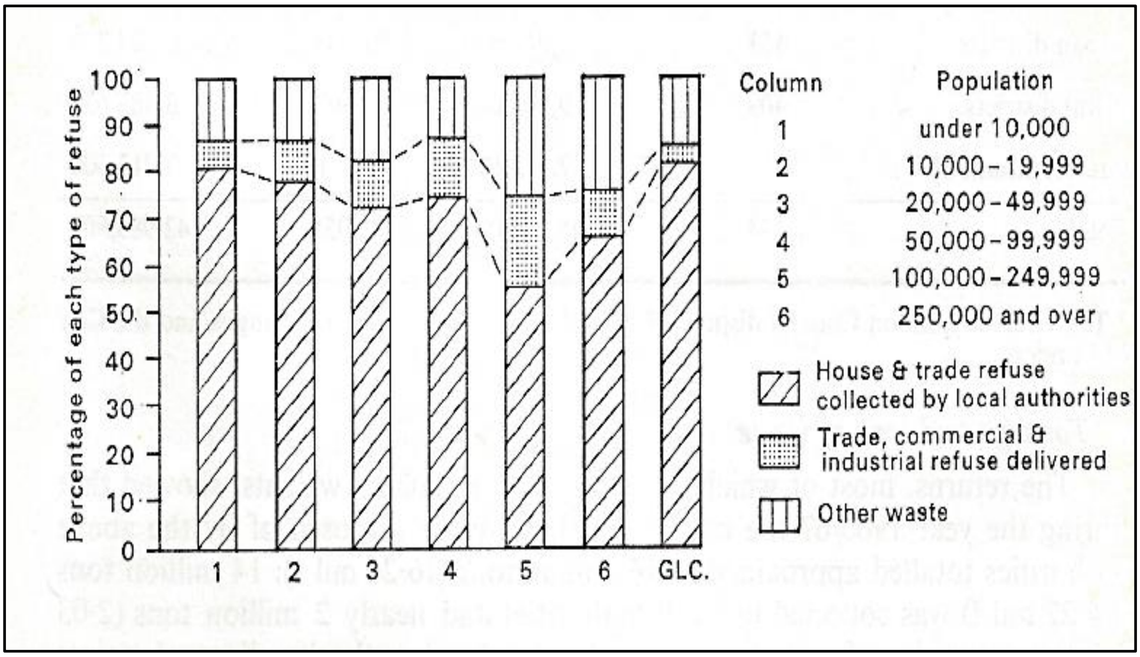


Figure 4.10: Local authority waste disposal by type and size of population for 1966/67.Figure reproduced from Department of the Environment (1971 p.6).

Table 4.4: Local authority MSW data for 1966/67 taken from 1051 questionnaires. Data reproduced from Appendix N, Table1 Department of the Environment (1971 p.177).

	Trade and commercial refuse delivered direct (tonnes '000)			
	Combustible	Non-combustible	Undefined	Total
County Boroughs	228.01	410.79	349.42	988.22
London Councils	10.65	93.77	-	104.42
Boroughs	102.41	162.31	105.96	370.68
Urban Districts	82.88	157.99	79.09	319.97
Rural Districts	49.80	82.58	32.26	164.64
Total:	473.75	907.44	566.73	1947.93

Table 4.5 presents the estimated wastes entering local landfills however, it is difficult to understand from the report as to how these quantities were obtained. Many landfills relied upon construction waste as either a supply of cover material

or, where demolition materials were concerned, a supply of materials for temporary road construction. Whilst such statements are accepted as being factual, little data to support the quantities landfilled are available.

Table 4.5: Local authority MSW data for 1966/67 taken from 1051 questionnaires. Data reproduced from Appendix N, Table1 Department of the Environment (1971 p.177).

	Industrial and Construction wastes (tonnes '000)		
	Excavation and demolition	Other types	Total
County Boroughs	1,476.76	153.61	1,630.37
London Councils	443.68	38.53	482.21
Boroughs	567.21	29.18	596.39
Urban Districts	778.72	58.64	837.36
Rural Districts	127.75	10.89	138.64
Total:	3,394.12	290.85	3,684.97

The literature review identified a 1995 project to estimate the current quantity of industrial and commercial wastes (including construction) flowing into English landfills (Leach et al., 1995). The authors accepted a succession of issues were likely to complicate whatever approach was adopted however, they persevered to compile disparate pieces of data (Leach et al., 1995 p.1) and concluded a series of waste outputs for different industrial classes for the UK. From these outputs 2 estimates were obtained for the total annual waste landfilled: i) 73,218,689 tonnes and ii) 83,267,102 tonnes. A further study by Warren Spring Laboratory (1993) reported 102 million tonnes of waste was being landfilled (CSERGE, Warren Spring Laboratory and EFTEL, 1993). This estimate was prepared from a number of sources principally the *Digest of Protection and Water Statistics* published annually by the Department of the Environment and other research by Warren Spring Laboratory, none of which is currently available. The authors caveat their work by referencing the large uncertainties associated with waste statistics generally and the requirement to convert waste volumes into weights.

From either of these publications, what cannot be established is into which landfills these materials were deposited. However, industrial landfills tended to be owned and operated by the waste producer (Department of the Environment, 1971). This also simplifies, to some point, the likely content of municipal landfills. To identify flows of industrial and commercial waste into MSW landfills would require specific knowledge of individual producers. This may be available at the local level but is beyond the scope of this study.

4.7 Salvage and reclamation

Dawes (1947) proposed the wartime salvage policy should be continued following termination of hostilities. For a limited period, this was the case but more out of necessity due to shortages than the implementation of a dedicated policy. Higginson (1960) proposed waste management practices that were only implemented in the new millennium. He commented in a presentation to the Institute of Public Cleansing in 1960:

> "*Can municipal salvage be conducted on more scientific lines with the aim of providing recovered materials to appropriate industries?*"

Post-war salvage was undertaken but at a cost to many of the local authorities that continued the practice. Data is very limited other than for wastepaper which, along with metals, were the primary targets. Table 4.6 identifies reported levels of salvage and recycling undertaken between 1966/67 – 1986/87. By 1966/67 material salvage was declining for both practical and financial reasons (Stokes et al., 2013).

Table 4.6: Summary of recorded annual salvage/recycled materials 1966/67 to 1986/87

	Glass (tonnes)	PPC[1] (tonnes)	Metals (tonnes)	Plastics (tonnes)	Textiles (tonnes)	Composting (tonnes)	Other (tonnes)
1966/67[2]	-	224,390	55,304	-	4,776	41,791	27,246
1974/75[3]	-	41,742	60,438	-	-	20,693	4,232
1976/77[3]	-	8,431	67,519	-	-	14,953	953
1979/80[4]	6,017	120,813	4,360	-	-	-	1,215
1984/85[4]	62,458	38,644	4,369	-	-	-	136
1986/87[4]	68,761	29,382	21,419	-	-	-	688

Notes: 1) PPC is paper, packaging and card. Data sources: 2) Department of the Environment (1971). 3) Department of the Environment (1978). 4) CIPFA.

Concern with regard to the recyclable content lost in MSW streams began to emerge during the mid-1970s. Bailly and Tayart de Borms (1977) reported to the European Economic Community of the lost value by disposing of wastes which equated to more than 400 million dollars for the (then) seven member states. Within the UK, waste management plans identified the 'desire' to recycle, but the costs of separation prevented its uptake. The Trade and Industry Committee (1984) published an evidence base report entitled *The Wealth of Waste* which identified the annual generation of 56 million tonnes of solid waste by industry, commerce and households. Only 15 million tonnes are reclaimed or recycled with large amounts of valuable material lost (Trade and Industry Committee, 1984). One statistic cited in the report is that only 6% of glass was collected for reuse. Despite a 1990 UK Government target to recycle 25% of waste by 2000 (Parfitt and Flowerdew, 1997) it was not until the implementation of the 1999 Landfill Directive that recycling became a serious alternative (Table 4.7).

Table 4.7: Summary of recorded annual salvage/recycled materials 1991/92 to 2006/07

	1991/92[2]	1996/97[3]	2000/01[3]	2006/07[3]
Glass (tonnes)	152,266	306,000	397,000	839,720
PPC[1] (tonnes)	150,527	555,000	909,000	1,535,000
Metals (tonnes)	60,471	60,471	217,000	337,000
Plastics (tonnes)	2,922	5,000	13,000	49,300
Textiles (tonnes)	7,418	7,418	30,000	41,000
Composting (tonnes)	9,817	9,817	278,000	940,000
Other (tonnes)	19,895	-	205,000	1,120,790

Notes: 1) PPC is paper, packaging and card. Data sources: 2) CIPFA. 3) Department for Environment, Food and Rural Affairs' Waste Surveys for respective years.

4.8 Discussion

4.8.1 Post war salvage and material consistency

The data associated with each of the four epochs is both relevant and important. Two issues are immediately pertinent. Firstly, is the consistency in the waste stream of the identified 'value' materials (Figure 4.11) that is the combustible content, the glass and the metals deposited into landfills from the household MSW stream. Added to the combustible element from the mid-1960s was the introduction of plastic. Their non-appearance in Figure 4.11 until 1970 results from sampling exercises classifying plastics into other categories, either combustible, non-combustible or unclassified. Higginson (1960) in a conference presentation, identified the presence of waste plastic as a looming problem particularly in its disposal to landfill.

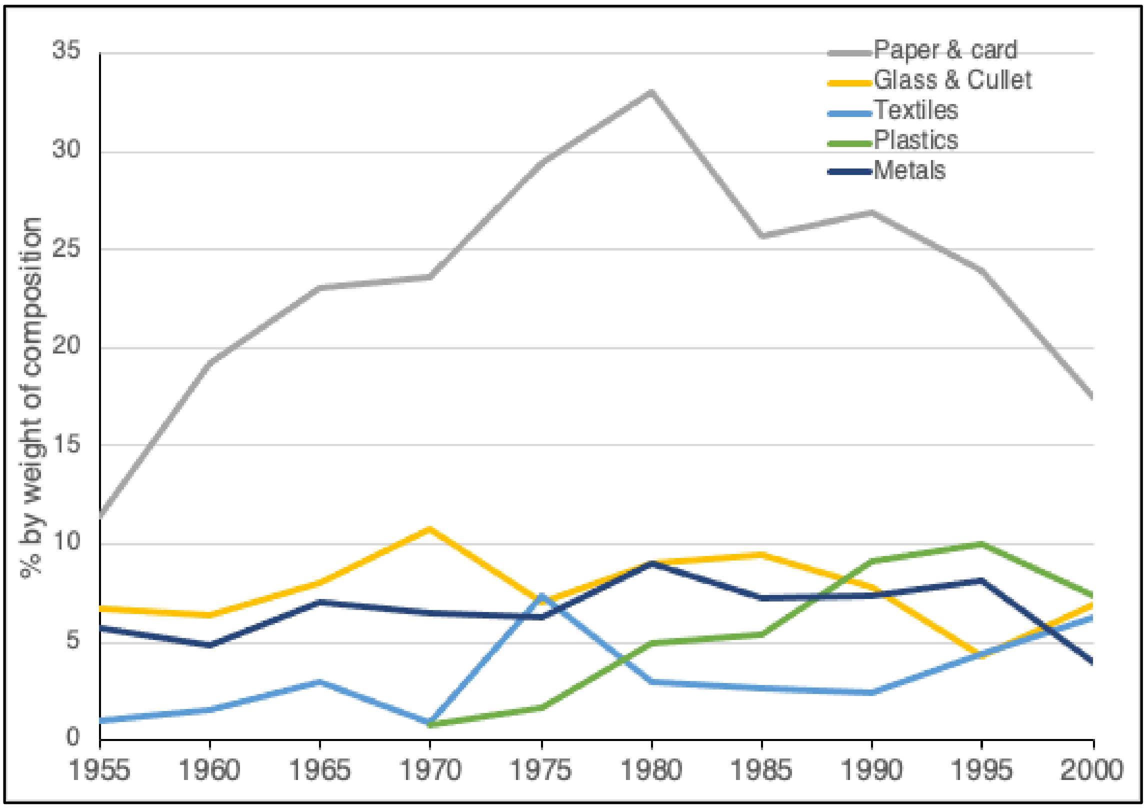


Figure 4.11: Percentages of value materials in the household MSW stream 1955 – 2000. Plastic was not categorised separately; however, it is recorded from 1965 by Higginson and 1966/67 by the Greater London Council.

Secondly, is the impact of material recovery on flows of these value materials into landfills. In the aftermath of the World War II the key targets for material recovery were wastepaper and metals (Higginson, 1960; Flintoff and Millard, 1969). Since 2000 this range has increased. Material recovery presents two problems: firstly, removing value materials from the MSW stream will devalue expected realisations that may result from any proposed landfill mining project where material recovery occurred. Any estimation methodology is required to include a proportionate reduction for the sake of accuracy. Secondly, although 81% of household MSW was being landfilled in 1999/00 (Department for Environment, Food and Rural Affairs, 2001), the availability of recycling data since that point provides a workable starting point from which to identify when and how much specific local authorities recycling would impact on materials flowing into landfills. This is particular useful in the analysis of individual landfills because local authorities, as waste collectors, are legally required to report their recycling annually.

For the most part, salvage or reclamation had little impact on the content of final MSW entering landfills. The marketability of waste newspapers and metals was long-standing and continued to be so thus providing an incentive for local authorities to salvage materials. Recovery of wastepaper from household collections equated to some 0.218 million tonnes in 1945, rising to 0.46 million tonnes in 1955 then reducing to .399 million tonnes in 1959 (Higginson, 1960). Stokes et al. (2013) cite recycled wastepaper from all waste streams within the UK to extend from 1 million tonnes in 1950 to 2 million tonnes in 1970 against a supply exceeding 3 million tonnes (1950) to 7 million tonnes in 1970. However, by the mid-1960s the volatility of markets and the costs associated with separating materials from refuse led to the gradual abandonment of material recovery (Ministry of Housing, 1967; Department of the Environment, 1971; Wilson, 1981; Stokes et al., 2013).

4.8.2 Missing MSW composition data

This section identifies how missing composition data is substituted with the strategy then applied to other missing data. The adopted approach is the use of MSW composition data taken from a year or period where the data are structurally similar, or the socio-economic backgrounds driving MSW composition are similar. Two periods where data is limited are 1982 – 1992 and 1994 - 1998. The 4-year period from 1994 to 1998 presents only one sample. From the start of this period is an example where the socio-economic background was building to one of stable growth and relative prosperity. The single study, made in 1996/97 and undertaken by the University of East Anglia, Figure 4.12 (a), is compared to the Resource Futures study (2006/07). For each MSW category the recorded difference is less than 5% of the total composition. Glass, metals and textiles differ by 0.8 to 1.2% of the total composition.

The earlier 10-year period (1982 – 1992) has different sampling profiles also the socio-economic backgrounds across the period vary quite significantly from the super-recession at its beginning, the Thatcher Boom of the mid-1980s and the boom's subsequent collapse in 1988/89. Organic MSW together with paper and card MSW differ with both displaying independent peaks. For organic and paper and card, Figures 4.5 and Figure 4.7 demonstrate a longer-term trend where organic MSW is increasing with paper and card reducing. For the shorter-term it is clear Wilson's (1981) comments are justified but for only 2 material types. For the value materials the data maintain a similar consistency. Plastic waste displays an increasing trend, which, in combination with Figure 12 (a), exceeds 14% of the total composition by 2006/07.

Filling data gaps is problematic however, where there are observable trends, these can be incorporated over the shorter term. To undertake detailed retrospective interpolations or extrapolations requires an understanding of both historical consumer preference and product availability together with an appreciation of prevailing historical socio-economic factors. Both can be obtained from the UK Government's retail price index publications which are available from 1947. However, such an analysis is beyond the scope of this study.

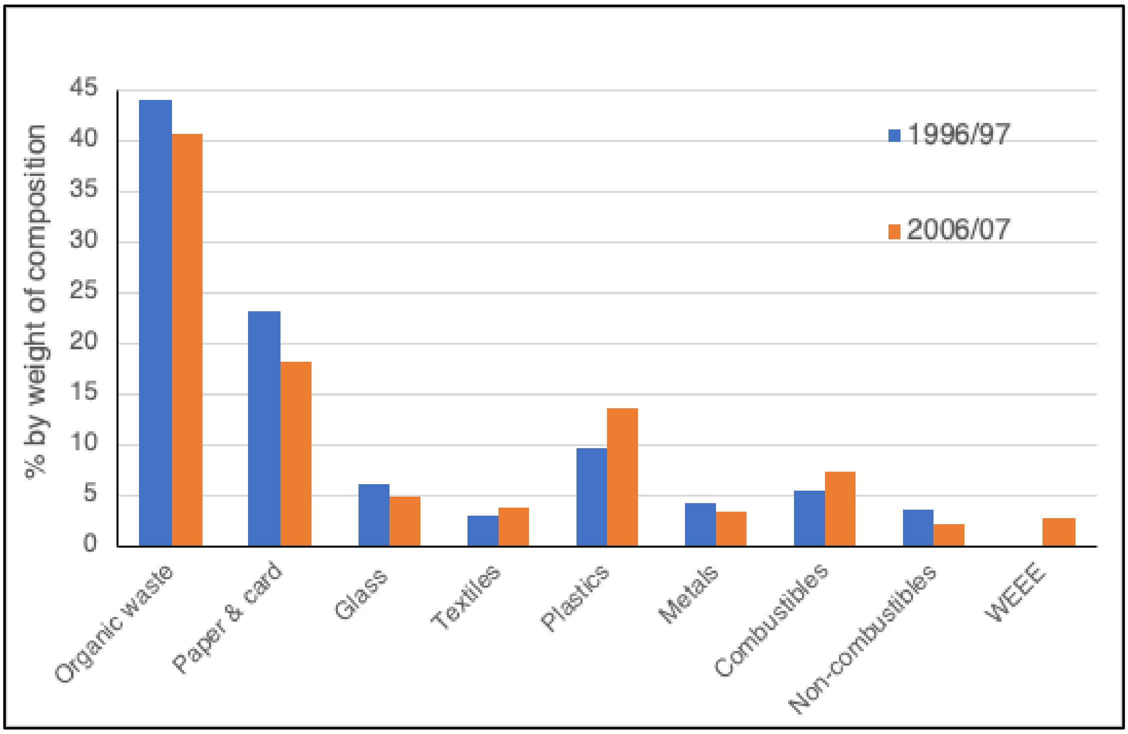


Figure 4.12 (a): Comparison of the University of East Anglia MSW composition study (1996/97) with the 2006/07 Resource Futures study. The combustibles category for the 2006/07 study includes disposable sanitary MSW and wood.

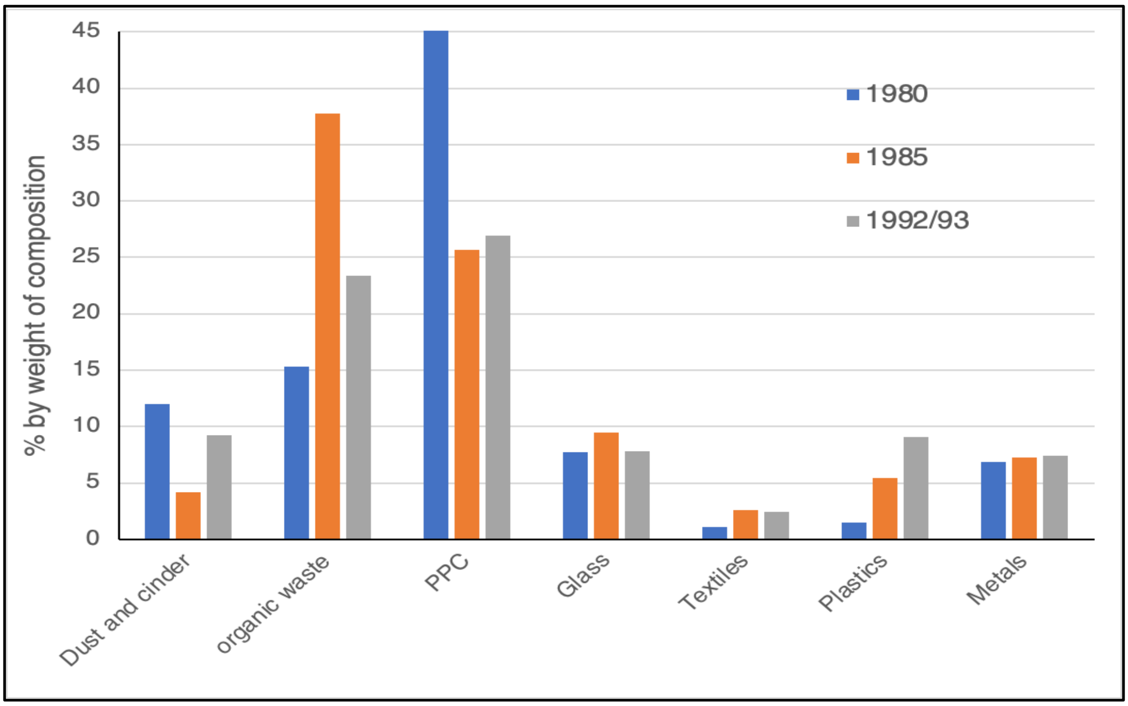


Figure 4.12 (b): Comparison of 1980, 1985 and 1992/93 MSW. The 1980 and 1985 studies were undertaken in Birmingham. The 1992/93 study is the National household waste study.

4.8.3 Material flow into landfills

Prior to the late 1990s, the Department for Environment, Food and Rural Affairs (2000) identified 75% of recycling in England resulted from civic amenity sites and not kerbside collections. Data released by the Department for Environment, Food and Rural Affairs (2010) is presented in Figure 4.13.

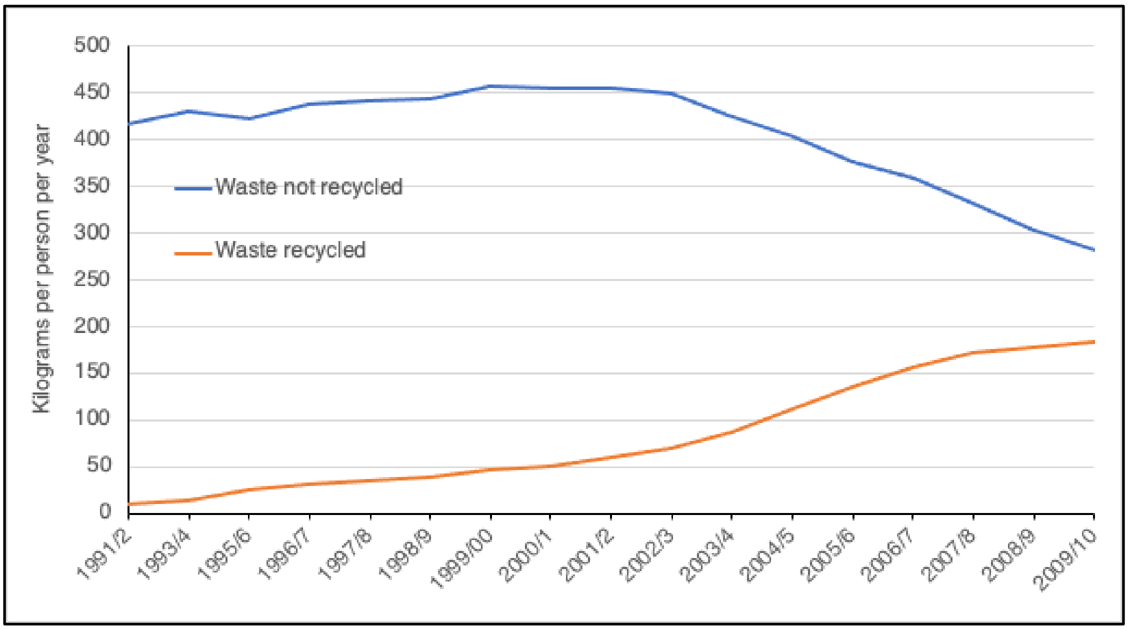


Figure 4.13: MSW generated per person per year 1991/2 – 2009/10 (Department for Environment, Food and Rural Affairs, 2010).

Individual waste generation hit its zenith over the period 1999 – 2003. Until this point the average quantity of waste recycled per person annually was below 50 kg therefore any reduction of value materials from the household waste stream will be limited to the period 1999/00 – 2006/07. In England, material recovery, salvage or recycling had little impact upon material flows to landfills from the mid-late 1960s until 2002/03.

In addition, the Department for Environment, Food and Rural Affairs (2000) reports the South East and South West achieving 12% and 13% recycling with remaining regions achieving, on average, 5% of 21.5 million tonnes of household waste generated. These inconsistencies present a problem at both the regional and local authority level. However, there are waste data available from the late 1990s and these data become more comprehensive leading to 2006. After 2006 WASTDATAFLOW provides the information necessary such that this study can use that date to terminate its analysis.

The implementation of recycling of MSW was directed not only at material recovery but also the diversion of biodegradable materials from landfills as routes to increasing landfill reaction kinetics (Stegmann, 1983). Research had established the detrimental impact of rapidly degrading organic waste (food, kitchen and garden waste) on the landfill biochemical environment. The impact of this decision to divert these rapidly degrading organic wastes away from landfills is developed in Chapter 6.

4.9 Conclusion

Three primary factors control the materials flowing into landfills hence their future resource potential. The first is dictated by economics that is the cost of landfill disposal when compared to other disposal options. Second, is the composition of material components contained within the MSW stream and third, the regulatory controls in place at any one time.

Chapters 1 and 3 have identified landfilling to be favourable, economically, which is further reinforced by the recorded data that approximately 90% of MSW was landfilled. To accommodate both the variations and the developing regulatory framework, 4 landfill epochs are proposed in which material flowing into landfills are dictated by a series of primary and secondary determinants. For the first 2 epochs waste composition is the primary determinant where direct regulatory controls have little or no impact other than to determine the location of landfill sites (Environmental Protection UK, 2012). For the third epoch, compositional variations in the waste stream occurred but regulation started to determine landfill operation and which wastes were acceptable at the point of disposal. By the fourth epoch, regulation required the diversion of specific biodegradable wastes away from landfills and the implementation of material recovery.

MSW composition sampling is an unpleasant and expensive process that lacked an accepted standardised methodology (Higginson, 1966; Parfitt and Flowerdew, 1997; Dahlén and Lagerkvist, 2008). However, MSW composition sampling provides a baseline for 13 material categories that constitute the content of historical landfills. Along with MSW generation data, these provide a series of sources from which to estimate the types of materials and the quantities flowing into landfills, hence their potential resource value.

This study acknowledges the available data are not complete, However, i) data substitution offers an opportunity to fill gaps where composition sampling is limited or was not undertaken and ii) for a more complete picture of landfill content, there is the likelihood a year by study of consumer purchasing preferences and product availability will add to this study’s findings. This latter point is included as an area for further research.

Chapter 5: Modelling Landfill Content

5.1 Introduction

Chapter 1 described the near complete reliance on landfills by English local authorities as they became the cornerstone of a developing and integrated waste management system. In addition, the chapter advanced the basic concept of a system and introduced: i) the computerised modelling technique known as system dynamics and ii) a framework for modelling the hierarchical structure of MSW management in England where this structure comprises 3 sub-units: a) MSW generation, b) available disposal strategies and c) MSW composition.

To this point, the system dynamics sketches accompanying Chapters 1, 2 and 3 are abridged illustrations of the 3 sub-units of the waste management system represented by a series of flows, each connected to a stock. A system dynamics model is a map of flows to and or from different stocks (Sterman, 2000). For this study, flows are the stream of materials, materials being the components of MSW, that accumulate as stocks in one of four disposal options, with landfill being the largest by a considerable margin for the time frame of this study. Incineration and salvage/reclamation, although notably smaller, are nevertheless included as each has an impact. Incineration for the generation of residual ash and salvage/recycling for the diversion of value materials out of the MSW stream likely going to landfill.

Reducing the waste management system into 3 sub-units suggests that what is essentially a complex structure can be simplified. However, such disaggregation is common in formulating models across many disciplines and particularly in systems analysis (Sterman, 2000). This is not to suggest any of the sub-units can be solved more easily in isolation, it is that each sub-unit can be analysed or constructed independently. System dynamics modelling is often approached in this way so as to allow objective and or separate scrutinization of structure and variables or to allow collaboration between specialists across different disciplines (Meadows, 2008; Garcia, 2020).

This chapter proposes a developed system dynamics model to determine material flows into landfills. The model is presented as 2 stages with the first stage, identified as Tier 1, combining MSW generation and disposal data to estimate the quantity of MSW generated from 1945 to 2007 and the quantity disposed into landfills. The second stage, Tier 2 introduces MSW composition data to categorize and quantify the materials within a landfill mass. As a route to realizing this, the pertinent sections in this chapter are structured as follows:

- Section 5.3 provides a brief background to the system dynamics method (Section 5.3.1), how it can be applied to a waste system (Section 5.3.2) and how interpolation forms the basis for estimating surrogate data (Section 5.3.3).
- Sections 5.4, 5.5 and 5.6 describe each sub-unit of the proposed system dynamics model and each sub-unit's output. The MSW generation (Section 5.4) and disposal (Section 5.5) sub-units combine to form a basic, Tier 1 model which when added to the landfill content sub-unit forms a complex Tier 2 model (Section 5.6).
- Sections 5.71 and 5.72 introduce and consider methods for verification and validation of the proposed system dynamics model. For verification (Section 5.7.3) the models output is compared to the results of a truncated mass balance. To validate the model's output (Sections 5.7.4 – 5.7.6) comparisons are made with sampled or excavated landfills in the UK and Europe. Section 5.7.7 reviews the model's output in the context of a conceptual "standardised landfill" expounded by van Vossen and Prent (2011) resulting from the study of 60 landfill mining projects.
- A final section (Section 5.8) presents a series of radar graphs to identify the percentage content of resources contained in English landfills through the 4 epochs defined by this study.

In system dynamics the variables, particularly their values, determine the credibility of the model and, more generally, a system dynamics model's ability to mimic the real-world situation (Sterman, 2000). Variable inputs, can be derived mathematically, supplied from recorded data or as in this case, a fusion of both. At some point, a purely mathematical approach necessitates the inclusion of aspects of human behaviour. Kennedy (2012) comments:

> "*The modelling of human behaviour is not at all obvious. First, humans are not random. Second, humans are diverse in their knowledge and*

abilities. Third, besides being controlled by rational decision-making, human behaviour is also emotional." (Kennedy, 2011 p. 167)

Allowing human behaviour to be reflected by the recorded data may prove less problematic however, these data are incomplete, and some use of mathematics for estimation is necessary.

5.2 Relevance to the aims and the objectives

Research objective 4 proposes the development of a model to identify the materials or resource potential within landfilled wastes. This chapter presents a system dynamics model in an original approach to improve identification of landfill content and achieve this objective. The chapter will consider the first two research questions: i) can historical and geographic data provide a means of predicting landfill mass content? And ii) do significant connections between sampling data and MSW content exist?

5.3 Modelling Method

5.3.1 System dynamics: definition and overview

Initially, system dynamics was considered a tool for "top management problems" by the discipline's acknowledged creator J W Forrester (Forrester 1961, cited in Sterman 2000, p.41). However, the application of system dynamics to numerous academic fields has led the methodology to be classed as a mathematical modelling technique to enable understanding of non-linear, complex behaviours.

The Sloan School of Management within the Massachusetts Institute of Technology [MIT] describe system dynamics as helping to understand, design and manage change by modelling all parts of a system and how the relationship between those parts influences that system's dynamic behaviour (MIT, 2021). The System Dynamics Society submits that system dynamics unites social and behavioural sciences with the fundamental detail of planning and accounting (System Dynamics Society, 2021). Dyson and Chang (2005) identified system dynamics as a methodology to examine interrelationships between socio-economic, environmental and managerial factors where data scarcity creates problems for both planners and modellers alike. The basic framework of system analysis requires an understanding of the system, construction of a time-dependent model and a review of the simulated results (Close and Frederick, 1995).

5.3.2 The waste system as a series of stocks and flows

Resources contained in landfills occur as a result of the materials discarded by households and commercial enterprise as MSW. Representing the hierarchical structure of MSW management as a series of stocks and flows permits a wide range of modelling opportunities which can be analysed across many levels: i) at the individual level whether this is from the perspective of the waste generator, a single landfill, a local authority, or a settlement which can range from a small town to larger conurbation, ii) a defined group or cluster of individuals and iii) a national inventory. To add to these combinations, system dynamics allows each analysis to be matched against a specific time frame or over the entire study period.

The following sections present the proposed model as a series of stages by building from the addition of each sub-unit. At each stage, the model will be calibrated using a national scale, that is for England with the resulting output included within that section.

5.3.3 Estimating missing data

The modelling in this chapter considers the entire 63-year time-period from 1945 to 2007. In addition, the proposed system dynamics model contains 25 variables which are sourced from published, historical data. For some years, these data are incomplete and require estimation. During this period, many variations occurred in the composition of MSW together with extraneous factors which combined to impact not only on the composition of waste flows entering disposal options but on the disposal options themselves. To better understand how these variations and extraneous factors interacted, this study has proposed 4 periods or epochs (Chapter 1, Section 4.2) which structure the prevailing socio-economic and regulatory contexts and their concomitant impact on these material flows. As such, each epoch encompasses specific determinants that control the dynamics of waste generation and its management.

Waste composition studies are undertaken to understand the structure of waste and guide policy. Mid-1960s composition studies were considered useable data from which to predict future waste flows, that is inferring the unknown from the known or extrapolation. However, the use of extrapolation to predict waste flows has encountered problems. This was the case exemplified by the period 1970 – 1973 which saw a reversal in the expansion of paper and card in the MSW stream. Wilson (1981) identifies this phenomenon as a reason to avoid the 'pitfalls' of extrapolation.

The 4 epochs proposed in Chapter 1 and elaborated upon in Chapter 4 constrain estimations to remain within specific boundaries thus a basis for interpolation emerges. What results is a more stable functional time-series where discernible patterns and processes occur but are restricted within these boundaries. As such, extrapolation is avoided with the exception of the period immediately following the termination of World War II. For this period, little data exists and with post-war conditions extending into the 1950s (Zweiniger-Bargielowska, 2000; Marwick, 2003), extrapolation is reasonable.

For each epoch, missing data was substituted with the arithmetic mean derived from adjacent values. For even numbers of years, where these were equal to or exceeded 4, then the 2 mid-point values were repeated. Each section identifies where data was required to be estimated with the precise detail for MSW generation presented in Appendix 5.1, disposal in Appendix 5.2 and MSW composition in Appendix 5.3.

5.4 Modelling annual MSW generation

5.4.1 Introduction

In the following sections a system dynamics model is proposed and presented to estimate the quantity of materials or resource potential in landfills. The model is structured as two tiers: i) Tier 1, a basic structure which includes 2 sub-units and identifies the generation and flow of waste to the 4 disposal strategies adopted in England between 1945 and 2007 (Figure 5.1 and Section 5.5) and ii) Tier 2, a more complete structure which includes a disaggregation of the waste stream as a means to identifying the resource content in English landfills (Section 5.6). Data, for use as inputs for model variables, for respective sub-units has been introduced and reviewed in earlier Chapters. Chapter 3 (Section 3.6.1) defined three data sets of individual, daily MSW generation to estimate annual MSW generation which are identified as: i) Local Authority collected MSW [MSW_{coll}], ii) Local

Authority amended MSW [MSW_{amnd}] and iii) Local Authority all MSW [MSW_{all}]. Generation rates, taken from 14 randomly selected years across the study period, were reviewed as a histogram (Figure 3.3) and a series of boxplots (Appendix 3.5 (a – d). Data in respect of MSW disposal is similarly presented and reviewed in Chapter 3. Data for the Tier 2 model, MSW composition is presented and reviewed in Chapter 4.

5.4.2 Presentation of the proposed Tier 1 generation sub-unit

The proposed system dynamics sketch for the Tier 1 MSW generation sub-unit is presented in Figure 5.1. From this point the model graphics will encompass the usual Vensim® designation for a stock which is a rectangle and a flow identifiable as a thickened black headed arrow. The curved, narrow arrows are connectors and relate variables to constants, flows or stocks. The cloud-like shape to the left of the flow is termed “a cloud” and in each case, throughout this study, is an empty set with no impact on the model but is a necessary construct of a flow.

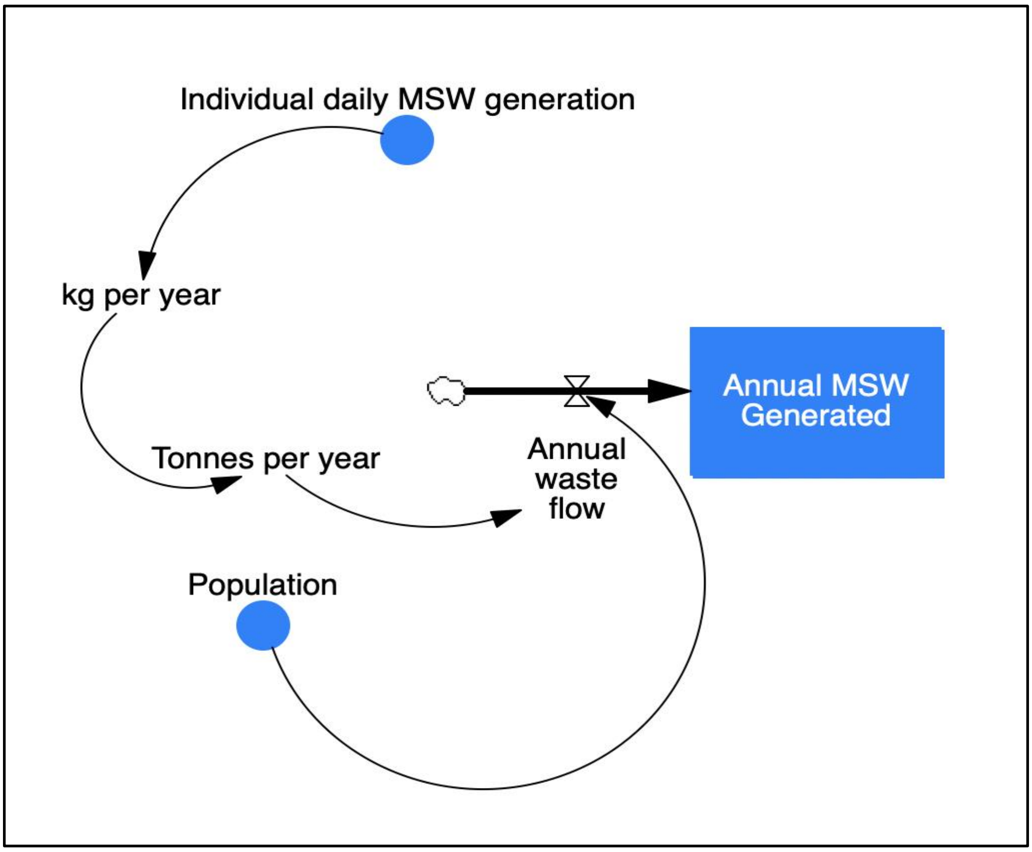


Figure 5.1: Waste generation sub-unit of the proposed Tier 1 system dynamics model. The model has 2 variables, population, and the rate of individual daily waste generation. which are extracted directly from published, historical data. The model calculates the annual MSW generated and the cumulative total, Figure 5.3 (a & b). The colouring is that used in previous chapters.

Adopted designations for this sub-unit include a circle which represents a variable and plain text representing a constant. System dynamics ‘sketches’ produced using Vensim® usually indicate variables and constants with plain text but this can make larger structures more difficult to comprehend. Colouring of the model’s structure follows that used in previous chapters.

Input data is directly uploaded into the model’s variables from a series of data matrices populated onto Excel spreadsheets. The Vensim® software is then able to generate a continuous plot of generated MSW over the study period.

5.4.3 Model framework - MSW generation

Annual generated MSW is calculated from the 3 input data sets: i) Local Authority collected MSW (MSW_{coll}), ii) Local Authority amended MSW (MSW_{amnd}) and iii) Local Authority all MSW (MSW_{all}) from separate simulations of the model. The software reports MSW generation for an individual year and a cumulative total extending over the study period. The general equation required to obtain the annual waste flow is:

$$MSW = \left({}^{365}/_{1000}\right)PG \qquad (5.1)$$

where MSW (tonnes/year) is a generic reference representing 3 series of results. P is the population and G the daily MSW generation rate sourced from each of the 3 input-data sets (kg/person/day). The conversion coefficient equates to 365 days per year and 1000 kg per tonne.

5.4.4 MSW generation – input data for model variables

Total MSW generation data is available for 37 years from the data set of 63 years with 26 years required to be estimated. These include: i) the previously identified 8-year period 1945 – 1952 where the estimated quantity of MSW was proposed by Wylie (1957) to be approximately 10 million tonnes, ii) the 4-year periods 1970 – 1973, 1981 – 1984, 1988 – 1990, iii) the 3-year period 1992 – 1994 and iv) the single years 1979 and 1986. These missing data were estimated by interpolation. However, this was constrained by the bounds of respective epochs (Chapter 1, Section 4.2) with further consideration given to the 3 and 4-year periods in particular the periods 1970 – 1973 and 1988 – 1990 where the composition of MSW was changing.

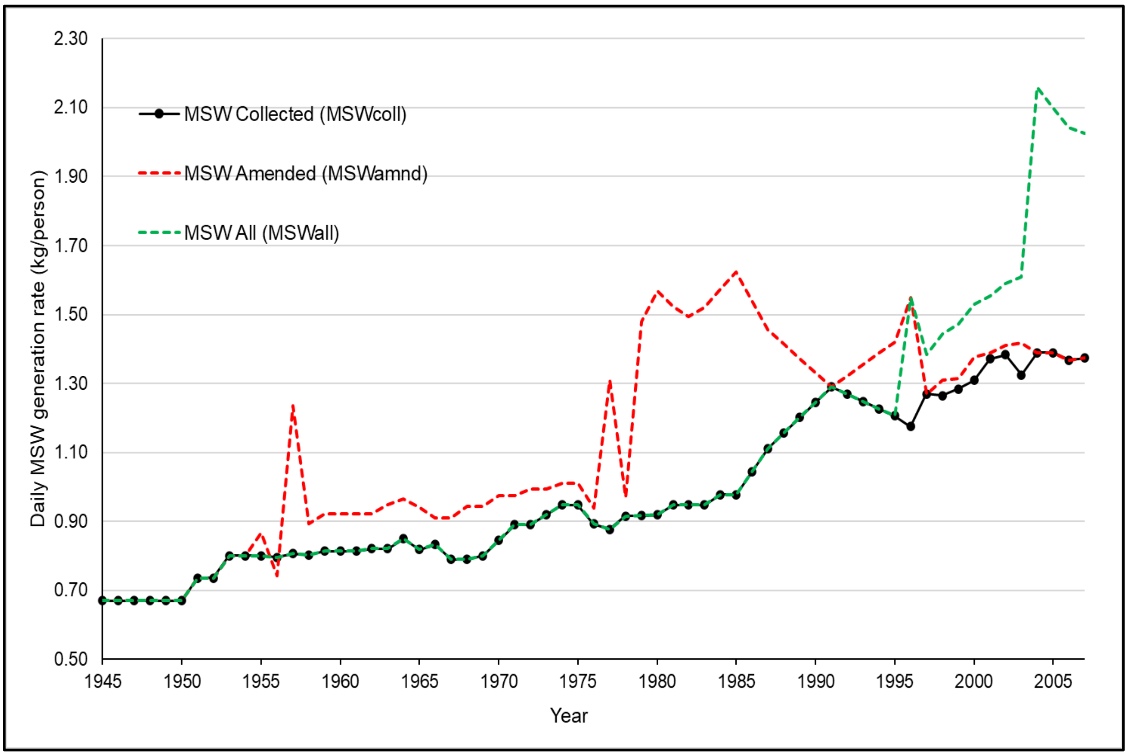


Figure 5.2 Graph of the input variable, individual daily MSW generation, obtained from recorded data. The combined black and green plots represent LA collected MSW with the green plots diverging, following the adoption of the expanded EU definition of MSW. The red plot reflects the amended data values derived from the increased generation rates quoted by different sources.

The full range of input-data used to generate the 3 model simulations is presented in Figure 5.2. MSW_{coll} is coloured black, MSW_{amnd} is coloured red and MSW_{all} coloured green. These colours are maintained throughout the remainder of this chapter.

5.4.5 MSW generation – model output

Chapter 3 utilised 3 input-data sets over 14 randomly selected years to estimate generated waste quantities, this first sub-unit of the model provides 2 sets of continuous data for the entire study period. Figures 5.3 and 5.4 present annual and cumulative MSW generation for each of the data sets. Whilst each data set in Figure 5.4 demonstrates the general pattern of increasing MSW generation over the study period, the all-data (red) plot is volatile and likely reflects the suggested enhanced individual waste generation rates reported to have occurred with MSW statistics and referenced by the Department of the Environment (1971) and Professor Coggins (2014). Statisticians within the Department of the Environment questioned the practice (Department of the Environment, 1978).

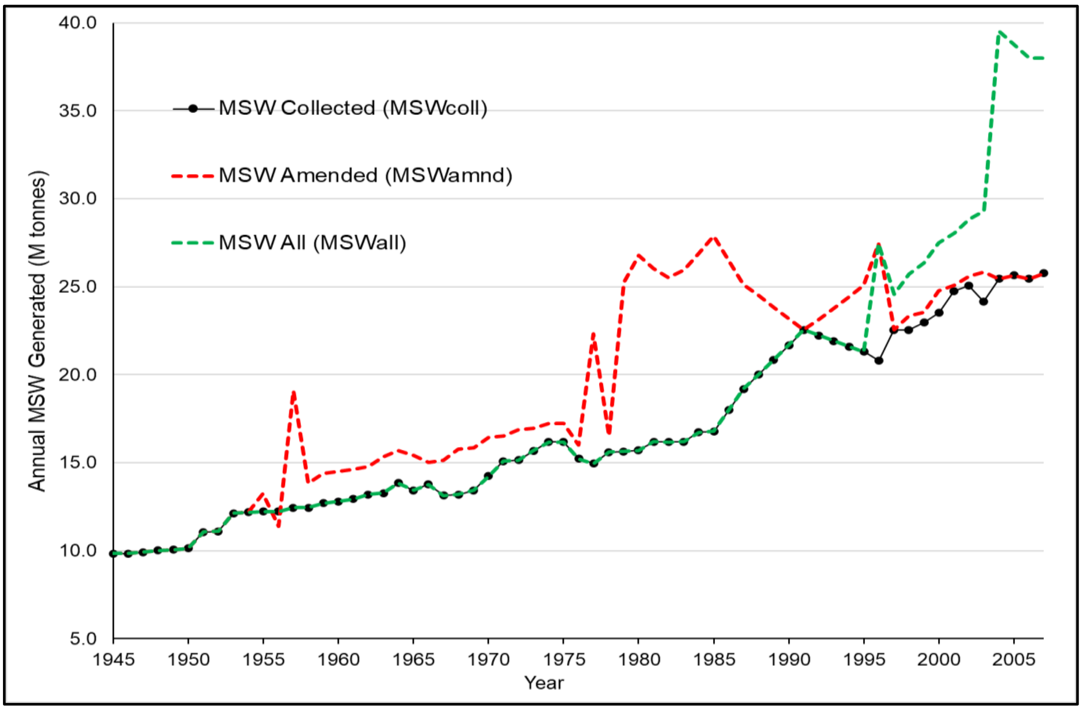


Figure 5.3: Annual MSW generated for England for the period 1945 – 2007. The combined black and green plots represent LA collected MSW with the green curve using the expanded EU definition of MSW by England. The red curve reflects the amended data values.

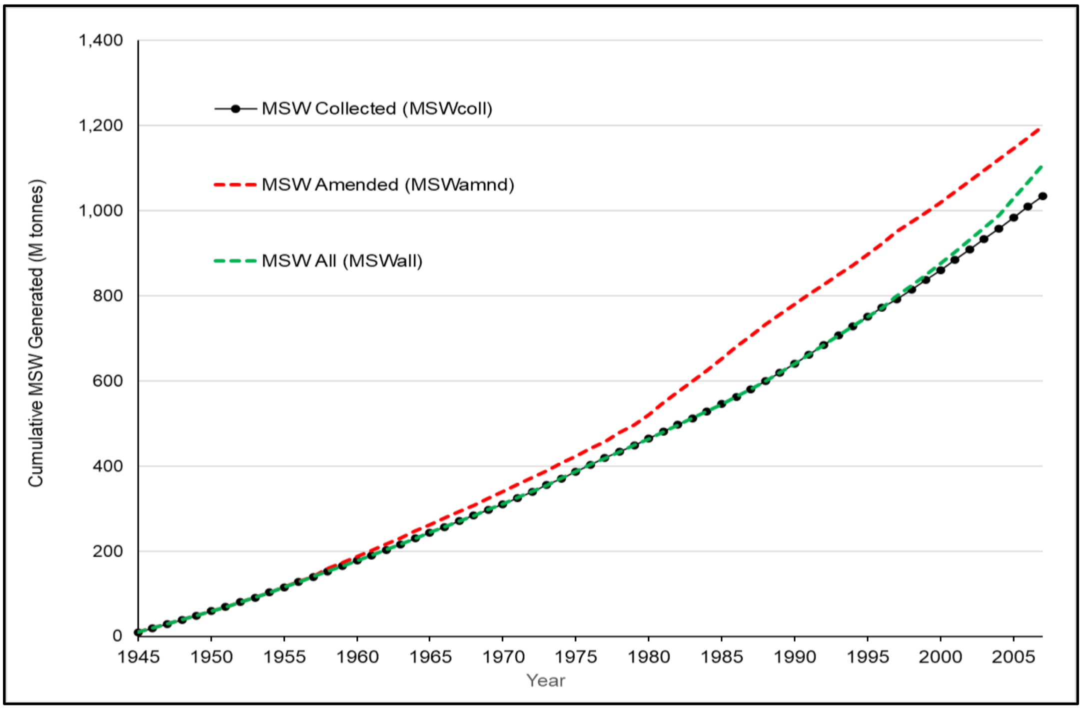


Figure 5.4: Cumulative MSW generated for England for the period 1945 – 2007. The combined black and green plots represent LA collected MSW with the green plot the adoption of the expanded EU definition of MSW by England. The red plot reflects the amended data values which generate an additional 70 million tonnes of MSW over the study period.

However, there are clear discrepancies marked by 3 rapid shorter-term increases and a sustained period of increase from 1980 through to 1986 which had reduced by 1990/91 to the levels reported by those local authorities weighing 50% or more of generated MSW to rise again.

Modelling at the regional or local level can be accomplished with data sets relevant to those locations. Furthermore, it is a relatively simple matter to amend or substitute the estimated waste generation rates with actual or real-time data if or when these become available. A future project may have the capability and resources to review historical paper sources archived at the UK National Archive. However, it was beyond the scope of this study.

5.5 Modelling MSW disposal

5.5.1 Introduction

For the greater part of the study period, landfilling dominated local authority disposal strategies. Chapter 3, Section 5.4 presented and reviewed waste disposal data where, from an anecdotal perspective, 90% of waste was assumed to have been disposed into landfill with 9% incinerated and 1% recycled or disposed by other means. To incorporate disposal options into the developed model necessitates greater detail as presented in Figure 5.5.

5.5.2 Presentation of the proposed Tier 1 model

MSW is distributed as a series of flows (thickened arrows) into respective disposal options or accumulated stocks (pink-coloured rectangles). Variables, identified by circles comprise recorded or estimated data. In addition, the angle-bracketed function (<annual MSW flow>) is a shadow variable and is a repeat of the same variable from the generation sub-unit. Shadow variables are incorporated into system dynamics models for compactness reducing the requirement for connectors which can result in clutter and confusion where they cross.

Flows into the 4 disposal options are determined by recorded or estimated data. Incineration also has an outflow into landfill where the residues of the incineration process are landfilled. The residual generation rate is taken as 35%, the average between the two generation rates for wet and dry input namely 40% and 30% respectively (Neal, 1979; Wilson, 1981).

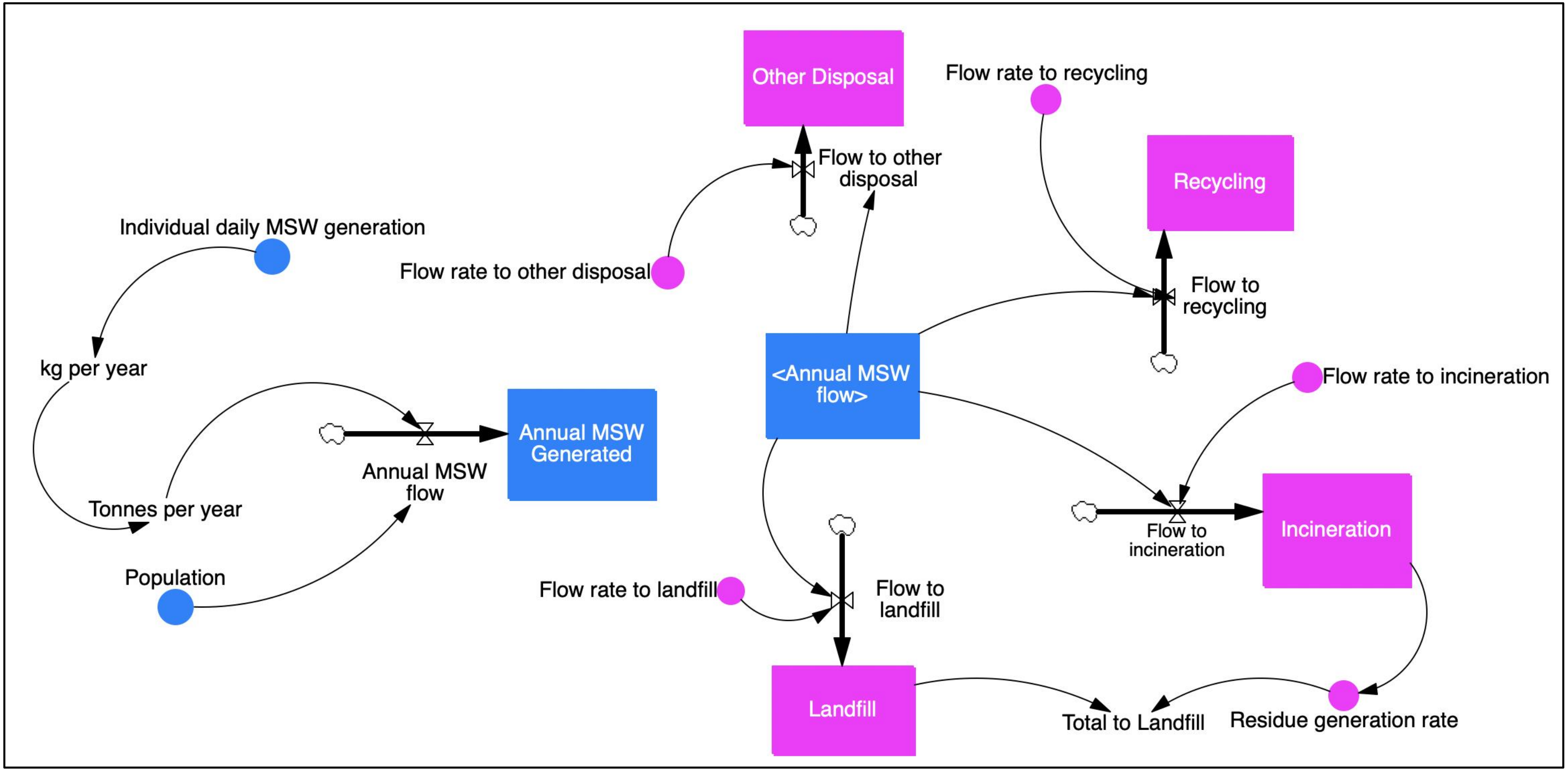


Figure 5.5: The proposed Tier 1 system dynamics model where waste generation and waste disposal sub-units combine to reflect the waste management system operated by English local authorities. Annual MSW flow is present in both sub-units but is a shadow variable in the disposal sub-unit. Shadow variables include the same data and their use avoids what can be confusing connectors. Flows into the 4 disposal options are determined by recorded or estimated data. Incineration also has an outflow into landfill where the residues of the incineration process are landfilled. The residual generation rate is taken as 35%, the average between the two generation rates for wet and dry input namely 40% and 30% respectively (Neal, 1979; Wilson, 1981).

5.5.3 Model framework - MSW disposal options

To obtain the annual waste flow to the four disposal strategies adopted in England the following general equation is used:

$$D_i = MSW_{annual} \times {}^{FR_i}/_{100} \quad (5.2)$$

where D_i (tonnes/year) is the annual quantity of MSW disposed to a particular disposal strategy, FR_i is the percentage rate of MSW flowing to a particular waste disposal strategy obtained from published or where necessary, estimated data.

From the general equation, equations for specific strategies are formulated for example for the flow to landfill:

$$D_{LF} = MSW_{annual} \times {}^{FR_{LF}}/_{100} \quad (5.3)$$

where D_{LF} (tonnes/year) is the flow of MSW to landfill and FR_{LF} is the percentage flow rate to landfill obtained from data. Similar equations can be written defining D_{inc} (tonnes/year) of MSW to incineration, D_{rec} (tonnes/year) MSW recycled and D_{oth} (tonnes/year) MSW flow to other methods.

5.5.4 MSW disposal – input data for model variables

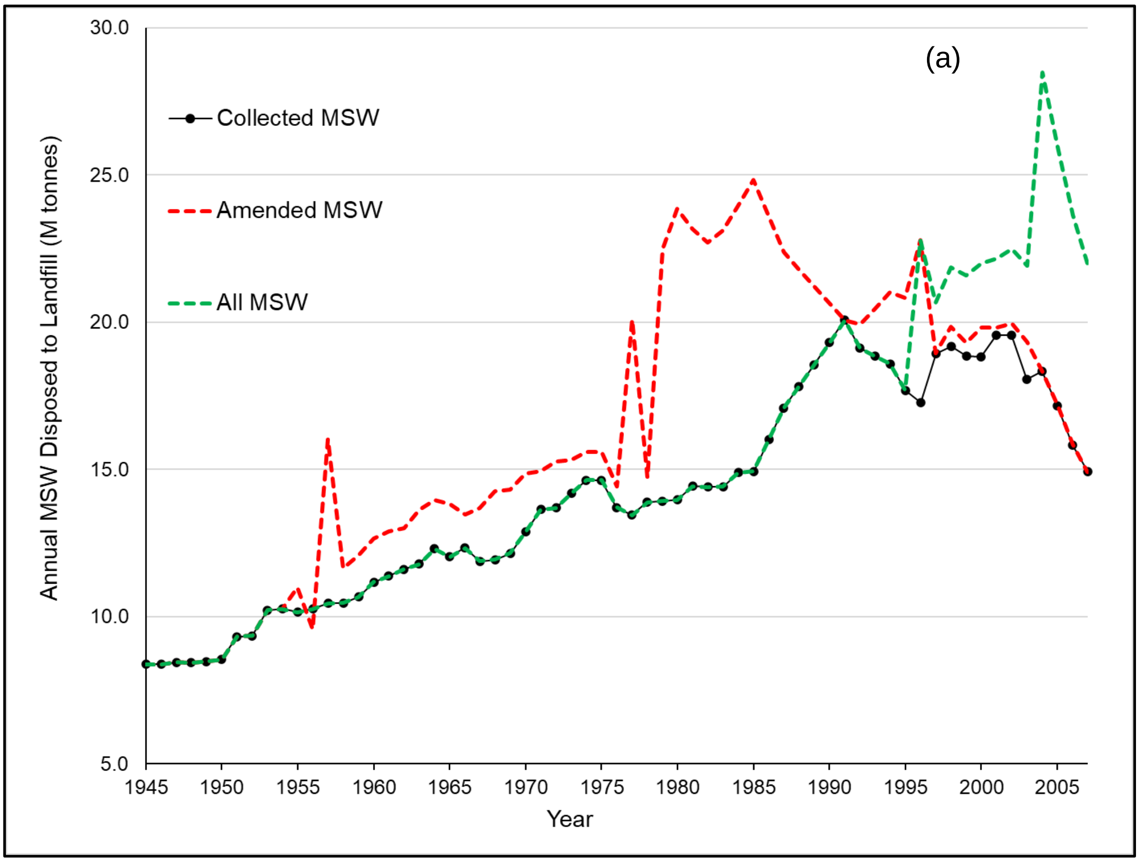
Published MSW disposal data was accessed for 25 of the 63 years of the study period with 38 years of the data set required to be estimated by interpolation. This is possible due to the consistency of adopted disposal methods. These include the 9-year period 1945 – 1953 where Section 3.5.4 identified approximately 15-16% of waste was disposed in alternatives to landfill. This was also the case for the period 1955 – 1959. 1954 and 1960 have specific percentages identified by the Ministry of Housing which reinforce these estimates. The Department of the Environment report (1971) identifies 90% of waste landfilled for the period 1967 – 1971 with 91% reducing to 89% for 1974 – 1978. These data arising from contemporary waste surveys.

For the period 1979 – 1994 CIPFA data provides sufficient detail to estimate that 89% of waste was landfilled with approximately 10% incinerated and less than 1% recycled. CIPFA did not offer complete data for this period. For the period between 1988/89 -1990/91 when no publications occurred, a general reduction from 89% reducing to 86% has been estimated. Publication resumed in 1991/92 and from 1995 was augmented by the Department for Environment, Transport and Regions and in 1997 by the Department for Environment, Food and Rural Affairs

5.5.5 MSW disposal – model output

Estimated waste disposal into landfill and incineration for England based on the proposed Tier 1 model are presented graphically in Figures 5.6 (a & b) and 5.7 (a & b). Figure 5.6 (a & b) represents the annual and cumulative MSW flow into landfill. Figure 5.7 (a & b) represents the annual and cumulative MSW flow into incineration.

Output from the model estimates 1.05 billion tonnes of MSW was generated in England during the study period, with 885.5 million tonnes landfilled. Waste diversion to recycling or other disposal methods are not presented as these flows were very small for the greater part of the study period. Furthermore, and from those sources available, data for the period extending from the mid-1960s to the mid-1990s are considered to be too limited to be statistically significant.



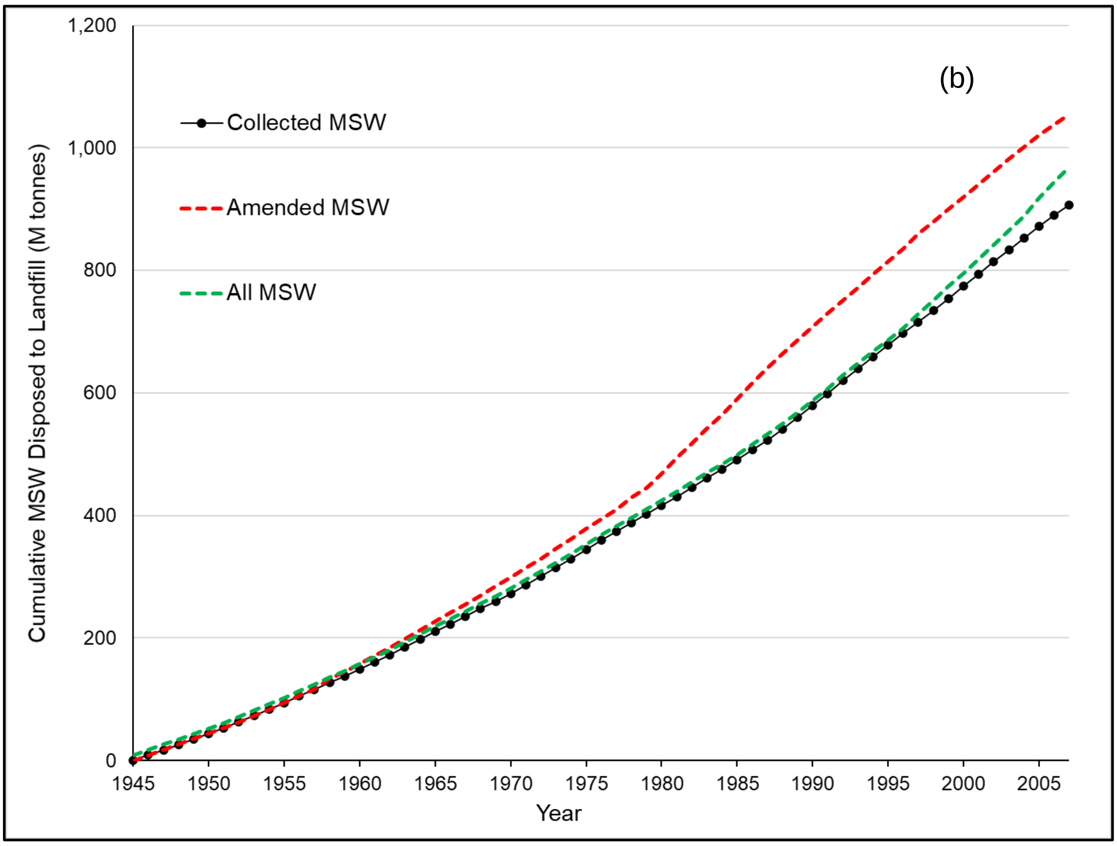
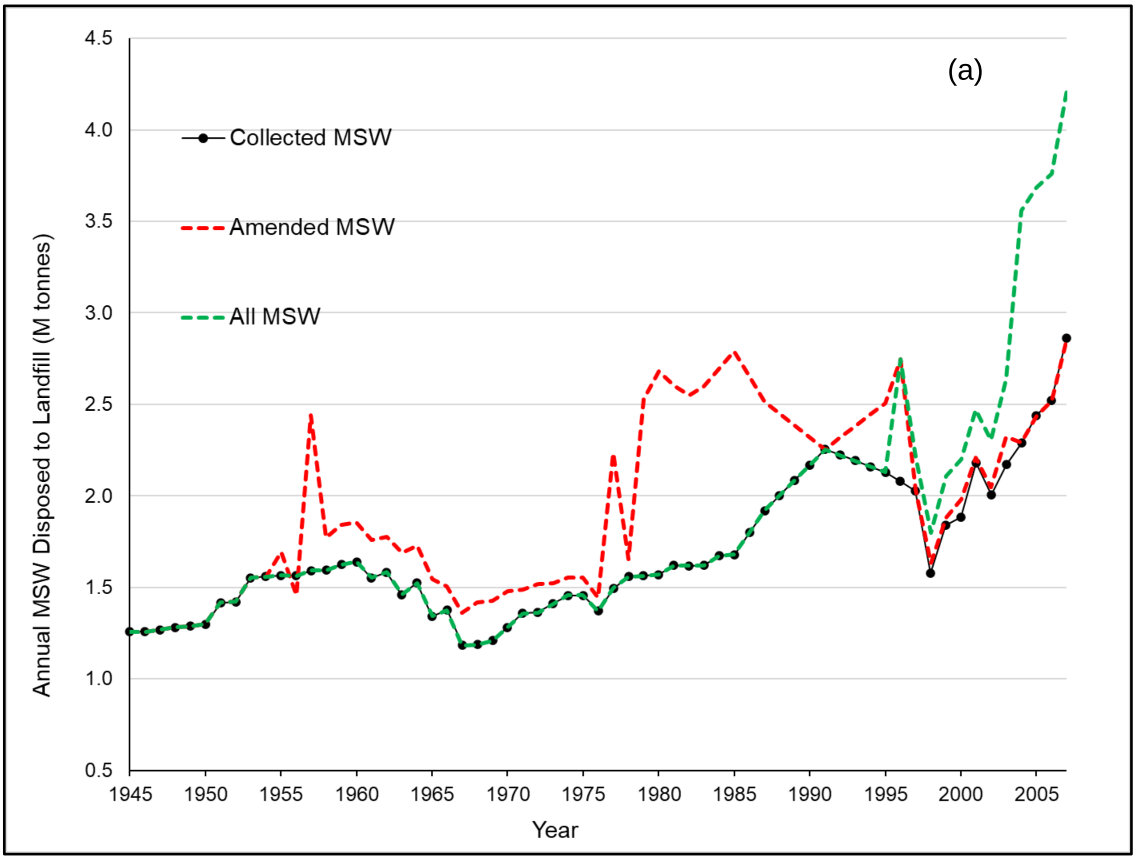


Figure 5.6 (a & b): a) Modelled annual % by weight of the flow of MSW to landfills in England 1945 - 2007. b) Cumulative flow of MSW to landfills during the same period.



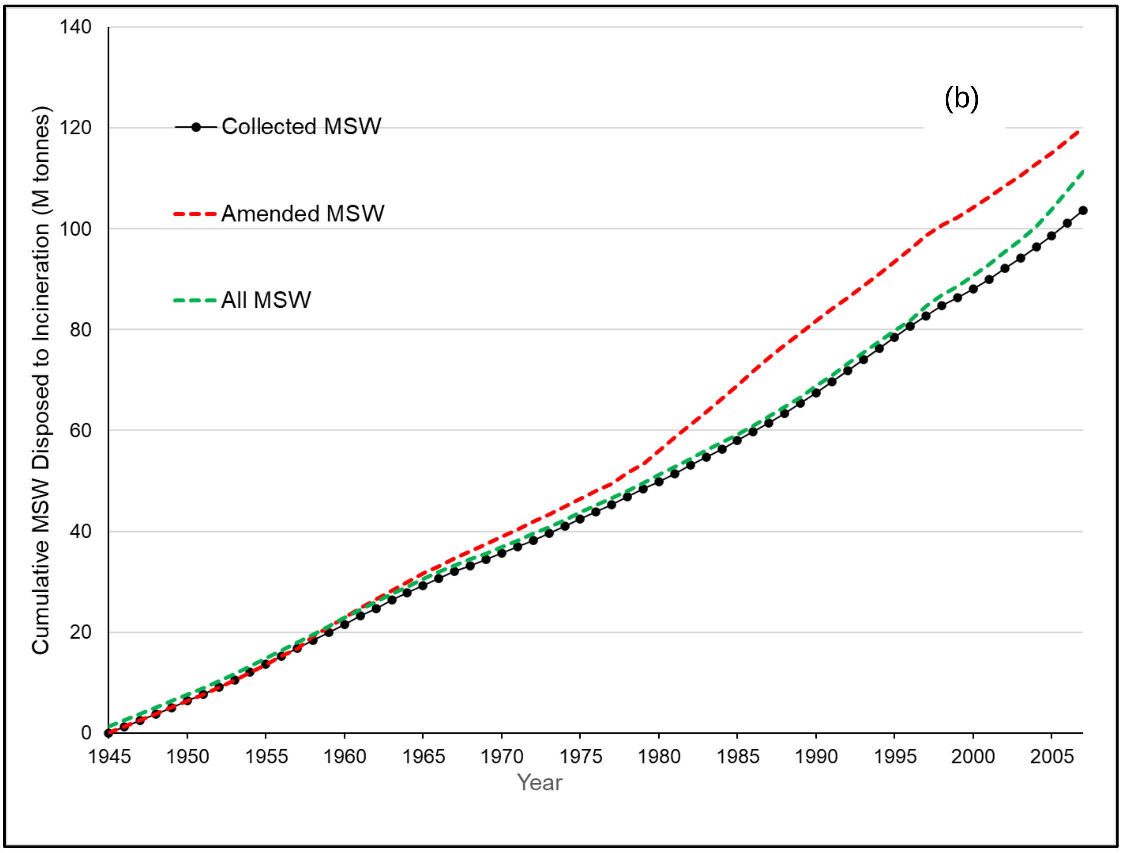


Figure 5.7 (a & b): a) Modelled annual percentage by weight of MSW to incineration in England. b) plots the cumulative MSW to incineration some 106.5 M tonnes.

Although incineration plays only a minor role at the national level, the process has an impact as a disposal mechanism in a limited number of conurbations along with necessitating inclusion of the process' residues.

5.5.6 Tier 1 model – summary of output

The modelled outputs presented in the previous sections combine published data and estimated values, where data is missing, to obtain a series of waste generation and disposal scenarios occurring for England. It is proposed that because of this study a realistic, data-based estimate for the quantity of MSW generated in England between 1945 and 2007 is now available. Furthermore, whilst this thesis presents the basic postulate by applying it to England, the model is equally applicable to different waste scenarios and generators whether these are towns, cities, countries, or individuals where actual or indicative data is available. It is expected that users of the proposed model will calibrate the model with data sets that reflect the specific locations being modelled. Such an approach is adopted in Section 5.7.

This study has identified and discussed the general opinion of published waste data prior to the inauguration of WASTDATAFLOW. The primary reason for inclusion of the amended data series is to identify the differences arising from using a mixture of weighed and unweighed waste along with the inclusion of outliers from data sets. This difference amounts to 70 million tonnes over the study period, that is a little over a million tonnes of additional waste each year.

5.6 Modelling landfill content

5.6.1 Introduction

Landfill content hence resource potential is determined by material flows into landfills with these forming the model's output. Chapter 4 presented MSW composition analyses undertaken and published since 1945. These analyses drive the variables forming the final sub-unit of this proposed Tier 2 model, presented as Figure 5.8. Post-war composition analyses focus on 10 – 12 major components. Latterly these have been extended with the inclusion of sanitary waste and WEEE. Flow rates for each MSW component are introduced into the model by grouping respective percentages over a specific time period and formulating a percentage range determined by the components low and high sampled values from which a mean value can be obtained. Material salvage/recycling, degradation of specific organic materials and daily soil covering are included within the Tier 2 model's structure.

5.6.2 Presentation of the proposed Tier 2 model

The structure presented in Figure 5.8 includes the MSW component and recycle variables (coloured red) together with 3 shadow variables: i) MSW to landfill, ii) flow to landfill and iii) incineration and 3 model parameters: i) cover material factor, ii) humus conversion factor and iii) residual generation rate, identified as triangles, and included as constant values. Assigning these as constant values facilitates Vensim's SyntheSim function where these constants become variables and allow the model's behaviour to be observed under changing conditions. This is particularly useful when considering likely decomposition rates for organic materials and the products arising (included here as the humus conversion factor) along with the daily soil-cover factor where uncertainty arises in respect of both the quantities of materials generated as humus or used as daily cover.

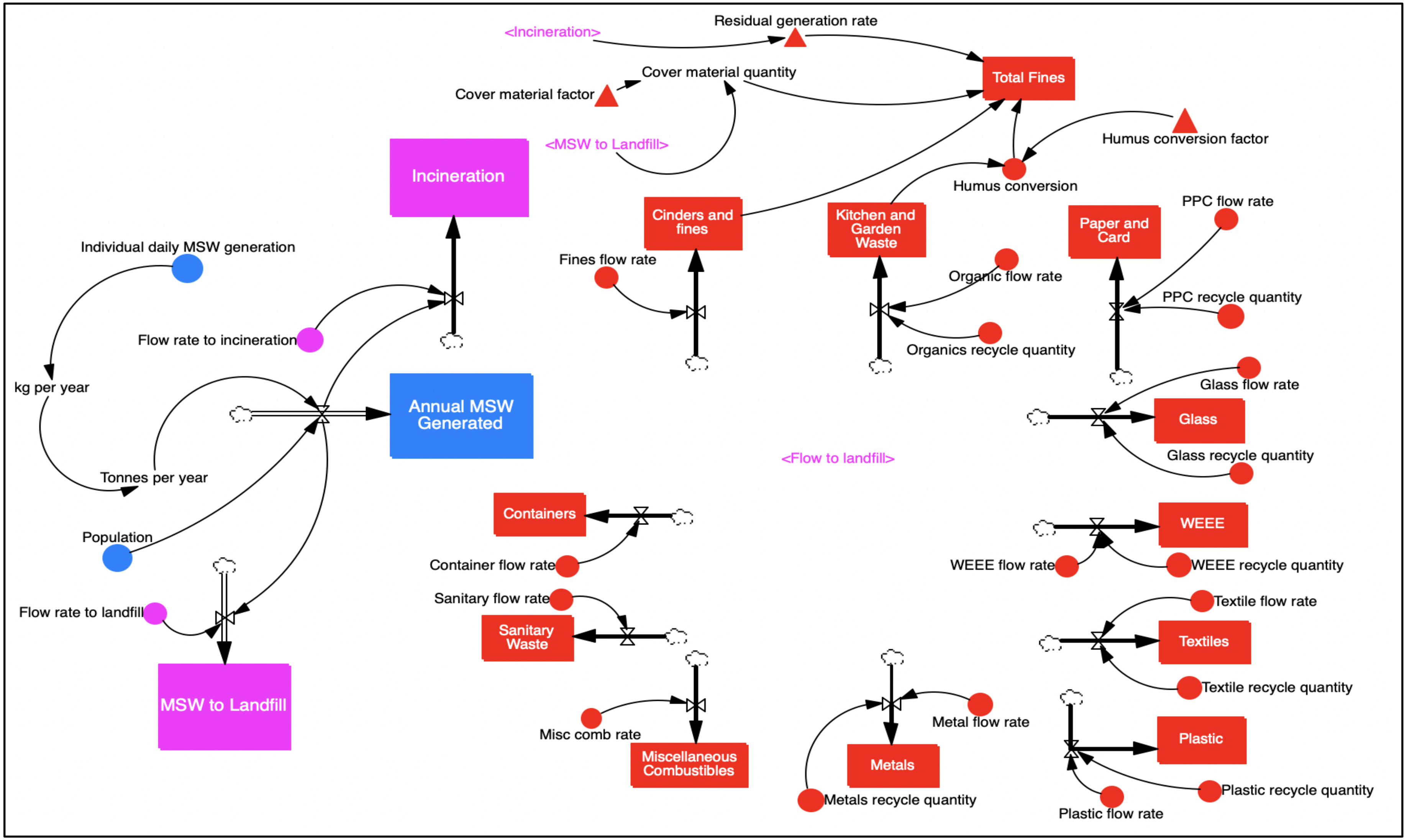


Figure 5.8: The proposed Tier 2 system dynamics model used to estimate the quantity of 11 components within landfills. Flow rates of each component are determined by respective composition percentages for local authority MSW. Where recycling is adopted nationally or by an individual local authority, these quantities are subtracted from respective flows. Connectors from the shadow variable “Flow to landfill” are hidden to improve the appearance of this sketch. In addition to the previous model sketches, triangles identify constant values.

The model structure includes 11 general MSW categories included in composition studies with 2 omitted. The 2 omissions are materials considered to be unclassifiable and non-combustible materials. Miscellaneous combustibles are included although they are treated similarly to non-combustibles and unclassified materials by waste composition analyses in that combustibles were included in other categories and often had a zero figure. However, combustibles have a potential economic value where Enhanced Landfill Mining identifies these materials as a fuel source (Jones et al 2013, Jones 2016). Value materials were defined in Chapter 4, Section 5.1 and include those MSW components having resource potential when removed from a landfill, The impact of other disposal methods which include the composting or digestion of food and garden waste are accounted for within the recycling variable. From the 165 composition analyses reviewed for this study 82 allocated a small percentage of waste as ‘unclassified’ and 80 studies accounted for non-combustible material. A separate mass balance exercise has identified a maximum of 55 – 56 million tonnes of these waste types are likely to have accumulated in English landfills during the study period compared to 885.5 million tonnes arising from the 11 modelled categories. Data for the mass balance is included as Appendix 5.4.

5.6.3 Model framework – landfill content

To obtain individual material fractions forming the landfill content, the following general equation is used:

$$MF_i = (D_{LF} \times {}^{MFR_i}/_{100}) - RQ_i \tag{5.4}$$

where $MF_i$is the individual MSW fraction (tonnes/year), MFR_i is the percentage rate of a component within the MSW stream and RQ_i (tonnes/year) the quantity recycled of material i. For those materials not recycled RQ_i is equal to zero.

From the general equation, equations for specific materials are formulated for example for the metal fraction:

$$MF_{Met} = (D_{LF} \times {}^{MFR_{Met}}/_{100}) - RQ_{Met} \tag{5.5}$$

where MF_{Met} is the metal fraction (tonnes/year), MFR_{Met} is the percentage rate of metal in the waste stream and RQ_{Met} (tonnes/year) is the quantity of metal recycled.

5.6.4 Landfill content – input data for model variables

Variables for the Tier 2 model are, as with the Tier 1 model, data driven however, MSW generation and disposal inputs into the Tier 2 model comprise only data from the Collected MSW data set. These data include those local authorities weighing 50% or more of generated MSW. Chapter 4 presented waste composition as a series of ranges. From these ranges, the arithmetic mean together with the high and low values are used as inputs to model landfill content.

Additionally, the Tier 2 model includes Salvage and (later) recycling as a variable. Both activities diverted potentially valuable materials away from landfill. The published literature (Dawes, 1953; Wylie, 1959; Higginson, 1960 and 1966; Ministry of Housing, 1954-1966; Bailly and Tayart de Borms, 1977; Neal, 1979; CIPFA, 1979-2000; Trade and Industry Committee, 1984; Waite, 1995; DEFRA 1996-2008, Zweiniger-Bargielowska, 2000; Stokes et al., 2013; Thorsheim, 2015) provides limited historical data in respect of the weights of materials salvaged or recycled however, these data are included in the material recycling variables indicated by Figure 5.8. For the most part, salvage can be confined to the first epoch and by the mid-1960s has dissipated to reappear as waste recycling in the mid-late 1990s. Good fortune prevails, and the better data is available for these periods. It is accepted some materials, metals, paper and card and glass were lost from the MSW stream as a result of bottle banks and the like however, recycling was not effective in the UK until the late 1990s (Stokes et al., 2013).

5.6.5 Landfill content – cover materials and rates of decomposition

Landfill mining projects reviewed in Chapter 2 (Section 4.3) identify soil/fines content to be the largest component where MSW landfills contain fifty to sixty percent by weight. Fines and soil content occur from their presence in the MSW stream however, whilst these levels reflect pre 1960s waste composition their content reduces progressively to below 10% (Chapter 4, Section 5). Soil/fines content are introduced into landfills as daily cover materials, incineration residuals and from rapidly decomposing organic materials, the latter occurring in MSW as food and garden wastes. The municipal waste stream contains organic components in paper and card, textiles and different forms of plastic. However, for the purposes of this study, it is assumed only EU (2008) defined biodegradable wastes undergo rapid degradation given this study's time frame. Rapid degradation applies to a period extending to some 4 years following deposition (Ehrig and Stegmann, 2019). EU (2008) article 3, paragraph 4 identifies kitchen, garden and park waste from households, restaurants, caterers and retail premises as biodegradable.

The literature proposes that 1st order kinetics with respect to landfilled organic substrates is the most accepted decay process however, factors including moisture availability, nutrients and temperature also have an impact. Proposed values for *k* are similar to that used in methane generation (Haarstrick and Völkerding, 2007; Barlaz et al., 2010; De la Cruz and Barlaz, 2010; Lyons et al., 2010; Quaghebeur et al., 2013; Wolfsberger et al., 2015; Reinhart and Stegmann, 2019; Cossu et al. 2019; Andreottola et al., 2019). In formulating degradation as a model parameter, it becomes necessary to establish a decay rate together with resulting products. Due to the rapidity with which these wastes degrade whether in an aerobic or anaerobic environment degradation is assumed to be complete. This detail, together with the resulting degradation products are defined by a general modelling assumption where 50% by weight of the imported kitchen and garden MSW are added to the soil/fines content as humus with 50% emitted as gas or with leachate. Furthermore, these are entered as constants into the model however, model simulation using the SyntheSim function allows this to be adjusted to any chosen level. Opinion as to the degradation half-life times of other solid organic materials differs however, there is good evidence to suggest that in the anaerobic environment of the landfill, whilst unprinted paper undergoes degradation, newsprint and coated paper remains and will remain in the landfill for a considerable and undefined period (Bevan, 1967; Rathje and Murphy, 2001).

Controlled tipping and engineered landfills require the application of cover materials during the landfilling process. There is no clear methodology to assess how much covering material was used at any landfill. Cover material was either site-won or imported and comprised excavated soil and incineration residues. Bevan (1967) and Wilson (1981) both described the materials used and reported studies where the ratio of covering materials to MSW were measured. However, these were heavily dependent on the fines content of imported wastes and the plant used to spread and compact both the waste and covering material. Bevan (1967) identified in the Manchester Experiments a waste to cover material proportion between 20.5% - 49.6%. Wilson (1981) cites K. J. Brately's tests reported in *Solid Wastes* (1977) where cover material used per tonne of waste ranges from 0.37 tonnes – 0.96 tonnes, that is 37% - 96%. The use of cover materials necessarily increased due to the proportion of fines and ash in MSW reducing as alternatives to solid fuel heating were adopted. For the Tier 2 simulations presented in the Sections below, the cover material factor is set at 30%, 40% or 50% of imported MSW unless specific data is available. However, model simulation using the SyntheSim function allows this to be adjusted to any chosen level. From discussions with contemporary landfill operators, daily soil covering is assumed to be 30 - 50% of MSW input (by weight) with the honest assessment that daily covering is reducing landfill volume.

5.6.6 Landfill content – model output as material flows

Chapter 4 identified MSW components as fitting limited ranges of values as opposed to discrete values. As such, the model generates three output values. These outputs are presented in graphical form (Figure 5.9 (a – h)) as a series of material flows entering English landfills.

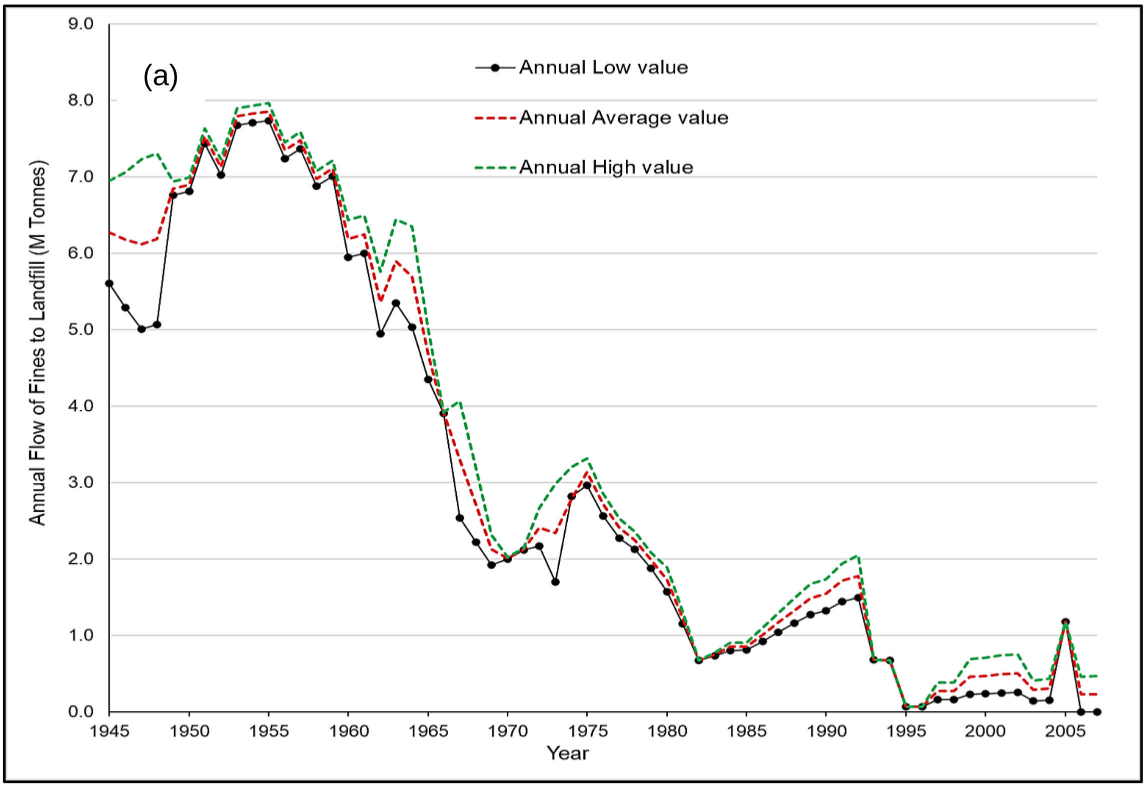


Figure 5.9 (a): Modelled annual flows to English landfills of soil and fines 1945-2007.

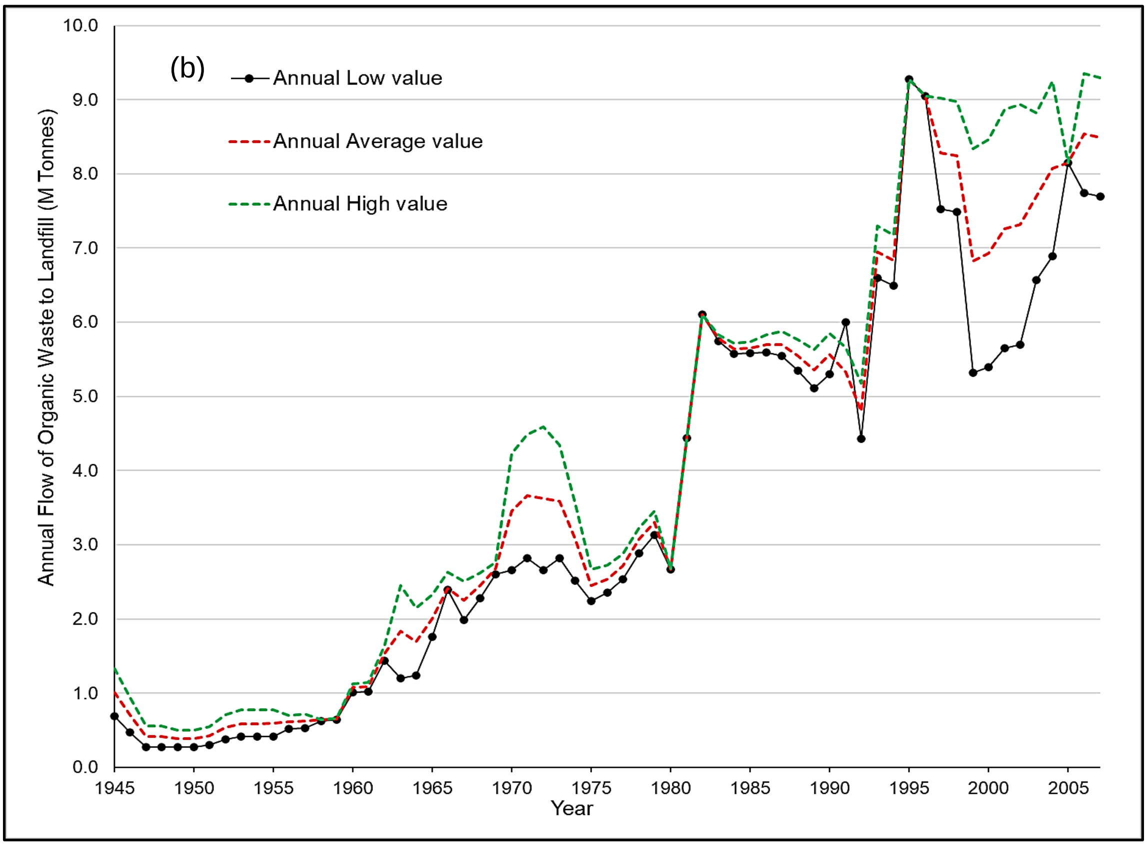


Figure 5.9 (b): Modelled annual flows to English landfills Kitchen and garden (organic) waste 1945-2007.

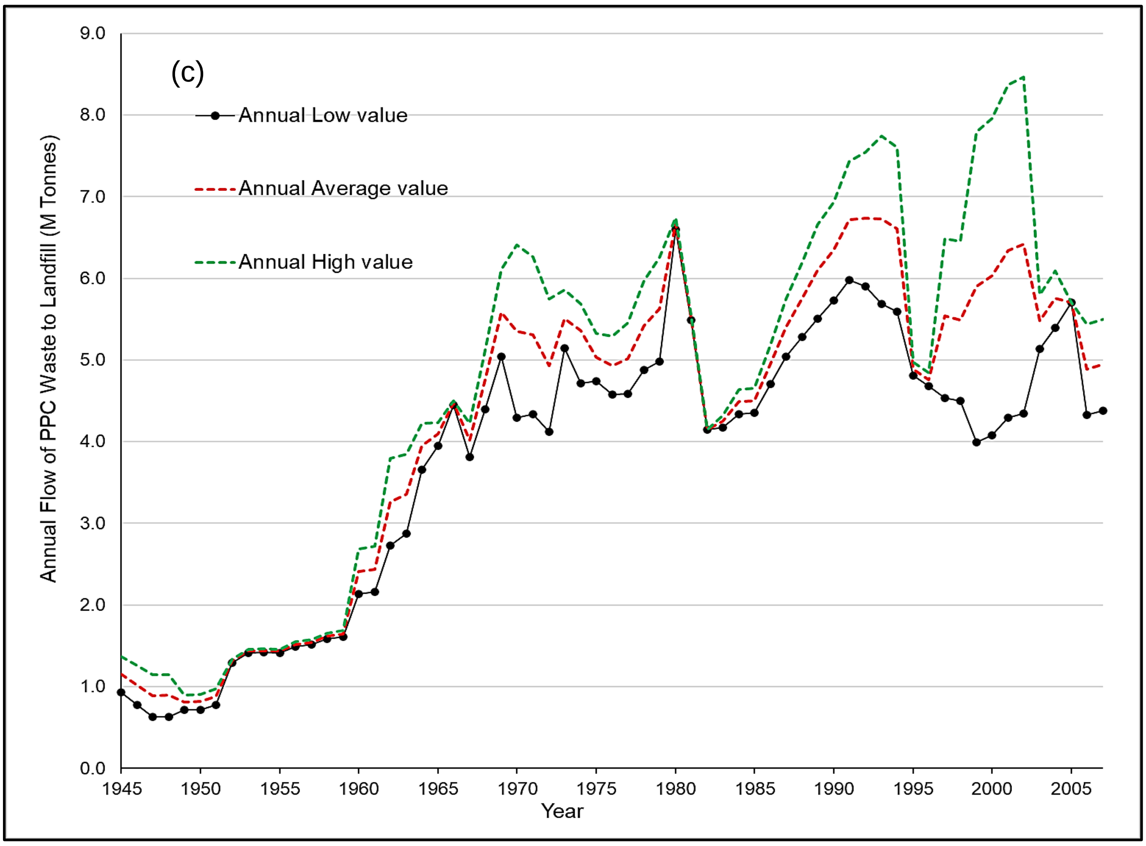


Figure 5.9 (c): Modelled annual flows to English landfills of paper, packaging and card (PPC) 1945-2007.

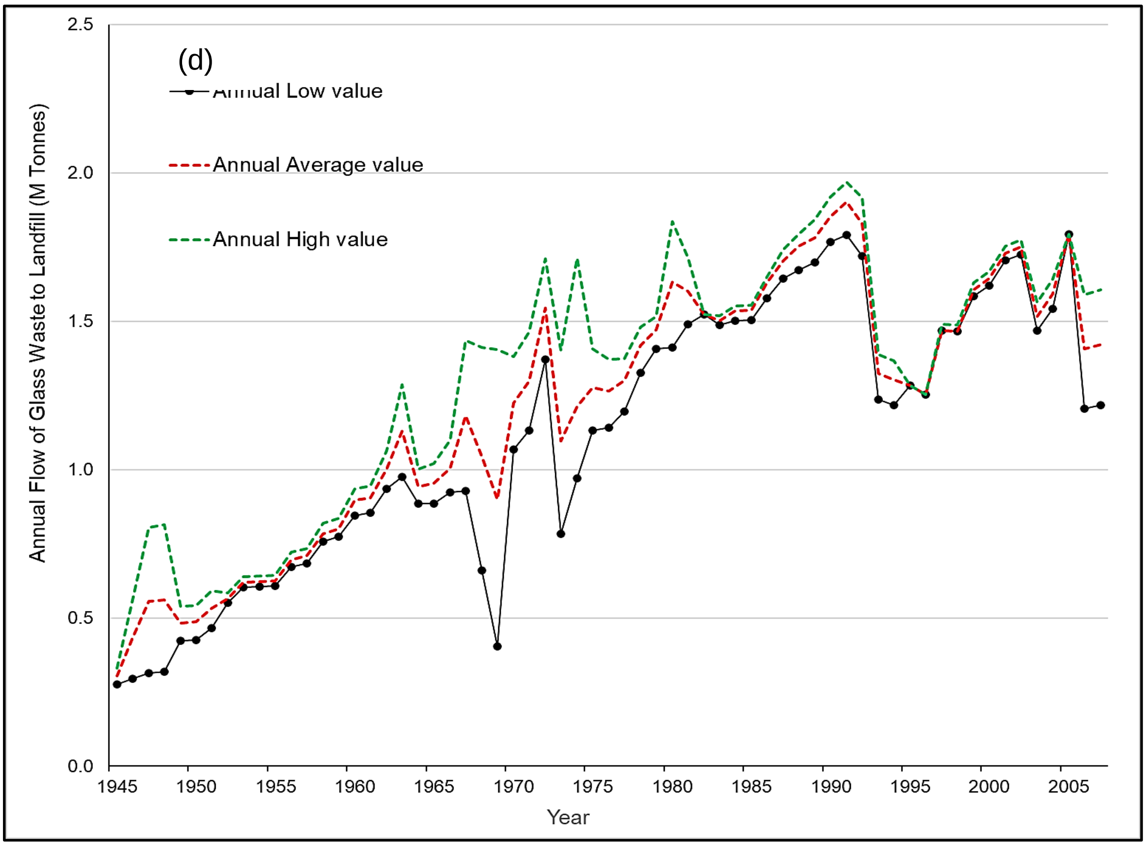


Figure 5.9 (d): Modelled annual flows to English landfills of glass waste 1945-2007.

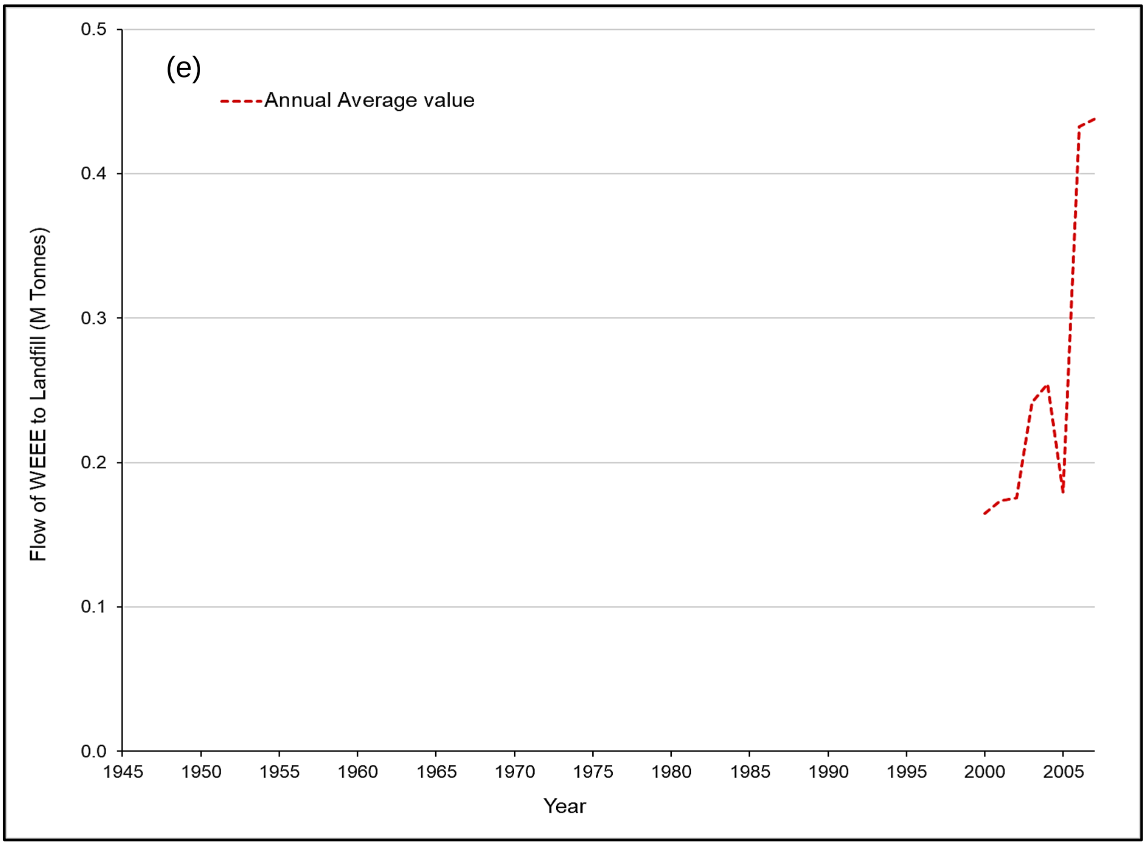


Figure 5.9 (d): Modelled annual flows to English landfills of waste electrical and electronic equipment (WEEE) 1945-2007.

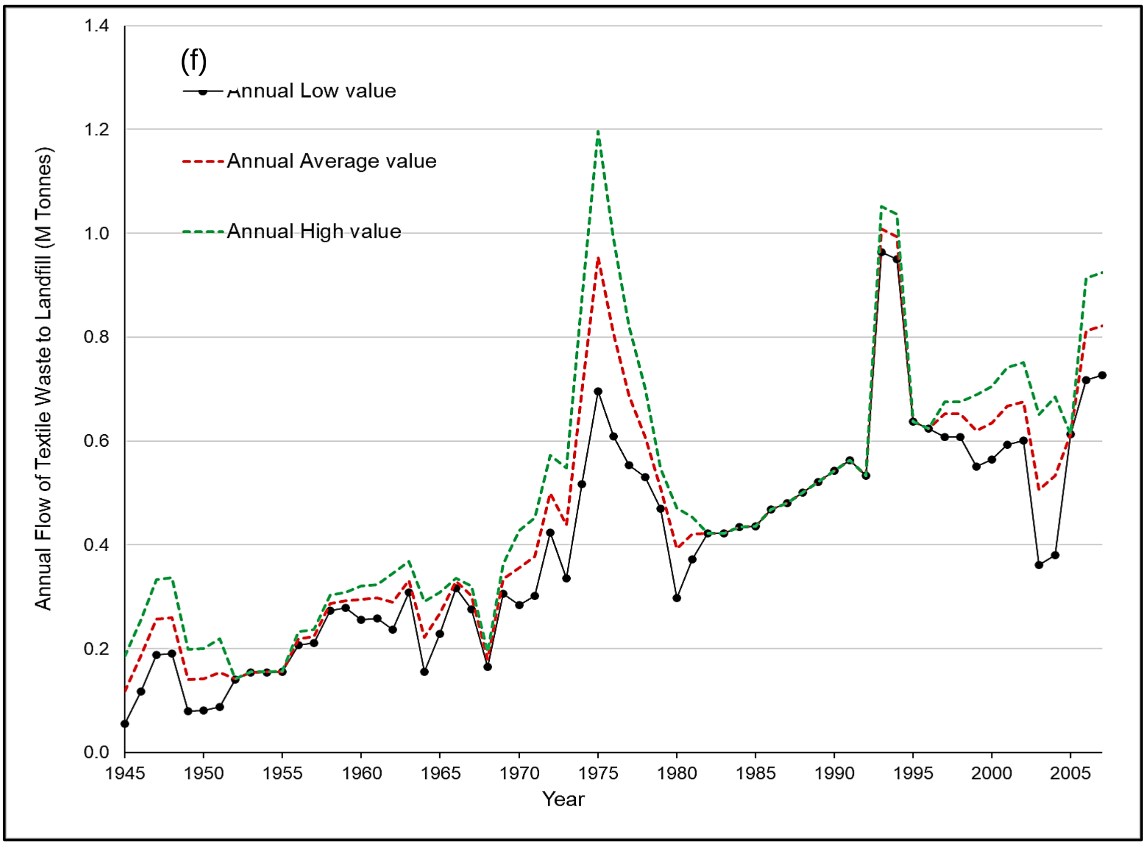


Figure 5.9 (f): Modelled annual flows to English landfills of textile waste 1945-2007.

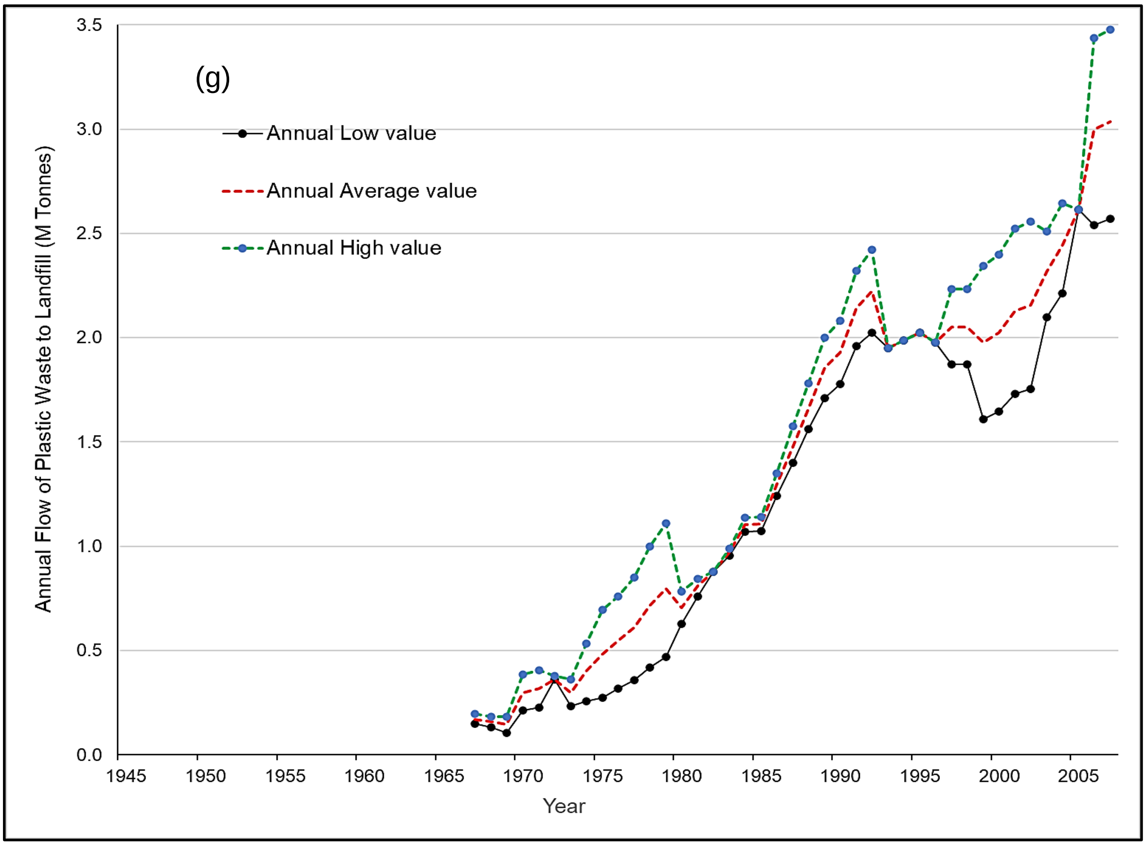


Figure 5.9 (g): Modelled annual flows to English landfills of plastic waste 1945-2007.

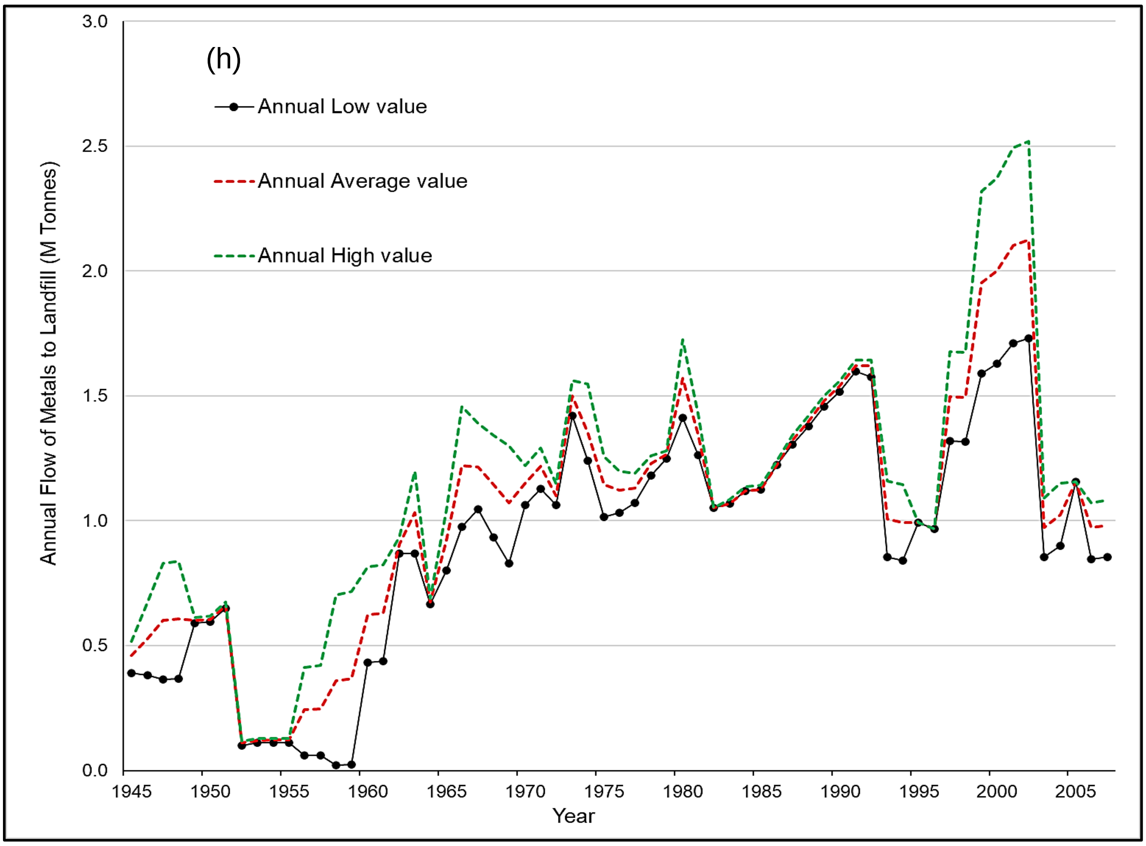


Figure 5.9 (h): Modelled annual flows to English landfills of plastic waste 1945-2007.

The cumulative total for each material presented in Table 5.1.

Table 5.1: Model output for cumulative low, average, and high values at 2007/08 for materials flowing into English landfills (no degradation of organic content is included).

Material composition from the MSW stream	Cumulative Low value	Cumulative Average value	Cumulative High value
Cinders and fines (tonnes ‘000)	158,537	170,581	182,826
Kitchen & garden waste (tonnes ‘000)	171,823	190,514	268,786
Paper, packaging & card (tonnes ‘000)	182,888	207,995	232,238
Glass (tonnes ‘000)	52,362	57,567	62,648
WEEE (tonnes ‘000)	1,312	1,544	1,731
Textiles (tonnes ‘000)	20,523	23,194	25,790
Plastic (tonnes ‘000)	41,307	46,322	51,238
Metals (tonnes ‘000)	43,100	49,204	55,241
Combustible content (tonnes ‘000)	27,977	41,792	42,455
Sanitary content (tonnes ‘000)	15,274	19,827	22,863
Containers – counted as metals after 1959 (tonnes ‘000)	5,107	5,293	5,532
Total (tonnes ‘000)	720,210	813,833	951,348

5.7 Verification and Validation

5.7.1 Introduction

Confidence in any numerical model results from the methods used to verify and validate a given model. Verification and validation are processes that demonstrate how correctly and accurately a model is able to represent the scenarios it was designed to replicate. Verification is a process to determine a model's output concurs with what was intended. Validation is the process of determining to what degree a model's output is representative of the real world. In short, validation is an assessment of a model's ability to predict. Furthermore, neither verification nor validation can confirm a model's accuracy for all its designed applications however, it can present evidence a model is sufficiently accurate to deliver its intended outcomes (Thacker et al., 2004).

To verify the system dynamics model its outputs are compared to an alternative mass balance framework using the same data as inputs. Methods to validate system dynamics models have received little attention (Barlas, 1996). Best practice in the use of system dynamics modelling has been researched and published by Martinez-Moyano and Richardson (2013). Leading exponents were asked to attribute a level of importance to 6 stages forming the modelling process. These were further graded in importance from highest, high, average and low. From the outset it became this study's intention to incorporate those practices particularly in respect of the highest category: (i) identification of the initial problem, (ii) the model's conceptualization and (iii) the formulation of the model that is, how it represents the real-world scenario. A 4th consideration is evaluation where there is consistency in the way a model's behavior reflects historical elements or reference factors.

5.7.2 Proposals for testing

To verify and validate system dynamics models Barlas (1996) proposed: i) structural assessments and ii) behaviour pattern tests. This study proposes 2 approaches to verify and validate the presented model where, from this point, modelled output is presented with kitchen and garden wastes fully decomposed. The first uses a bottom-up mass balance to verify the model's output for each sub-unit thus testing its structure through each Tier. For the behaviour pattern testing the model's output is compared to data from exhumations of landfill sites: i) Experiments undertaken in Manchester and reported by Bevan (1967), ii) results published by Wagland et al. (2019) where sampling was undertaken in UK landfills and iii) data from 2 investigations of European landfill content undertaken at the REMO site in Belgium and an unnamed site located in Austria. However, whilst sampling and modelling are able to provide an indication of a landfill's content, only excavation will reveal the true extent of any landfill's content.

5.7.3 Verification - comparison to a mass balance alternative

To verify model output a truncated mass balance was produced with Excel using the same local authority collected data set and average composition values for each MSW component. The results from the mass balance, the Tier 1 and Tier 2 models are compared in Table 5.2 where the same values are generated for each modelling approach with the values obtained from the mass balance concurring with those obtained from the system dynamics model.

Thacker et al., (2004) identify verification as a comparison to an accurate benchmark solution as a means to checking the mathematics of the model. The mass balance is a long-hand method using the same data and equations however, it provides a robust methodology with which to compare results generated by the system dynamics model. Clearly, with the large number of cells requiring data, it is possible for errors to occur and numerous checks are necessary even when using a truncated mass balance format. Furthermore, the necessity to check and recheck the mass balance input data reinforces confidence in the system dynamics model's output, where there is agreement.

Initially, it was intended to generate a series of mass balances using spreadsheets. It was not until the fourth year of this study that system dynamics was adopted as a better alternative. Generating a mass balance using Excel requires the same contemporaneous data to that used in the system dynamics model however, what is produced by a spreadsheet is a static picture of a dynamic situation. However, by using Excel to deliver these data as a series of scenario matrices to the Vensim software, a dynamic set of outputs is obtained. Furthermore, input data can be simply adjusted using the SyntheSim function to create multiple simulations so as to examine changes in modelled behaviour.

Table 5.2: Results of the mass balance comparison used as verification of the system dynamics model output. The inputs to the mass balance exercise are identical to those used in the system dynamics model as such, the expected outputs should be the same, which is the result.

Calculated output	Mass balance calculation steps	Mass balance collected value	Tier 1 collected value	Tier 2 collected value
Total MSW generated 1945 - 2007	Population of England multiplied by local authority collected generation rate for each year (range 0.670 - 1.389 kg/day)	1,051,094,859 tonnes	1,051,094,859 tonnes	1,051,094,859 tonnes
Landfilled with no salvage or recycling	Total annual MSW generated multiplied by % being disposed to landfill (average percentage approx. 84.5%)	885,511,000 tonnes	885,511,000 tonnes	885,511,000 tonnes
Cumulative quantity of incineration residues	Quantity incinerated multiplied by 35%	36,274,700 tonnes	36,274,700 tonnes	36,274,700 tonnes
Total quantity landfilled	MSW landfilled + incineration residues	921,785,700 tonnes	921,785,700 tonnes	921,785,700 tonnes
Cumulative quantity of incinerated	Total annual MSW generated multiplied by % being incinerated (average percentage approx. 10.1%)	105,502,600 tonnes	105,502,600 tonnes	105,502,600 tonnes
Cumulative quantity recycled	Total annual MSW generated multiplied by % being salvaged or recycled (average percentage approx. 5.18%)	55,744,100 tonnes	55,744,100 tonnes	55,744,100 tonnes
Resources landfilled following salvage or recycling		(tonnes)		(tonnes)
Kitchen and garden waste (organics)	Flow of resources to landfill = weight of resource based on percentage by composition less resource removed due to salvage or recycling	190,514,070	-	190,514,070
Metals		49,204,014	-	49,204,014
Glass	In each case MSW composition values are the average of the low-high range produced from collected sampling exercises.	57,567,360	-	57,567,360
Paper, card and packaging (PPC)		207,995,189	-	207,995,189
Plastic	In each case, the mass balance calculation and the Tier 2 model utilise the same published weights for recycled materials and where these are not available, estimates determined by contemporary practice and interpolation.	46,322,150	-	46,322,150
WEEE		1,544,372	-	1,544,372

5.7.4 Comparison to the Manchester Experiments

The first behaviour pattern test compares the model's output to 2 experiments conducted in Manchester during the early to mid-1960s:

- Experiment 1 was undertaken to measure the effect of compaction on the decomposition of MSW and was undertaken during Autumn 1960.
- Experiment 2 examined waste, disposed in 1938, to review longer-term waste decomposition.

Both experiments are reported by Bevan (1967). Whilst the original objectives of each experiment are detached from this study's objectives, aspects of the results offer a record of what would be expected from the excavation of a contemporary English MSW landfill and provide a comparison to the proposed model's output. For Experiment 1 and to enable the comparison the model is simulated using MSW generated in 1959 with specific data resulting from a waste composition study completed for the Corporation of Manchester and presented as Table 5.3.

Experiment 1 was conducted during September and October 1960 using 4 individual cells or what Bevan describes as "plots", Plot N, Plot C, Plot S and Plot New. Bevan reported that whilst no record was made in respect of individual MSW components used in the experiment, MSW composition reflected that presented in Table 5.3. The composition of the individual plots is presented in Table 5.4

Table 5.3: Manchester MSW analyses undertaken in 1959 for 3 major property groups.

Waste component	Old-terraced house	Semi-detached	Higher rateable value
Fine dust and cinder	56.82	50.35	42.02
Kitchen and garden waste	5.24	8.01	13.76
Paper & card	13.13	14.6	23.76
Glass & Cullet	6.09	5.25	5.40
Textiles	2.19	2.39	1.91
Metals	5.67	8.67	7.59
Bones	0.09	0.08	0.11
Miscellaneous combustibles	2.22	2.9	0.58
Miscellaneous non-combustibles	8.54	7.74	4.87
	100%	100%	100%

Table 5.4: Waste and cover material weights for the 4 plots used in Experiment 1. The cover material content comprised incineration dust for Plots N and C and decomposed refuse for Plots S and New.

	Plot N Imports	Plot C Imports	Plot S Imports	Plot New Imports
House refuse (tonnes)	77.22	117.35	204.89	78.49
Cover materials (tonnes)	75.54	45.98	52.83	25.55
Cover materials as % of total imports	49.6%	28.15%	20.5%	24.6%

Figure 5.10 illustrates the prepared cells and progress of Experiment 1 with filling taking place within Plot N. The plot is located to the far right-hand side of the photograph with attendant personnel hand raking the waste to recreate the pre-war scenario. The fourth plot, Plot New is yet to be constructed and will occupy the far left-hand side of the experimental site.

Figure 5.10: Site of the experimental plots for Experiment 1. Plot N is the right-hand plot with men hand raking waste. Plot New will be constructed to the left of Plots N, C and S. Reproduced from Bevan (1967 plate x).

Whilst no record was made of the refuse disposed into the experiment's plots, Figure 5.11 provides an indication of freshly deposited 1960s household refuse although no precise date is given for this photograph.

Figure 5.11: Whilst no records of the refuse used in the experiments are available, this contemporary photo of Manchester Corporation's waste operations provides an indication of freshly deposited MSW. However, no precise date is given for this photograph. Reproduced from Bevan (1967 plate xii).

Good landfilling practice necessitates the formation of discrete layers to improve compaction. With larger landfills, this method of operation introduces likely temporal variation into successive layers. In respect of these experiments the limited size of each plot allowed the filling of each to be completed very rapidly. As such, the excavated samples should reflect the components presented in Table 5.3 subject to the addition of cover materials, decomposition of readily degradable organic components and damage caused through the mechanical process occurring during deposition. The addition of cover materials increases the soil/fines content whilst decreasing the proportional percentage of other MSW components when measured following deposition and excavation. It is well established that kitchen, and many garden wastes degrade very quickly adding further to the soil/fines content as humus.

The plots created for Experiment 1 were sampled twice: i) in 1961 approximately 12 months after filling and ii) during August 1965. For the first sampling exercise, the most notable compositional change was the lack of organic matter. The report states "*no vegetable and putrescible matter, except woody twigs etc., could be detected*". The second series of samples revealed noticeable deterioration to food containers particularly rusting and pitting to tins (Bevan, 1967 pp.90-91).

The system dynamics model was calibrated with data specific to Manchester for the period 1958-1960. Four MSW to cover soil ratios were used in separate simulations to replicate those used in the experiment: i) 50%, ii) 30%, iii) 25% and iv) 20%). MSW composition is drawn from the sampling data presented in Table 5.3 where the high and low values are taken from each composition category over the 3 series of samples to form a range within which other values lie. A similar approach is adopted in respect of the 4 experimental data sets. The population in Manchester reported to Ministry of Housing during 1959/60 was 672,300. MSW generation data provided to the Ministry of Housing identified 17.7 hundredweight (899 kg) per 1000 head of population per day, approximately 900 grams per person per day. Manchester Corporation's disposal strategy included controlled tipping (85%) and incineration (15%). Some separation of components for salvage was reported but this occurred only prior to incineration (Ministry of Housing and Local Government Costing Returns, 1959/60). The model output includes 220,700 tonnes of generated MSW of which 188,720 tonnes were landfilled, 32,000 tonnes of waste incinerated generating 11,200 tonnes of residual ash. The model's output and results from the 1961 and 1965 sampling exercises are presented in Figure 5.12 (a – e) as a series of boxplots.

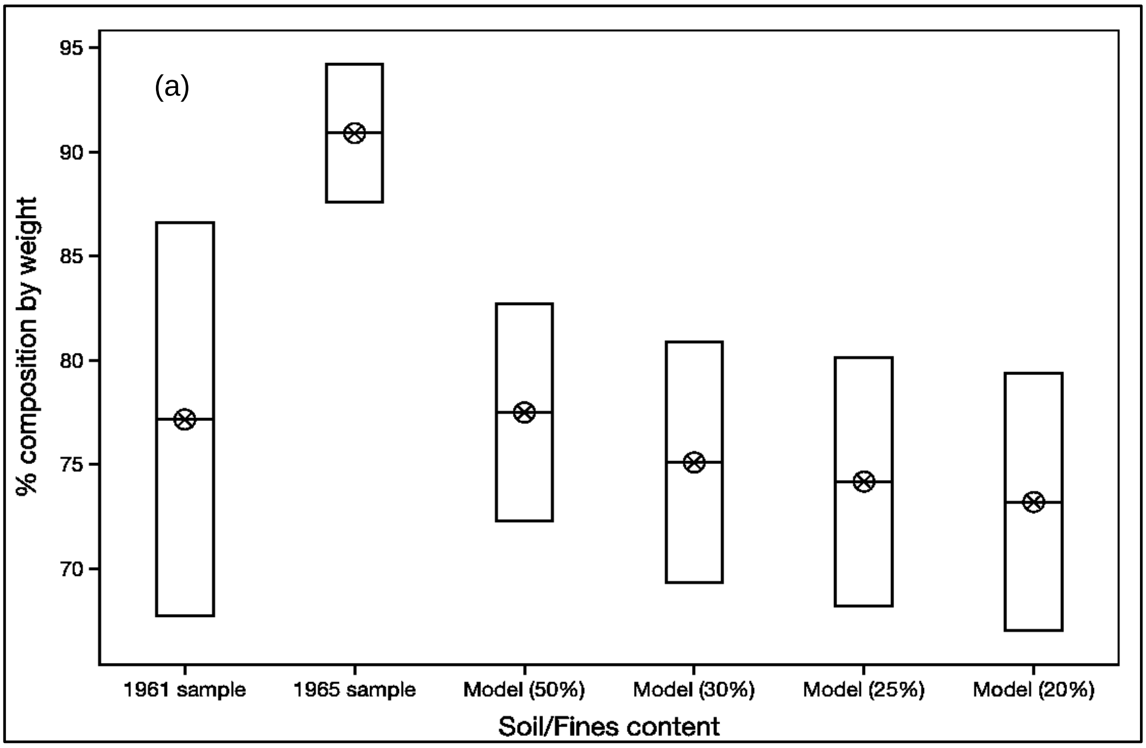


Figure 5.12 (a): Comparison of model output to 1 and 5-year-old excavated MSW. Cover materials used in the model equate to 50%, 30%, 25% and 20% of imported MSW to reflect those used for the experimental 4 plots – soil/fines content

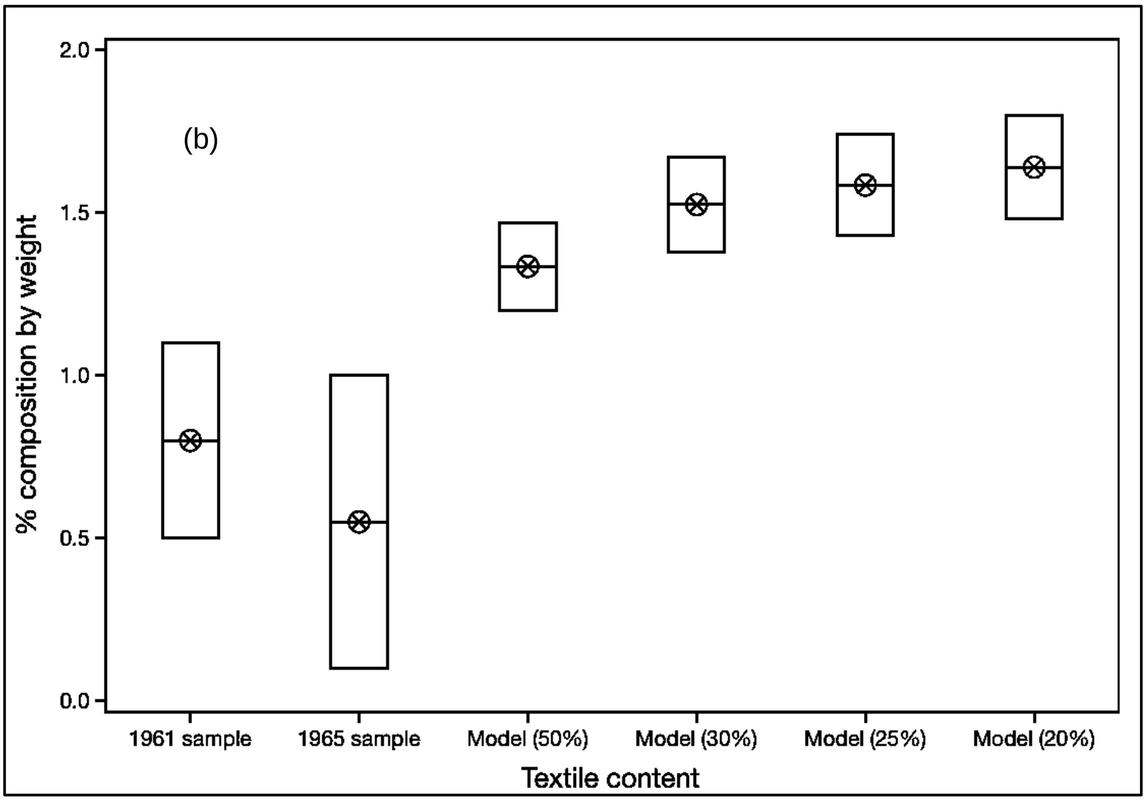


Figure 5.12 (b): Comparison of model output to 1 and 5-year-old excavated MSW. Cover materials used in the model equate to 50%, 30%, 25% and 20% of imported MSW to reflect those used for the experimental 4 plots – textile content

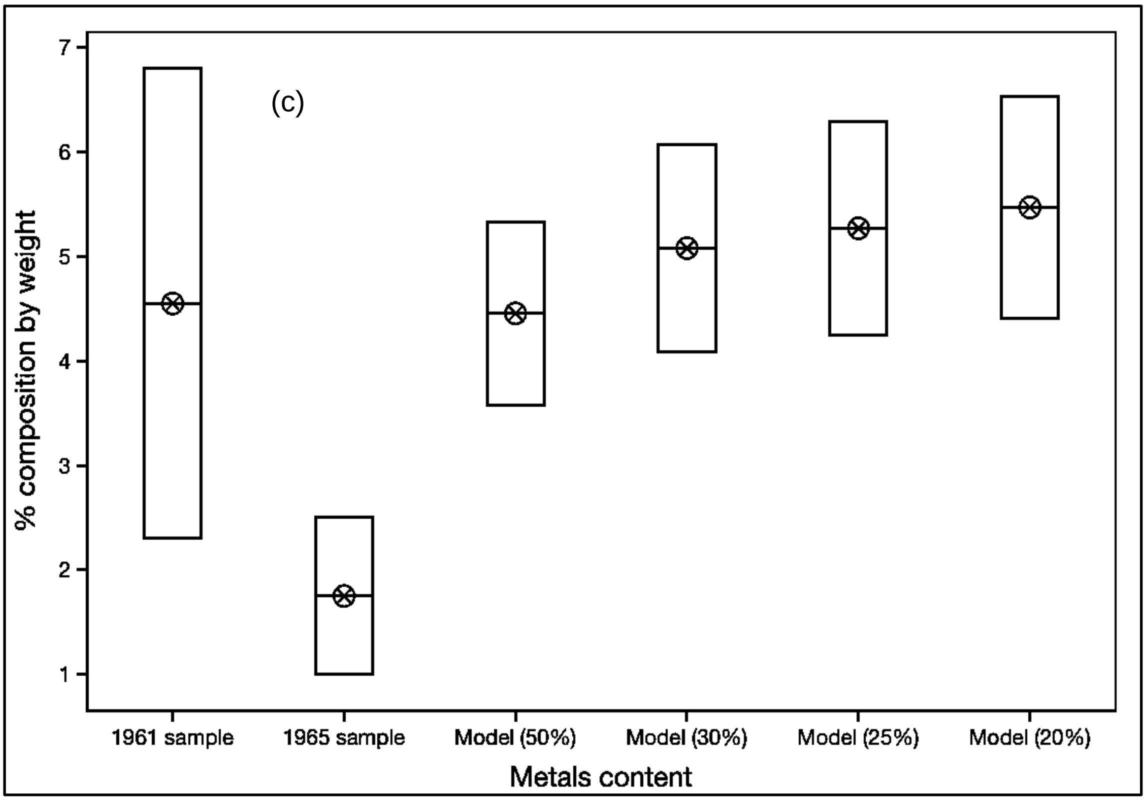


Figure 5.12 (c): Comparison of model output to 1 and 5-year-old excavated MSW. Cover materials used in the model equate to 50%, 30%, 25% and 20% of imported MSW to reflect those used for the experimental 4 plots – metals content

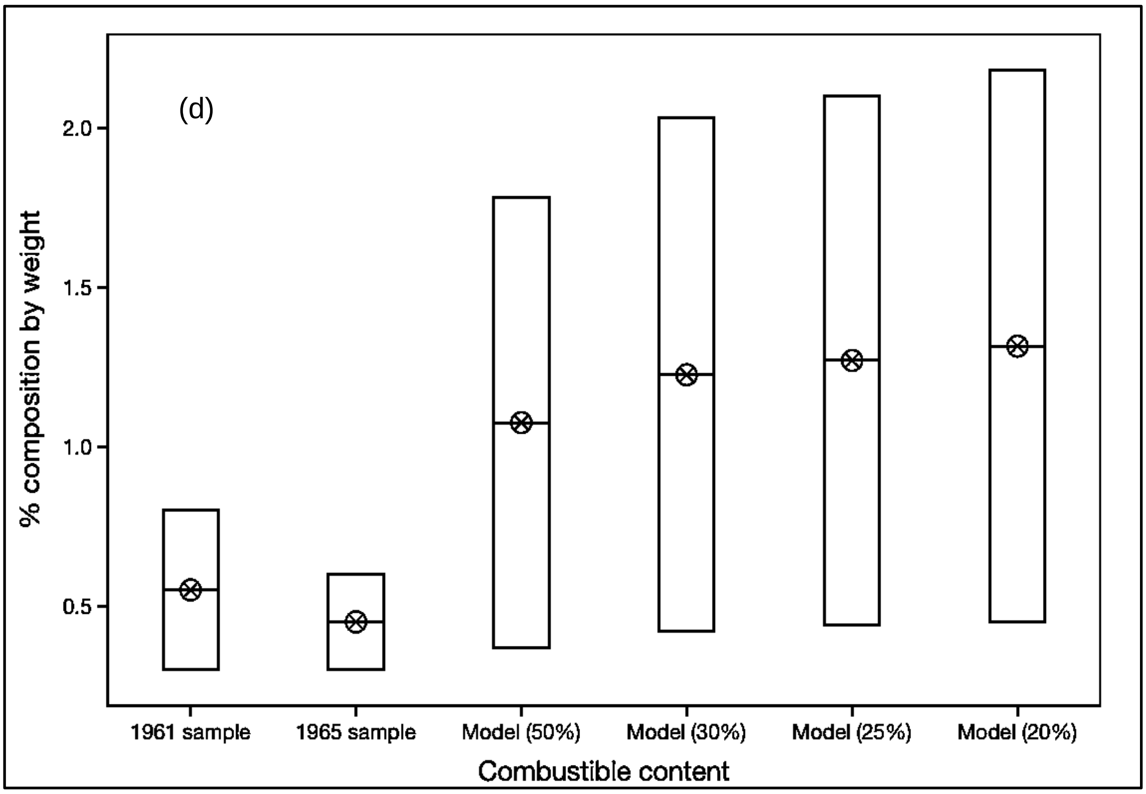


Figure 5.12 (d): Comparison of model output to 1 and 5-year-old excavated MSW. Cover materials used in the model equate to 50%, 30%, 25% and 20% of imported MSW to reflect those used for the experimental 4 plots – combustible content

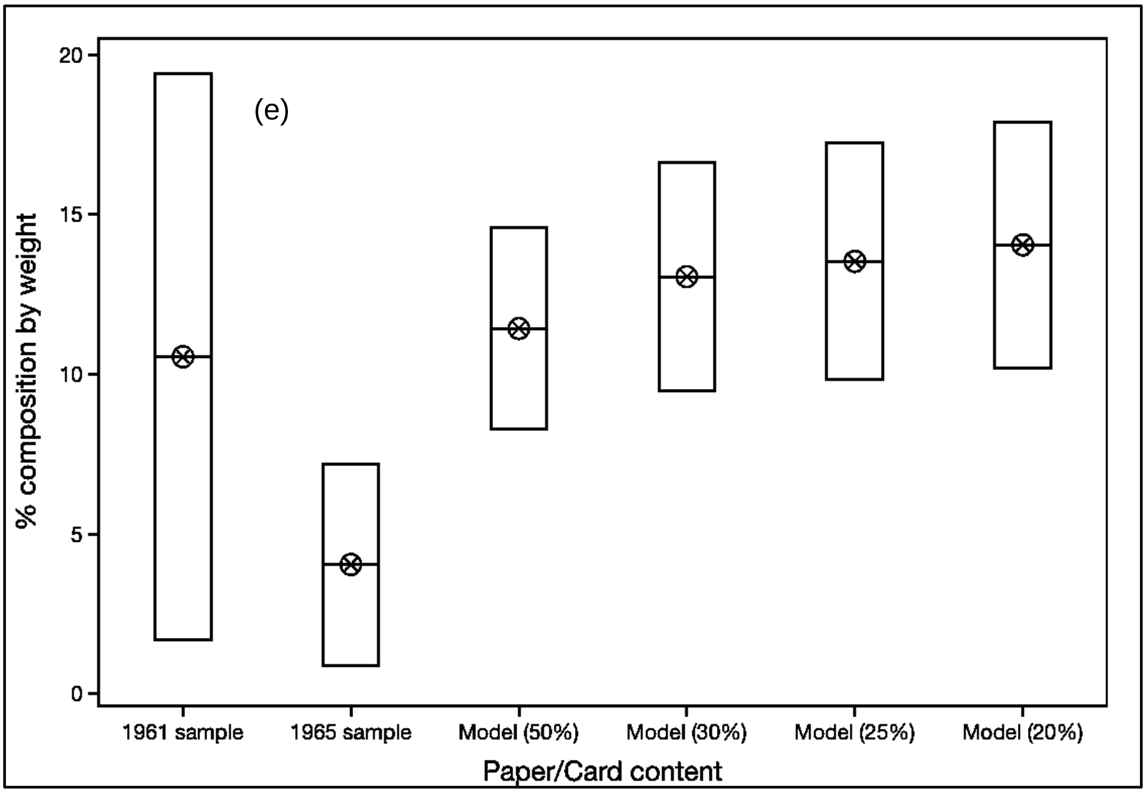


Figure 5.12 (e): Comparison of model output to 1 and 5-year-old excavated MSW. Cover materials used in the model equate to 50%, 30%, 25% and 20% of imported MSW to reflect those used for the experimental 4 plots – paper and card content

Note: i) The median for each boxplot is indicated by the circled x.

The model output when compared to the range of samples for all plots taken in 1961 lie within the sample range for glass, soils and fines, metal, and paper and card and, as such, are good estimators. The model output in respect of miscellaneous combustibles and textile contents overstate the sampled content however these are small with neither exceeding one percentage point as such and in respect of each plot's content, the model simulation reproduces the experimental samples.

For the second series of samples taken in 1965, approximately 1 cubic yard (0.765 cubic metre) was taken from each plot during August. Unfortunately, no detail in respect of location or depth is reported by Bevan. There are some noticeable differences between the 1965 samples, the model output and the samples taken in 1961. The presence of, and large increase, to such a high percentage of soils and fines, 87.6 - 94.2%, suggests the inclusion of a quantity of the final layer of covering within the samples, so skewing the analysis. Bevan identifies that only a "*rough*" analysis was undertaken with no record of the sampling locations or depths included in the published text. A further indication is offered by the model which was calibrated with 4 MSW to soil cover ratios to replicate that applied to each plot. As can be seen from Figure 5.10 (a), the median for the second series of samples exceeds those of the model and the 1961 sample by 14 – 18 percentage points. A further indicator is the absence of paper and card in the 1965 samples. Bevan (1967), in reference to the 1961 samples, refers to the lack of noticeable degradation in the paper and card fraction. Samples taken from Plots N, C and New contained 1.9%, 2.9% and 0.9% respectively with the Plot S sample containing 7.2%. The plot S sample might be considered an outlier as the average for the other 3 plots is 1.9%. The paper content from the MSW composition samples extends from 13.13% to 23.6% with the likely percentage imported into the plots lying within that range.

Whilst the model output is representative of the 1961 sampling it is proposed those samples taken in 1965 included a greater percentage of cover soils due to their being removed at or near to the surface of the plots and were thus unrepresentative of the waste mass. The small amount of paper and the increased quantity of fines suggest this is more than a possibility. Sampling in any landfill mass will present problems particularly in respect of sample size and location. These issues were discussed in Chapter 3 and are identified in the published literature.

Experiment 2 reviewed degradation occurring over a 25-year period from a 10724 pounds (4,864 kgs) sample of 1938 waste. For this study's purposes the results from the sampling analyses have 2 positive impacts: i) the sampling provides insight as to how materials degraded within the landfill environment given UK climatic conditions, and how degradation can be incorporated into a model's structure and ii) although this landfill was operational before the study period, MSW composition data is available for 1937/38 which allows a non-specific simulation. Specific and non-specific were defined in Chapter 1, Section 5.

From the published report, Bevan (1967) notes that a first inspection suggested little paper remained, the greater part having degraded. However, a fuller examination revealed paper, particularly bundled paper, had the appearance of cemented ash. Bevan proposes that bundled paper would take "*a very long time*" to disappear. Examples of excavated paper and textiles are presented in Figure 5.13. Rathje and Murphy (2001) report similar findings and suggest the well managed landfill is "*far more apt to preserve their contents than turn them into humus or mulch* ". The model does not provide for paper and card degradation.

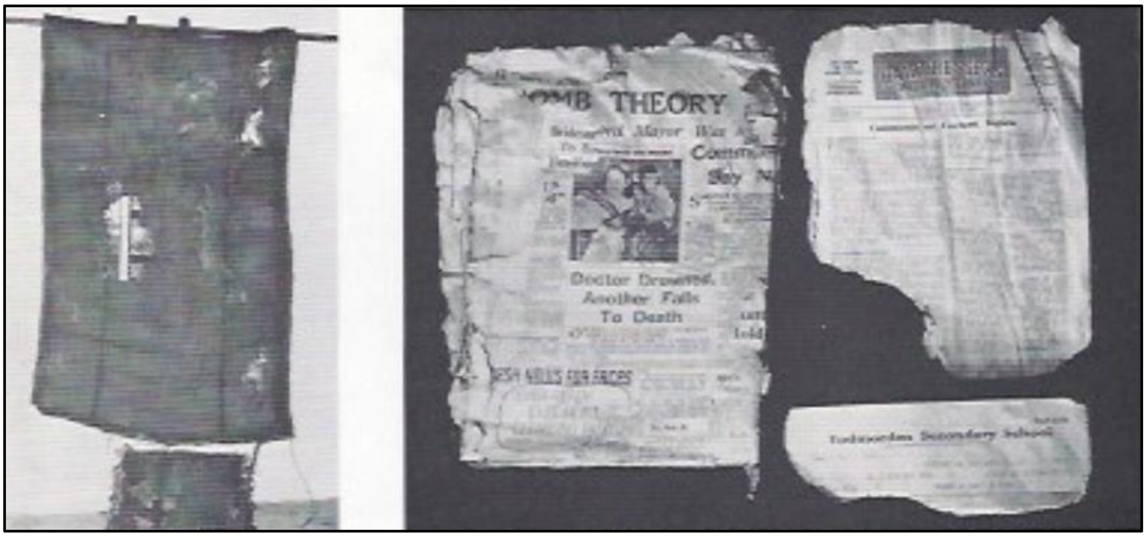


Figure 5.13: Materials recovered from the excavation of a 1938 Manchester landfill. The first photograph is coco-matting (sometimes referred to as coir-matting) and often used in doorways. The newspapers, despite being landfilled for 25-years, were fully legible.

The results of the sampling and modelled output for the value materials using 1937/38 MSW composition are presented as a direct comparison in Table 5.5.

The model was simulated with a 50% MSW to soil cover ratio as this was, as identified by Bevan, the adopted contemporary methodology.

Table 5.5: Results from the sampling of 1938 MSW taken from Northenden, Manchester compared to the model output using MSW composition data taken from Ministry of Health Public Cleansing (1937), the Department of Environment (1971) and Higginson (1960).

Composition by weight (%)	Paper	Glass	Metal	Textiles	Combustibles	Fines / cover
Excavation sample from 1938 MSW	4.97	2.54	0.39	0.46	1.67	85.07
Model average simulated with 50% MSW/soil cover ratio	4.84	1.31	2.0	0.54	0.92	82.23

The model generates similar percentages to those sampled with paper and textiles being a very close match. Whilst there is no suggestion the modelled output reflects the entire 1938 landfill, however, and when compared to the sample (4.86 tonnes), it is proposed the model is a good estimator.

5.7.5 Comparison to samples taken in UK landfills

Published data detailing the content of English landfills following excavation or site investigation is scarce. A number of historical landfills have been used for construction and recreational purposes where site investigation using boreholes has occurred, but these were to determine the depth of waste and its likely longer-term impact on foundation piling. Site investigation of this type is usually undertaken using 100mm bore-holes with the waste mass often described simply as made ground or landfill with no reference to individual waste components.

The identification of individual waste components is best achieved with the removal of the waste mass. Three projects were completed where the waste mass was removed and relocated: i) Packington Landfill near Birmingham, ii) Whinney Hill Landfill in Lancashire and iii) Jameson Road Landfill, again in Lancashire. These projects have been reviewed in the literature without specific detail of landfill content (Hayward-Higham, 2008; Ford et al., 2013). Furthermore, within the UK, remediation projects where removal of landfilled materials are a necessary precursor to development with their content remaining commercially sensitive. The author has been provided with summary data in respect of one project, but the data are too broad-brush for this study's purposes. Furthermore, the data were provided under the express agreement they would not be discussed or published.

Germane but limited data has been published by Wagland et al. (2019). The authors present sample data taken from 9, undisclosed UK landfills having received MSW and waste from commercial premises with each site identified by a reference number only. Data is provided from 36, 450mm cores from which 118 samples were categorised and presented in Table 5.6. The model was run with two series of non-specific MSW composition data: i) for the early 1980s and ii) for the 1990s to reflect the range of each sites waste acceptance periods. For each run, the low and high values were selected from a group of date determined MSW components with no specific location other than the UK. The MSW to soil cover ratio was set at 50% and the resulting output included in Table 5.6 for comparison with the results from the 450mm core samples.

Table 5.6: Comparison of sampled waste components from 8 landfills. Wagland et al. do not provide accurate ages of the waste samples however, deposition at each site is assumed by this study, based on the information provided, to have been undertaken for Sites 1 and 6 during 1980s; for Site 9 during 1990s; for Site 2 during 1990s; for Sites 3, 4 (a and b) and for Sites 7 and 8 during mid-late 1990s.

Site reference	Paper (%)	Glass & Metal (%)	Textile (%)	Plastic (%)	Other (%)	Organic (soil/fines) (%)
1	7.2	2.2	3.6	16.3	6.5	64.2
6	5.5	5.7	10.5	39.0	0.5	38.8
1980s model output	19.2 - 22.2	9.8 - 11.2	1.8 - 2.5	2.2 - 3.1	2.7 - 2.9	58.1 - 64.3
9	3.5	9.8	1.8	8.3	3.7	73.0
2	14.8	3.0	2.5	33.0	3.3	43.5
3	15.0	15.8	3.5	16.2	6.2	43.8
4a	16.5	3.1	3.1	20.0	20.1	37.8
4b	15.5	5.0	6.0	21.5	0.0	52.2
7	16.6	4.8	7.5	30.1	0.0	41.0
8	3.5	17.5	4.0	15.4	12.5	47.1
1990s model output	14.5 - 18.3	7.1 - 7.8	1.8 - 1.9	5.5 - 6.0	2.2 - 2.7	38.8 - 40.2

Notes: i) Data (as % by weight) for each site and material component was measured from the bar chart. ii) glass and metal output from the model have been summed.

Waste age is not given however, based on the published information, deposition is likely to have occurred for Sites 1 and 6 during 1980s; for Site 9 during 1990s; for Site 2 during 1990s; for Sites 3, 4 (a and b) and for Sites 7 and 8 during mid-late 1990s. Landfill 5 has been excluded from the comparison because waste deposition occurred during/after 2007/08 and is outside the scope of this study. The landfills are referenced by number therefore only broad and not specific regional or local modelling for England has been simulated. Similarly, the applied date range is for the range of years the waste was assumed to be landfilled. Accuracy is further reduced by reading waste composition values from the graph presented as Figure 1 by Wagland et al. Unfortunately, without site locations and waste age the model's precision becomes limited with the Model's output comparing, directly, with only 7/54 of the values read from the graph in Figure 1. Furthermore, Wagland et al. use averaged values with error bars providing an indication of the sample spread for each component. For most components, this spread is large and had the comparison been made against these values then a better comparison would have resulted. However, this comparison was made against the (constraining) average values

5.7.6 Comparison to samples taken in European landfills

One objective of this study was to collect and review MSW data for England (and on occasion, the UK) however, English and wider UK landfill excavation records are scarce. For that reason, this section utilizes data from Europe. European MSW differs to that of the UK and is compared and presented in Table 5.7.

Table 5.7: Comparison of UK and European MSW composition taken from sampling exercises undertaken during the 1970s/1980s/1990s (all % composition by weight).

% Composition by weight	PPC (%)	Organic (%)	Glass (%)	Metals (%)	Textiles (%)	Plastic (%)
EU Average[1]	20.0 - 40.0	20.0 - 40.0	0 - 10	0 - 10	-	0-5
Austria 1970s[2]	38.3	18.6	9.2	8.1	7.6	6.1
Austria 1990s[3]	13	31.0	3.0	3.0	4.0	9.0
Belgium 1970s[2]	15 - 30	40 - 45	8 - 16	0.3 - 5.5	1.5 - 2.0	5.0 - 5.5
Belgium 1990s[4]	17.0	47.0	3.0	4.0	3.0	17.0
UK 1970/80s average[5]	27.0 - 45.0	15.0 - 32.0	3.9 - 11.7	0.7 - 5.3	1 - 5.4	1.5 - 5.4
Late 1990s UK average[6]	23.0	37.0	8.0	6.0	3.0	9.0

Notes: 1) Bridgewater and Lidgren (1981), 2) Wilson (1981), 3) Wolfsberger et al. (2015) mean values only, 4) OVAM 2003, quoted in Quaghebeur et al. (2013), 5) Chapter 4 of this study and 6) Parfitt (2002).

Limited data from EUROSTAT and 2 journal publications have facilitated simulations to estimate and compare the materials recovered from 2 European landfills: i) a landfill located in Austria (identified as LFS1) and reported by Wolfsberger et al. (2015) and ii) the large REMO landfill, located in Belgium and reported by Quaghebeur et al., (2013). These publications are introduced and reviewed in Chapter 2 where Wolfsberger et al. propose a historical data methodology to identify the raw material potential within Austrian landfills and Quaghebeur et al. present landfill content data following intrusive site investigation.

Wolfsberger et al. proposed waste composition data could be incorporated into a general methodology to determine the raw material potential within a landfill. The Authors compared the resulting estimates to the results obtained from an analysis of excavated waste that was hand-sorted. The method required input values to be directly applied as output values with specific organic components undergoing biodegradation. There is no allowance for covering soils, reductions that result from recycling or diversion and the products of the degradation are not included. When measured as excavated materials arising from LFM, the given percentages will overstate the percentage quantities excavated although the dry weights of inert materials will remain comparatively similar, assuming dry weights were used during the pre-landfill sampling.

To compare the system dynamics model to Wolfsberger et al.'s results, 2 simulations of the proposed model were run. The first uses the mean MSW composition for regions similar to the siting of LFS1. These data are presented in Wolfsberger et al.'s publication as Table 3. Their basic methodology similarly reflects the approach adopted by this study where alternative, nevertheless similar, original source data is required to be used as averaged or indirect proxy data where specific data does not exist (Henriksen et al., 2019). Wolfsberger et al. identify regional MSW composition data, specific to LFS1, as unavailable. The second data set is that used by Wolfsberger et al. and compiled from the recorded annual weights of different wastes (although broad categories) disposed into cell VA02 of LFS1. These are multiplied by the given percentages for MSW constituents and recorded as Table 4 by the authors. Such disposal data is not available for English historical landfills. Furthermore, it is assumed these same values are those included as weight percentages and presented as Table 9 in their publication and identified as the raw material potential within LFS1.

Wolfsberger et al.'s results, the results from the hand sorting trials and the output from the 2 simulations are presented as Table 5.8. Only categories stated as having raw material potential are presented in the following tables as these are directly comparable to those contained in the model's structure. For this simulation of the system dynamics model, it has been assumed that daily cover materials equate to 40% of imported MSW.

Table 5.8: Comparison of the proposed system dynamics model to the results published by Wolfsberger at al., (2015) for LFS1. Only results for categories having a raw material potential (identified in this study as value materials) are presented as these can be compared directly. Population statistics and data for waste disposed to landfills/incineration were obtained from EUROSTAT.

% Composition by weight	PPC (%)	Composite (%)	Glass (%)	Metals (%)	Textiles & sanitary (%)	Plastic (%)	Soil & fines (%)
Hand sorting trials	3.2	3.8	1.0	4.7	5.7	18.1	47.0
Wolfsberger et al. (2015)	7.0	5.1	3.0	3.2	8.2	8.7	-
Model output using MSW sample data[1]	7.8	3.6	1.8	1.8	7.2	5.4	51.1
Model output using calculated MSW data[2]	5.3	2.4	1.2	1.8	5.3	4.1	49.8

Notes: Data taken from Wolfsberger et al. (2015) 1) Table 3. 2) Tables 4 and 9.

Unfortunately, no details of the hand sorting trials are given by Wolfsberger et al. These would be of value to this study in respect of sample size and numbers particularly if multiple samples were taken whereby a range of values could be identified. However, the use of composition data as inputs, in this case surrogate data, offers a realistic approximation to the hand sorted values. Furthermore, running a simulation using Wolfsberger et al.'s calculated input data (what is referred to by this study as the 'second data set') the system dynamics model improves upon these results when different material categories are compared directly. The system dynamics model proposes a good estimate for the soil/fines content within the landfill which will have a significant impact on the costs of handling materials with no (or limited) resource potential. This is important and necessary information when accessing the feasibility of likely LFM or the costs associated with a landfills remediation.

The REMO Landfill is a large site covering some 150 hectares and accepts MSW and industrial wastes into separate cells. Approximately 16.5 million tonnes of waste has been imported into the site, half of which is household MSW. The data used in this study is that for MSW. To compare the system dynamics model to the sampling results from the REMO site, 2 simulations were run using an MSW to soil-cover ratio of 40% and 30%. The MSW composition inputs comprised data from 1990 sampling taken from the Flemish region of Belgium. The Published data from the REMO site investigation and the 2 simulations are presented in Table 5.9.

Table 5.9: Comparison of model outputs to the results published by: Quaghebeur et al., (2013) for REMO with sampling standard deviations in parenthesis. Population statistics and data for waste disposed to landfills/incineration was obtained from EUROSTAT.

% Composition by weight	PPC (%)	Fines / soil (%)	Glass (%)	Metals (%)	Textiles (%)	Plastic (%)
REMO Location 6[1]	14.0 (8)	45 (18)	0.5 (-)	2.2 (2)	3.1 (5)	25 (13)
REMO study (14 – 29-year-old-waste)[2]	7.5 (6)	54 (12)	1.3 (0.8)	2.8 (1)	6.8 (1)	17 (10)
REMO Location 6 sample variation[4]	6 - 22	27 - 63	0.5	0.2 – 4.2	0 – 8.1	12 - 38
REMO study (14 – 29-year-old-waste)[4]	1.5-13.5	42 - 66	0.5 - 2.1	1.8 – 3.8	5.8 – 7.8	7 - 27
Model output 40% cover[3]	8.24	53.07	1.41	1.77	1.15	9.78
Model output 30% cover[3]	8.66	50.68	1.48	1.86	1.21	10.28

Notes: 1) Waste deposited 1995 – 2000. 2) Waste deposited 1982 – 1997. 3) Waste deposited 1994 – 2001. 4) Estimated sample ranges determined by 1 standard deviation from the mean.

Table 5.9 includes the average sample values (rows 2 and 3) and 2 estimated sample ranges (rows 3 and 4) derived from the published average values and their respective standard deviation. Comparison of each set of the model's output to the average sampled values gives a good fit with the exception of textiles and plastic. When compared to the estimated sample ranges, textiles fit within the range for Location 6 with plastic lying in the range for the 29 year-old waste.

5.7.7 Validity, assessment and data

To this point, the model's validity relies on an assessment of its output. This has been restricted to direct comparison of individual material categories and where these differ. As a validatory method, it is completely reliant upon suitable data to allow such a comparison. With the exception of the comparison to the data published by Wagland et al. (2019), the comparisons in Sections 5.7.4, 5.7.5 and 5.7.6 were realized with suitable data that is, data specific to the age range and siting of each landfill. However, the choice of site was determined by data availability and nothing else. It was for this reason this study included European projects in order to offer a more complete validation of the proposed model although finding data for a specific European region, as Wolfsberger et al. experienced, is not guaranteed. Reference has been made to the lack of UK data both generally and in respect to excavated MSW. Should that data, limited though it clearly is, not have been available any comparison would need to be made against non-specific data. However, data for validation and data for estimation should be treated as separate entities. One objective of this study is to build a data-base for use in the estimation of landfill content of which, the data for Manchester forms only a small part.

Non-specific data might extend to direct comparison with other landfills which are located in a different country to estimate a landfill's composition. In their report to Zero Waste Scotland, Ricardo-AEA[1] (2013) used landfill content data for 2 Swedish landfills presented by Joakim Krook, from the REMO Site and the content based on a 'standard landfill' comprising 500,000 tonnes of waste proposed by van Vossen and Prent (2011).

1. This is not intended as criticism. However, it does emphasise the problem associated with estimating a landfill's resource potential. Comparative data for the UK was not available.

Van Vossen and Prent used data from 60 LFM projects and averaged respective excavated waste compositions where this was available. Their results are presented in Figure 5.14 which includes the system dynamics model's outputs from 3 previous sections for comparison to the proposed 'standard landfill'.

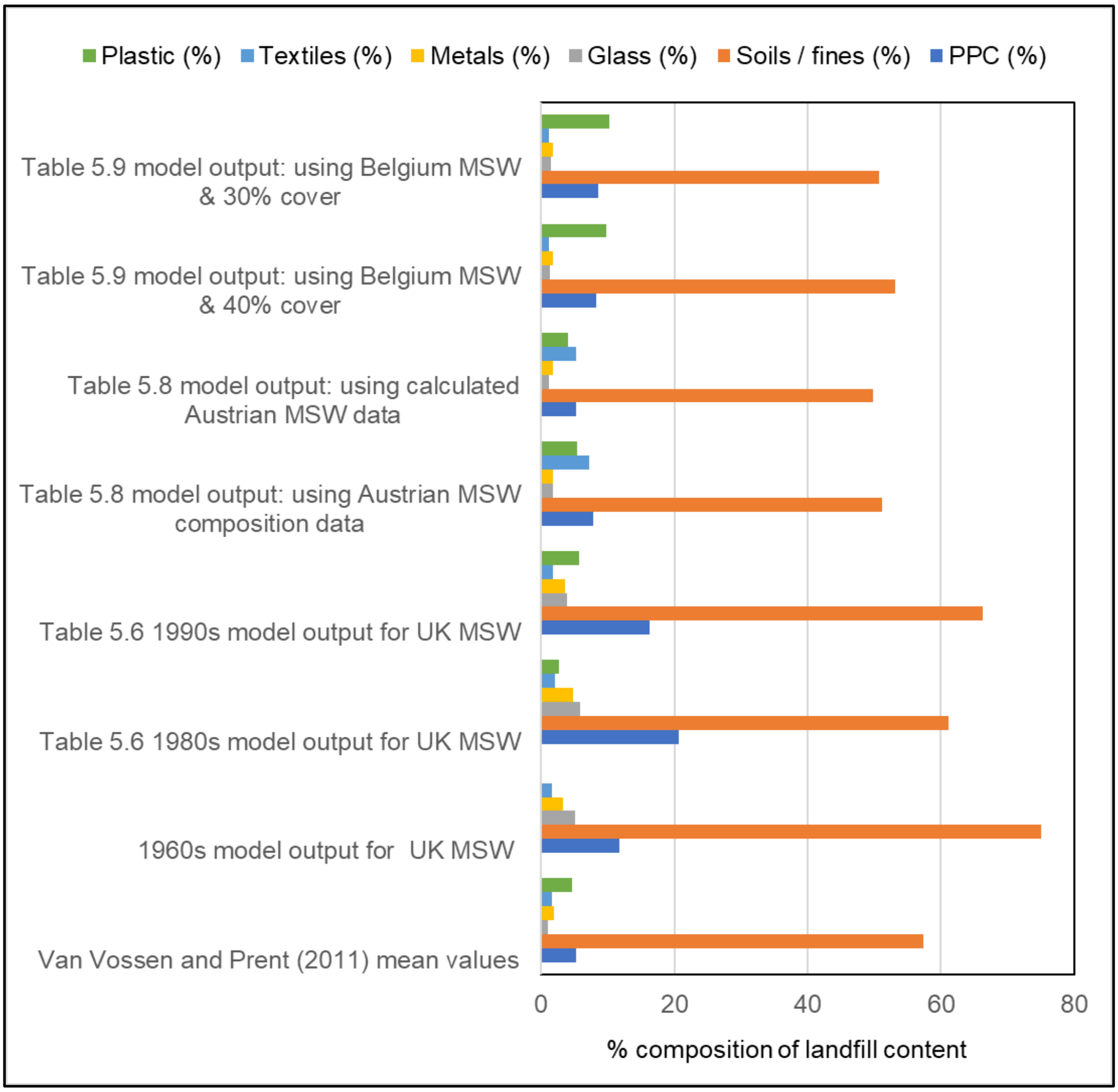


Figure 5.14: Comparison of model outputs for soils/fines and value materials presented in Tables 5.6, 5.8 and 5.9 to the results published by van Vossen and Prent (2011) from their study of 60 landfill mining projects. Model output refers solely to that produced by proposed system dynamics model. The data for Austria and the REMO site were published after 2011. Data for this chart is included in Appendix 5.5

In proposing a standard landfill van Vossen and Prent wanted to *'map the viability of the MFL* (landfill mining*) concept'* because there was no data with which to formulate *'a more accurate composition'*. Whilst Figure 5.14 contains similarities overall, individual values for glass and metals suggest a degree of uniformity while paper and card, textiles and plastic are more problematic. Assessing how well the model output fits the likely content of a landfill will present issues. Published data has confirmed variation in the content within a landfill mass (Baas et al., 2010; Quaghebeur et al., 2013). While metals, paper and card, soils/fines, textiles, and glass compare well to the published content data presented in Figure 5.12 and Tables 5.6, 5.8 and 5.9, model inputs for plastic underestimate the quantity found in samples presented both in the UK and Europe. The OVAM sampling cited by Quaghebeur et al., (2013) identified the plastic content in MSW to have increased from 17% in 1995/96 to 24% by 2001 however, this does not explain the elevated rates found during the mid-late 1990s. Wagland et al. (2019) propose the plastic content in commercial MSW might be an explanation.

Van Vossen and Prent propose their material values to represent a 'standard landfill'. However, as a validatory method using 'bulk' data requires further refinement so as to better reflect changing waste generation. One possibility is to create distinct epochs as this study has proposed and adopted. Data collection and inventory building are now ongoing with the RAWFILL project and WASTEDATAFLOW. A tacit objective of this study is that alternative modelling will be developed to complement this model's output. It is also the intention to further develop the system dynamics approach to identify contaminants in landfills in addition to estimating resource potential. The collection of such data will benefit any model's development.

5.8 Modelling England's landfill content

Identifying the likely quantity of other and/or value materials stored in any landfill is one element of content estimation. There is a similar requirement to estimate other fractions which include the soils/fines content, the overall waste to energy potential and the possibility of hazardous substances including asbestos. Whilst the latter will require the collection and review of specific data beyond this study's terms of reference, the system dynamics model estimates the soils and fines content and identifies value materials as landfill content having resource potential. Whilst the soils and fines are considered residues yet to be identified as offering value, there is a cost associated with their handling and replacement therefore a good knowledge of their quantity is of practical use. The model and literature propose 40 – 80% of materials excavated are so classed (Hernandez Parrodi et al., 2018).

Three material types: i) cinders, ash and clinker, ii) paper and card and iii) food and garden waste tend to dominate the MSW stream. Cinders, ash and clinker add to the soil/fines content of a landfill and, moving through each epoch, their reduction in the MSW stream improves the soil/fines to other/value material ratio such that by the 4th epoch (2000 – 2007), 60% of a landfill's content is comprised from other/value materials. Average soil/fines to other/value material ratios for the 4 epochs are presented in Figure 5.15.

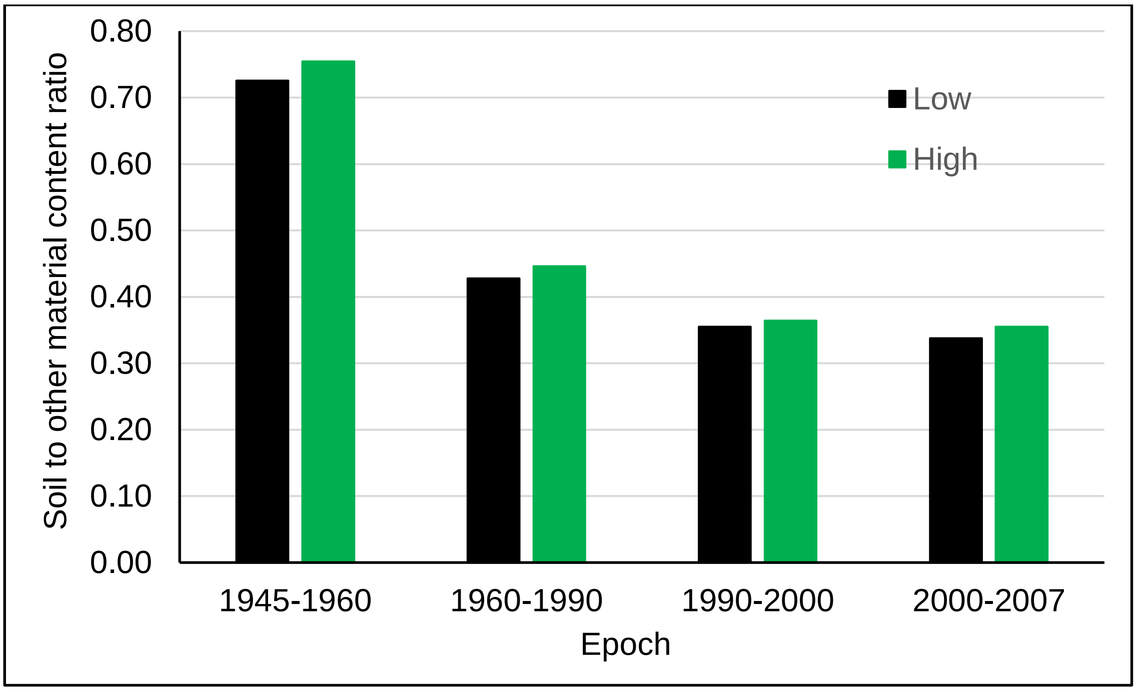


Figure 5.15: Average soil/fines to other material ratios for 4 defined epochs. The high cinders, ash and clinker content of post-war MSW increases the overall soils/fines content of landfills. Modelling was completed using 40% soil cover to MSW.

These data may confirm the anecdotal view that pre-1960s landfills contain little of value. However, this would depend upon the cinders/ash/clinker levels within the imported MSW. Quaghebeur et al., (2013) analysed soil/fines fractions taken from the REMO site for the ash/clinker content. The calorific value was found to be 2.2 – 4.8 MJ/kg dry weight where dry MSW components are typically ≈ 14.8 – 19.3 MJ/kg (Williams, 2005). However, the REMO samples were taken from 1980 wastes and younger. In view of the high ash/clinker content in pre-1960s UK MSW and the methods employed by Manchester Corporation during the late 1950s early 1960s and identified in Section 5.7.4 it may be considered imprudent to simply ignore the possible potential of these sites. The model estimates some 85 M tonnes of cinders/ash/clinker were disposed to landfills during the 1st epoch.

Figure 5.16 (a – f) present modelled estimates of 'other' or value materials stored in English landfills through each 6 decades of landfilling 1945 - 2007.

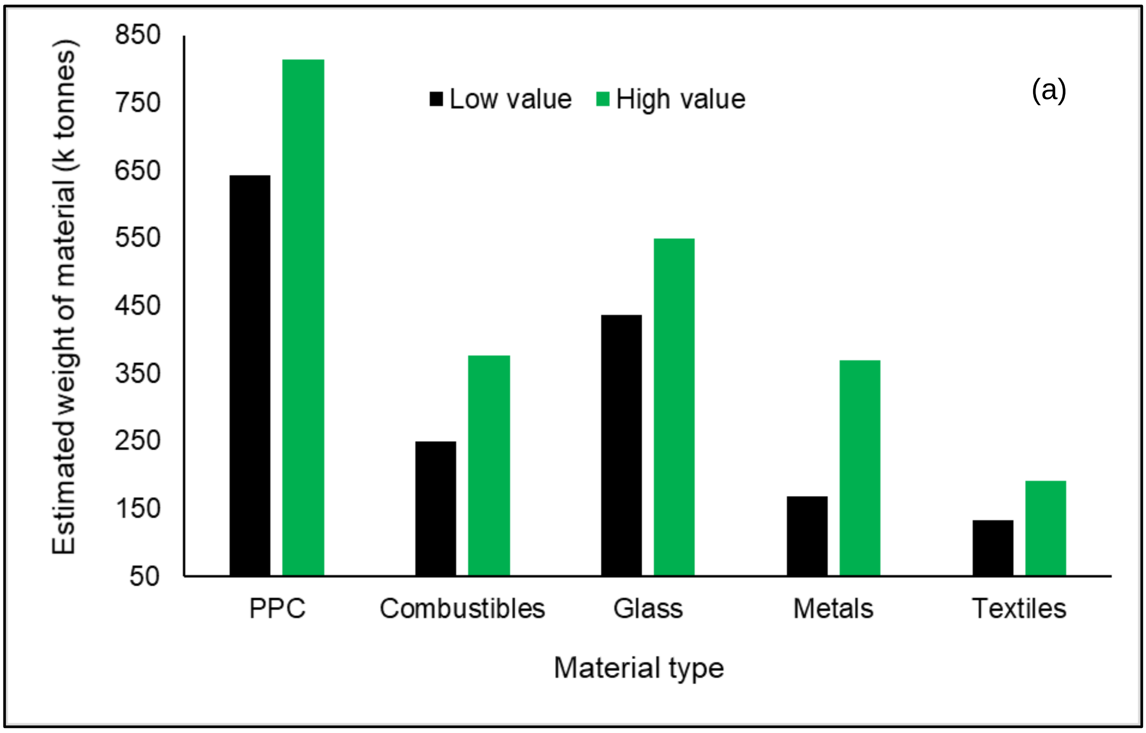


Figure 5.16 (a): Estimated weights of other/value materials stored in English landfills through 6 decades: 1945 – 1960 (PPC is paper, packaging and card).

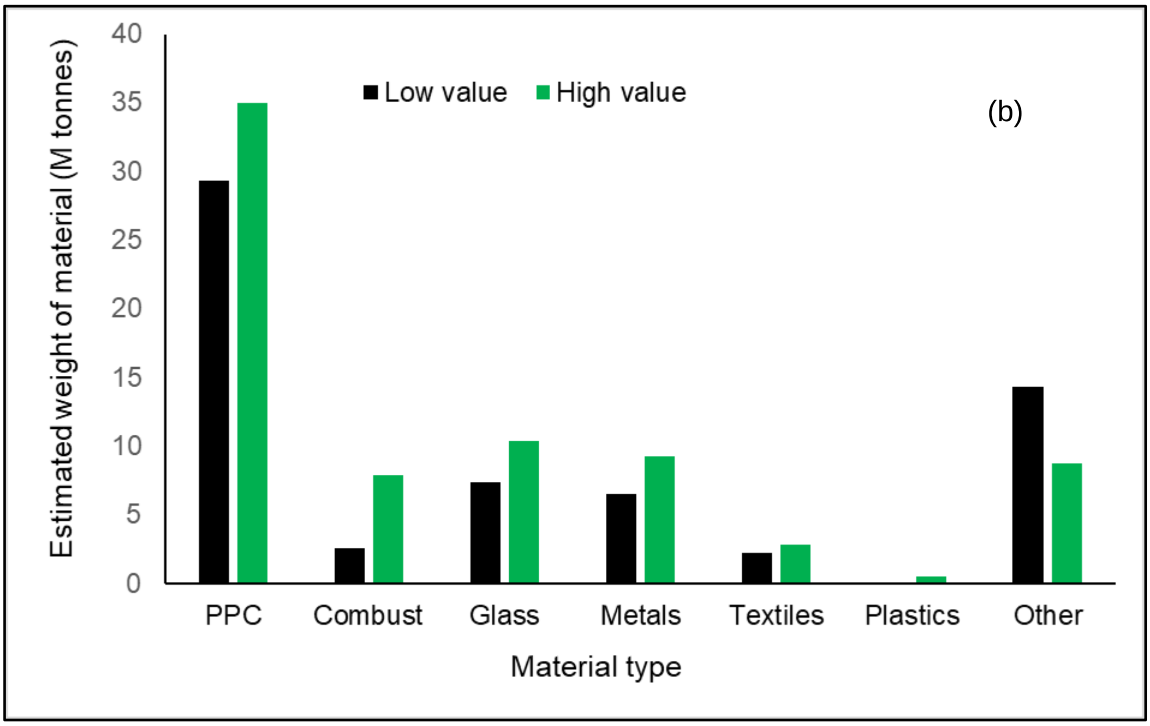


Figure 5.16 (b): Estimated weights of other/value materials stored in English landfills through 6 decades: 1960 – 1970 (Combust is Combustibles).

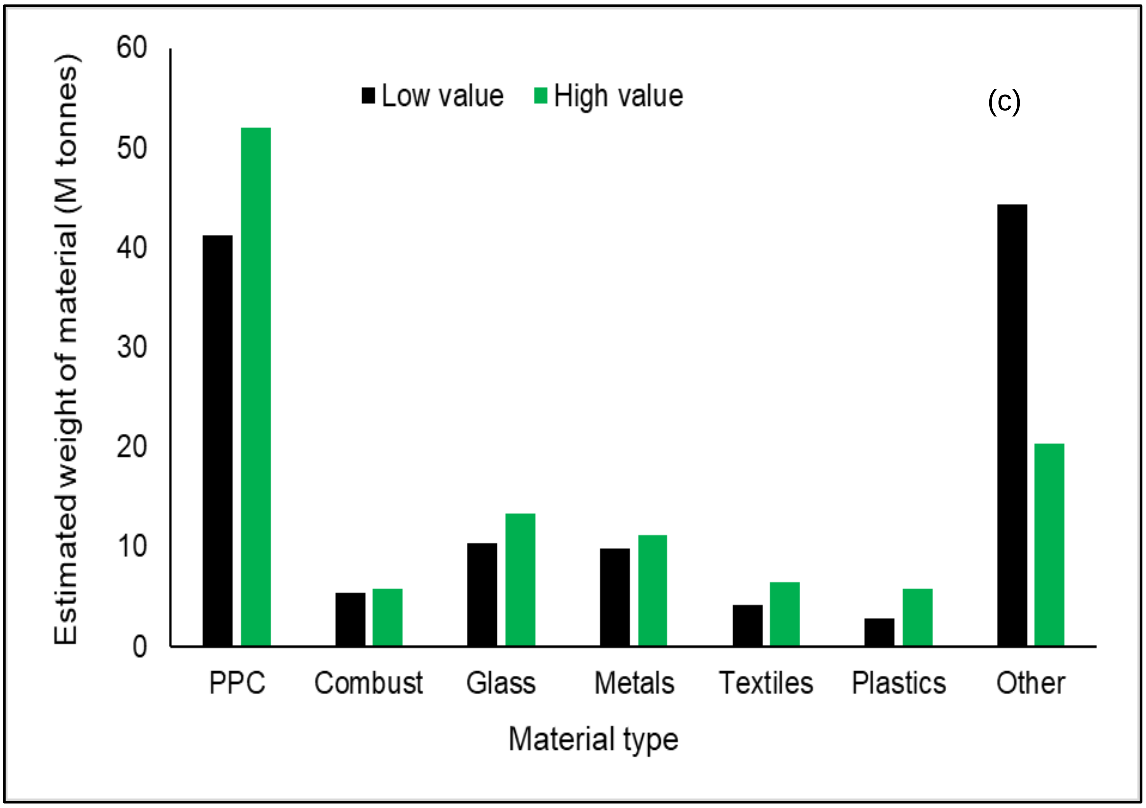


Figure 5.16 (c): Estimated weights of other/value materials stored in English landfills through 6 decades: 1970 – 1980.

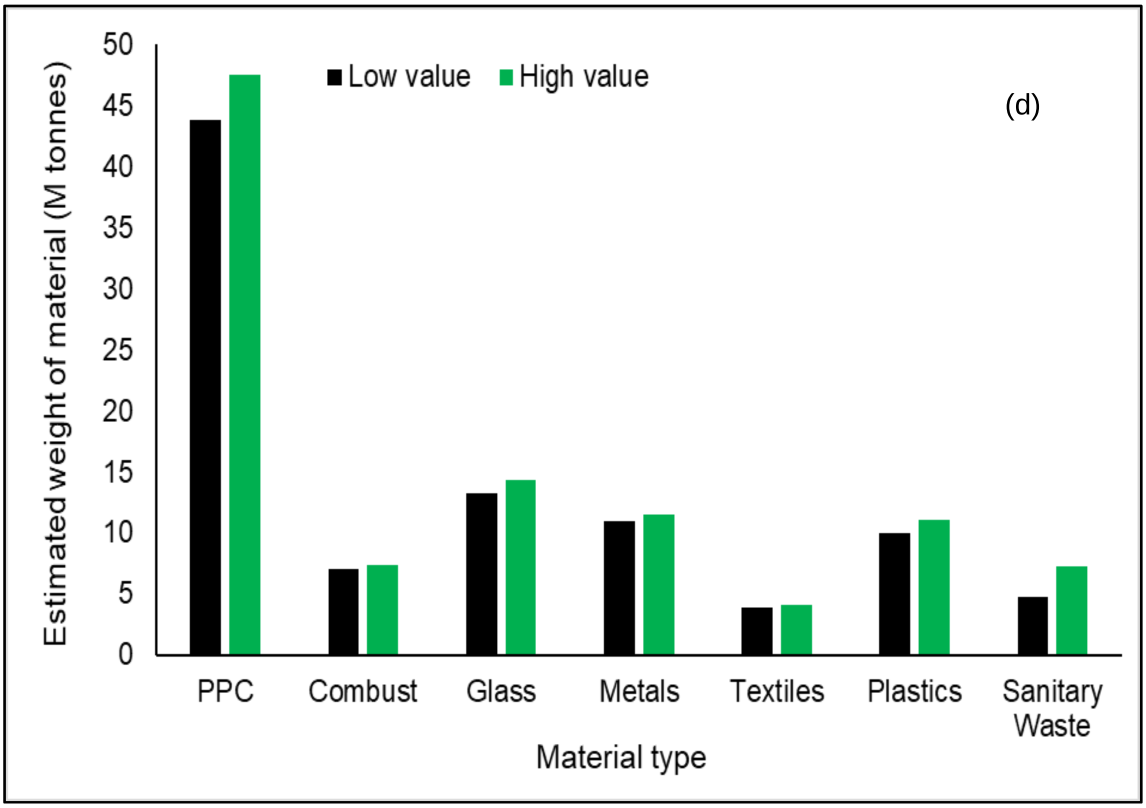


Figure 5.16 (d): Estimated weights of other/value materials stored in English landfills through 6 decades: 1980 – 1990.

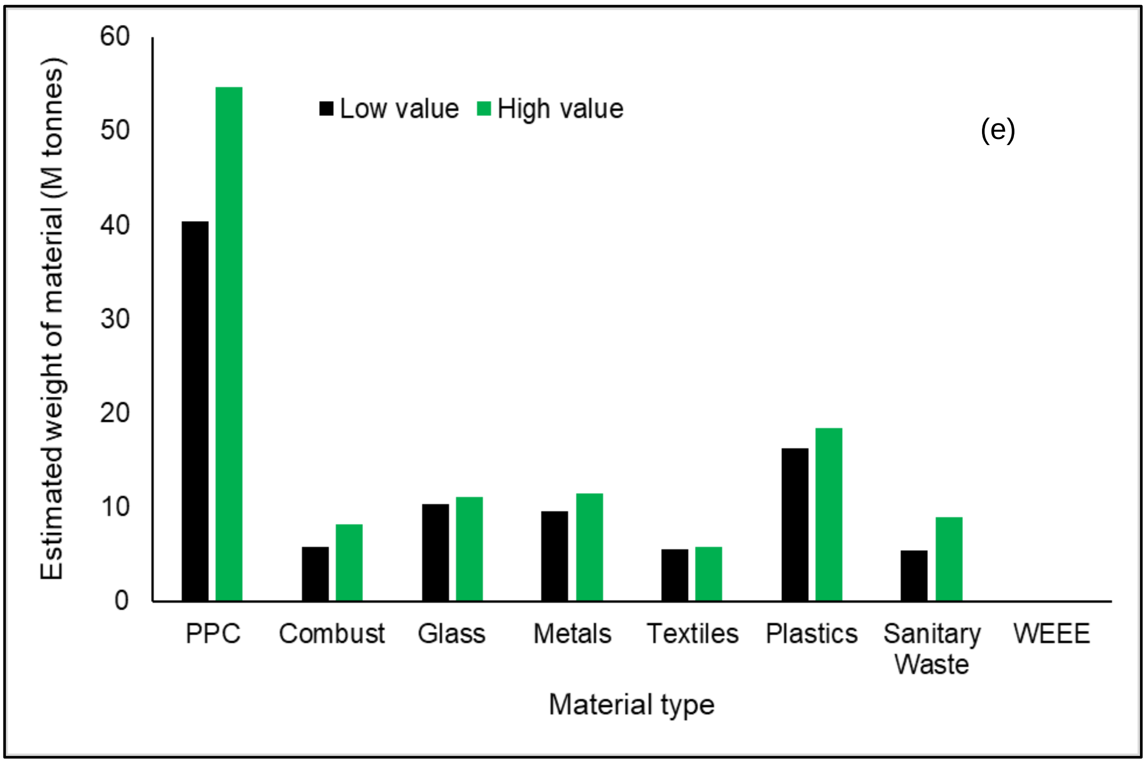


Figure 5.16 (e): Estimated weights of other/value materials stored in English landfills through 6 decades: 1990 – 2000.

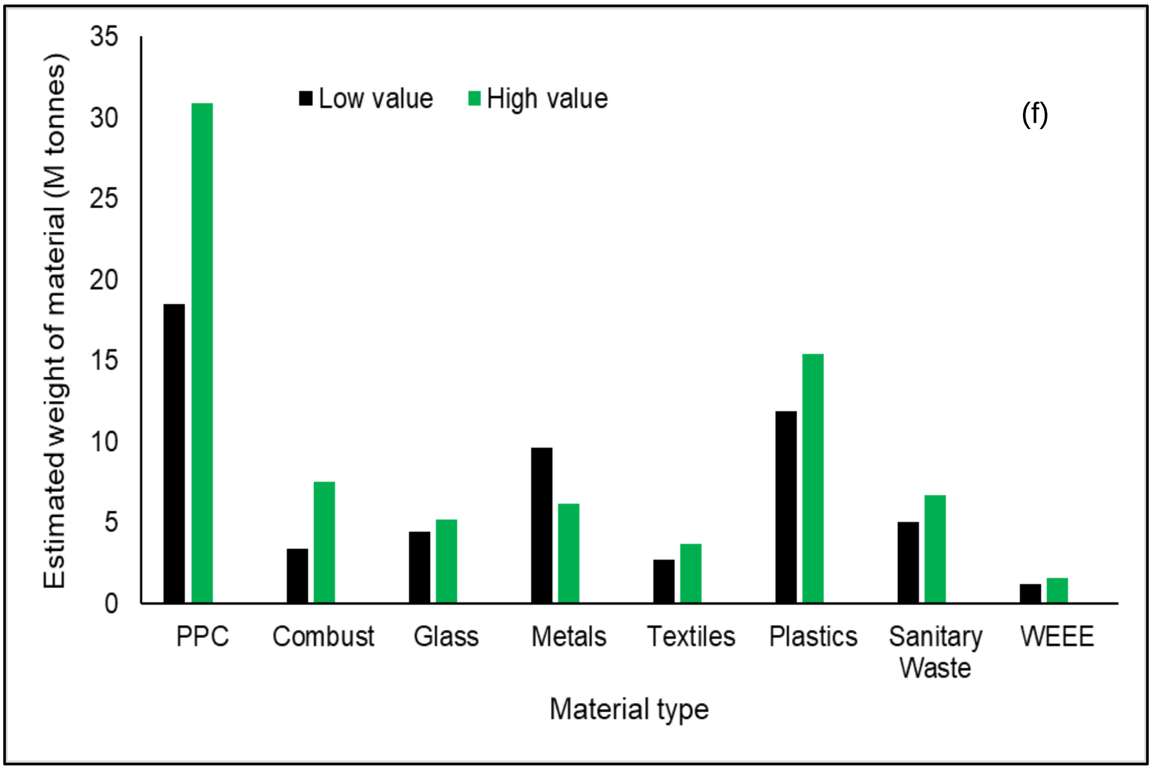


Figure 5.16(f): Estimated weights of other/value materials stored in English landfills through 6 decades: 2000 – 2007.

In terms of resource potential, epochs 2 - 4 present the better opportunities (notwithstanding the comments in respect of cinders/ash/clinker) based on the mix of materials and the developing significance of waste to energy within the UK.

Figure 5.17 (a – d) presents the percentage breakdown of the materials stored in landfills for each epoch.

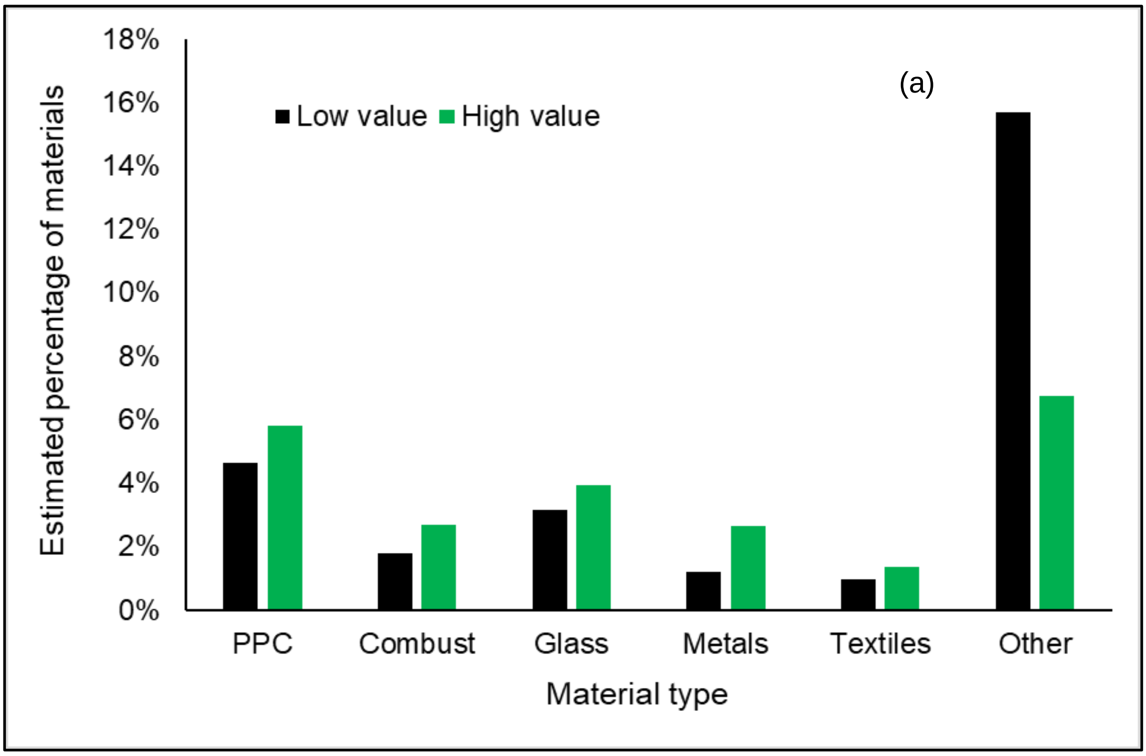


Figure 5.17 (a): Estimated percentages of value and other materials stored in English landfills through 4 epochs: 1st epoch 1945 – 1960 (PPC is paper, packaging and card. Combust is Combustibles). The differences through each epoch result from the changing nature of MSW.

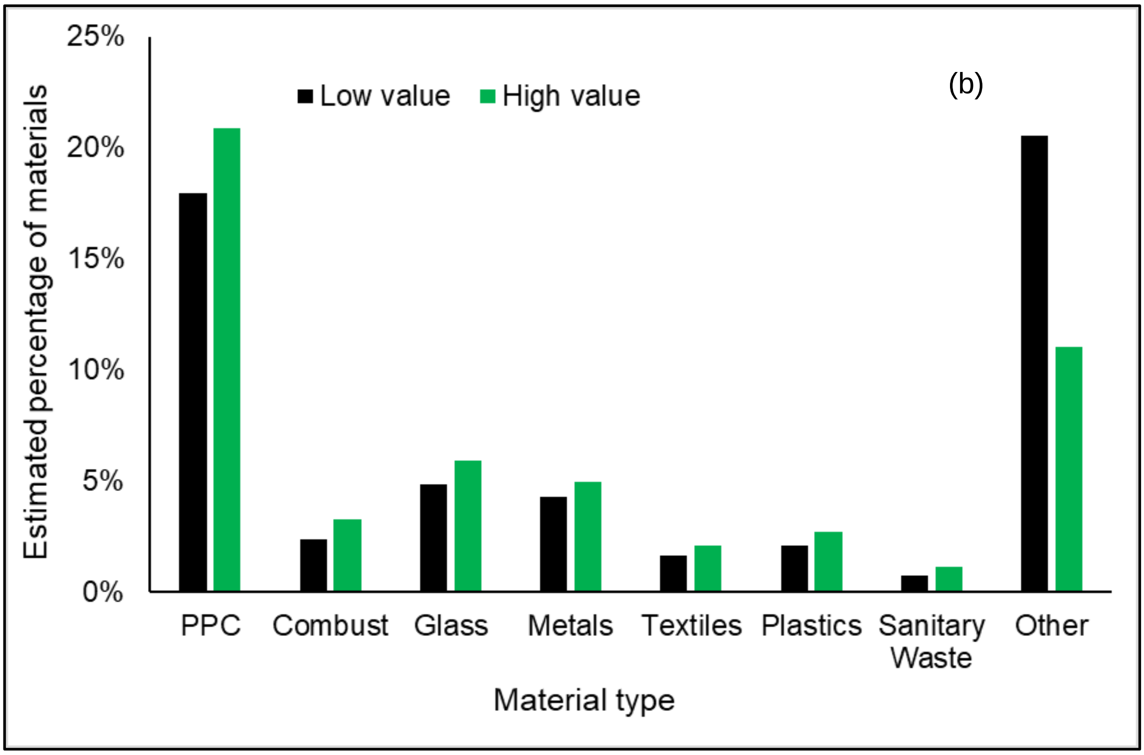


Figure 5.17 (b): Estimated percentages of value and other materials stored in English landfills through 4 epochs: 2nd epoch 1960 – 1990.

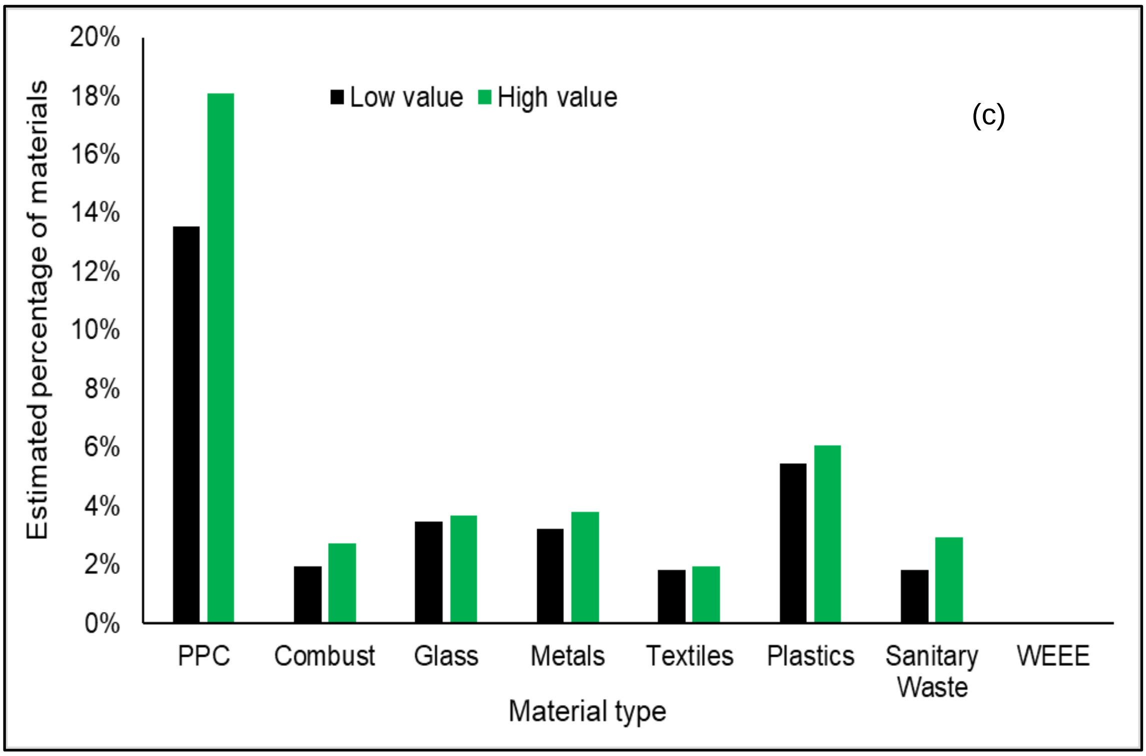


Figure 5.17 (c): Estimated percentages of value and other materials stored in English landfills through 4 epochs: 3rd epoch 1990 – 2000. WEEE is 0.4% and 0.5%.

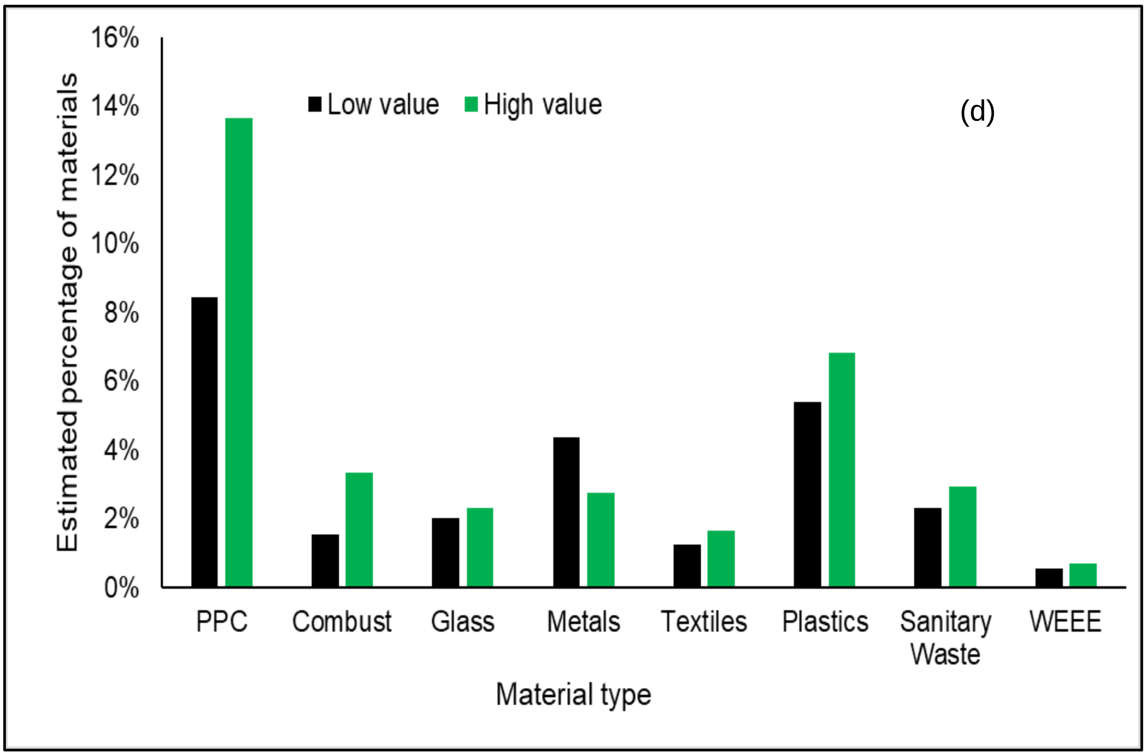


Figure 5.17 (d): Estimated percentages of value and other materials stored in English landfills through 4 epochs: 4th epoch 2000 – 2007.

The differences in material quantities and types through each epoch result from the changing nature of MSW. Paper, packaging and card potentially offer some 3.27 x10^{15} Joules of energy when measured as the landfilled quantity included in

Table 5.2. Figure 5.17 (a – d) presents the percentage breakdown of the materials stored in landfills for each epoch. Food and garden waste are not included in Figures 5.16 and 5.17 as these are decomposed into soil humus, landfill gas and leachate with the remaining soil humus not considered a value component by this study. In addition, the European Union instituted a policy of diversion away from landfill for these biodegradable MSW components as they have an impact on the early chemical/biological processes occurring in landfills. This is considered further in Chapter 6 which proposes a methodology to research early-stage decomposition in non-hazardous landfills.

5.9 Conclusions

A system dynamics model is proposed to estimate the material content in historical MSW landfills. System dynamics modelling assumes a series of stocks and flows as it endeavours to create a 'real world' scenario. The generation and longer-term storage of MSW in landfills are clearly stocks with their transfer from the household or commercial premises a flow. The model structure is arranged into 3 separate sub-units with each sub-unit generating output for a 63-year period. The first 2 sub-units form a basic structure, Tier 1, that uses historical data and mathematical equations to propose quantities of MSW generated and disposed to 4 different disposal strategies where, to this point, these data do not exist.

Three data sets were used as model inputs for the Tier 1 model. Local Authority collected data included only data from LAs weighing 50% or more of generated MSW. LA amended data included all available data and the All MSW data set included the EU definition of MSW. The provenance of the LA collected data set is considered more reliable and these were applied to the Tier 2 model.

The Tier 2 model offers a more complete structure whereby material components are included thus providing quantities for 11 categories of material. The model has the capability to estimate material content in individual landfills, landfills servicing a specific conurbation or region along with generating output at the national level. The Tier 2 model has the capability to vary specific parameters which include the rate of application of cover materials and the degradation product from EU (2008) biodegradable wastes. Output from the modelling produced a range of material quantities determined by the percentage fraction of waste categories entering landfills. These fractions were termed low, average and high and reflected the percentage range of a particular material likely to be stored within a landfill.

Methods for verification and validation of the model are proposed and implemented. To verify the Tier 1 model's output a mass balance exercise was undertaken using the LA collected data set as inputs. The figures generated by the mass balance were identical to the model output although the approach was time consuming and required many spreadsheets. Validation of the Tier 2 output focused on comparisons to landfill mining and specific site sampling data. Two experiments conducted at a landfill located in Manchester produced 2 sets of landfill composition data although there is some concern with the sampling protocols employed with the second set. These data sets were specific to 1958/59 waste imports and the excavated samples from Experiment 1 overlap the model's output for glass, soils and fines, metal, and paper and card and, as such, are good estimators. In addition, generic modelling for 1980s and 1990s MSW was compared to samples taken at 8 UK landfills. Unfortunately, the site locations and waste age were not published which limits the model's precision.

Further comparisons are made with European projects and one generic proposal for a 'standard landfill'. The simulations for these were completed with limited data from journal publications and EUROSTAT. Whilst this study's focus is largely upon data collection for England the use of generic data provides a clear estimate of the content in a landfill but with more specific data the system dynamics model's resolution provides a clear indication of the resource potential and the quantities of bulk materials required to be handled to exploit these value materials.

Data used in this study was collected from a number of different sources and reflects events over disparate periods of the UK's recent history. However, it can be categorized by source, for the most part from 4 primary organisations, the Ministry of Housing, CIPFA, Department for Environment and the Environment Agency (DEFRA). Whilst other sources published data (OECD, World Bank, EUROSTAT), it was derived from one of these four primary sources and was used by this study to fill gaps where primary source data was unavailable.

Furthermore, the data time-line is remarkably linear and does not present conflicting data. Waste generation and disposal data result from records submitted by local administrations or authorities which followed a very similar format and, as the format changed, this was explained and identifiable.

Waste composition data for the study period was limited to 13 material categories, not by this study but by those undertaking the sampling which, in one sense, offers an inherent simplification. However, seasonal and consumer preferences presented a more demanding challenge. Seasonal variation is accommodated with the use of an annual time step. Issues arising from consumer preference are reduced by the implementation of 4 identifiable epochs.

Integration of data as model inputs was supported by structuring the model as 3 sub-units where each sub-unit replicated an element of local authority adopted waste management practice. Such an approach provided a natural linkage between respective sub-units which were then incorporated as flows between stocks, as the general approach with system dynamics modelling.

To answer this study's first research question required the collection and review of historical waste data. Research objective 3 required the assessment of that data's potential as an estimator for the material content in landfills. Assuming the data was sufficient and fit for purpose, Objective 4 was to develop and propose a model able to use that data. To all intents and purposes, the suitability of the historical data and the success of the proposed system dynamics model can be assessed from the estimates of the quantities of waste generated and the results obtained from the comparisons detailed in the previous sections.

Chapter 6: Changing Landfill Environments[1]

6.1 Introduction

Over time, decomposing wastes within a landfill generate a series of differing environments with identifiable characteristics (Rees, 1980; Ehrig, 1983; Hoeks, 1983; Christensen and Kjeldsen, 1983; Stegmann, 1983). The European Commission [EC] has legislated to reduce and control the volume of waste disposed into landfill with the objective of reducing polluting landfill leachates and greenhouse gas emissions (Brennan et al., 2016). This has impacted upon material flows entering landfills in particular a reduction in biodegradable waste.

The Landfill Directive required member states to separate biodegradable wastes and develop recycling (EC 1999). Initially biodegradable waste [BW] included any waste capable of degradation either aerobically or anaerobically. Latterly, BW is defined by the European Commission as garden and park waste, food and kitchen waste from households, restaurants and caterers, retail premises and food processing plants (EC, 2008) and forms a rapidly degrading component of municipal solid waste (MSW). In addition to BW, MSW contains organic fractions where the projected biodegradation time, when occurring within landfills, extends to many centuries (Rathje and Murphy, 2001; Jones, 2008; Muaaz-Us-Salam et al., 2020). Latterly, further regulation now requires waste to be considered a resource and to further reduce biodegradable fractions disposed into landfill to 65% of their 1995 level (EC 2008, EC 2011). Municipal waste generation in the United Kingdom has increased over the period 1995 – 2014 from 28,900,000 to 31,131,000 tonnes per annum. However, the annual quantity landfilled has reduced from 23,990,000 to 8,656,000 tonnes (EUROSTAT, 2016). However, with the introduction of significant control measures, leachable contaminants remain within the landfill mass beyond the 100-year time horizon and may exist for a period of many centuries (Belevi & Baccini, 1989; Christensen et al., 1996; Manfredi & Christensen, 2009).

1. This chapter is based on the previously published paper: **Warwick S**., Duany-Fernandez P., Sapsford D., Cleall P. and Harbottle M. 2018. Altered chemical evolution in landfill leachate post implementation of biodegradable waste diversion. *Waste Management & Research,* 36 (9) pp. 857 - 868.

To implement the Landfill Directive 31/1999 in the UK the Environment Agency sought to significantly change landfill management methods and reduce the proportion of biodegradable municipal solid wastes (MSW) disposed into landfills. This represents a major shift in waste disposal strategy where landfilling now resides at the base of the waste hierarchy. Furthermore, at the personal and local authority level, many fractions from municipal solid waste streams are now regarded as recyclable.

6.2 Relevance to the aims and the objectives

This chapter describes research work undertaken to answer, in part, the second research question: *Do significant correlations between sampling data and waste content exist?* together with research objectives five and six*: to investigate the physical and chemical relationships between decomposition processes, leachate samples and landfill mass content* and: *to determine whether landfill leachate composition has changed as a result of changes in landfill practice*.

One of the challenges with this study was the necessary collection of monitoring data from operational landfills. A number of national and regional operators were contacted but only two responded favourably. For the UK, landfill monitoring was first directed in Waste Management Paper 27 Landfill gas (DoE, 1991a) now superseded by LFTGN02 (2003) and leachate monitoring by LFTGN03 (2003). Monitoring data and waste returns from operational landfills in England are required to be provided to the Environment Agency on a quarterly and annual basis (HM Government, 1994; Environment Agency, 2017). From a review of data supplied by Viridor for two landfills in the South West of England, this study was able to publish data demonstrating the intended objective of reducing biodegradable waste inputs into landfills and how this led to a successful evolution in landfill decomposition dynamics.

6.3 Background

6.3.1 Landfill processes

MSW decomposition in landfills is driven by a series of physical, chemical and biological processes (Christensen & Kjeldsen, 1989). Cossu (2019) considers the landfill to a black box reactor where the $Accumulation$ of contaminant mass over time can be described by a summarised mass balance equation:

$$Accumulation = IN - OUT - Reacted \qquad (6.1)$$

Where IN represents the incoming waste stream, OUT is the mass of emissions released as a result of decomposition and $Reacted$ is the mass of stabilised organic waste. Emissions are liquid, known as leachate, and a range of gaseous emissions which, for the greater part, include carbon dioxide and methane. Landfill leachate occurs when the field capacity of the waste mass is exceeded due to continued water infiltration.

Landfilling methods generate a brief aerobic period where oxygen becomes trapped within the waste matrix. With the onset of the biodegradation of the putrescible content, the rapid consumption of oxygen by bacteria leads to anaerobic conditions which develop into a sequence of consecutive phases. Rees (1980) and Ehrig (1983) identified the existence of different bio-chemical environments, and these occurred as a result of varying carbon sources in waste streams. Rees' paper included a review of operating practices at Aveley Landfill where a high gas generation rate was attributed to refuse compaction and the importance of a high-water content (55%) in the compacted landfill matrix. Ehrig identified decreasing concentrations of pollutants in landfill leachate as the waste mass aged. Christensen and Kjeldsen (1989) proposed a five-phase (idealized) degradation sequence with no time units other than the first of the phases, the short aerobic phase which, in most cases, is measurable in days. Phases 2 – 4 are anaerobic with phases 2 and 3, forming the 1st and 2nd intermedial phases. Phase 2, the acetogenic (or sometimes, fermentative) phase is characterised for the production of weak organic acids and carbon dioxide. During phase 3 methanogenic bacteria dominate, methane production commences, the pH increases, and sulphates decrease. Phase 4 establishes a period of stable methane generation with landfill gas [LFG] containing between 50 – 65% methane. Over time, with the reduction in available carbon, methane production reduces in the final phase to the point where diffusion from the atmosphere by nitrogen into LFG allows it to become the dominant species. Small aerobic zones appear in the upper matrix of the landfill as air penetrates into the surface layers.

Resulting from laboratory experiments researching this latter process, theoretical inference led to an eight-phase model, Figure 6.1, being proposed where the 5th phase continues through three additional phases (Christensen et al., 1996). Air mixing in the upper matrix oxidises slow-produced methane from lower levels of the matrix into carbon dioxide. These are the air intrusion phase (5), the methane oxidation phase (6), the carbon dioxide phase (7) and finally phase 8, the soil air phase. This extension to the original 5-phase decomposition model occurs very slowly taking decades or even centuries (Belevi & Baccini, 1989; Christensen et al., 1996; Manfredi & Christensen, 2009).

Each phase is identifiable by a distinct set of chemical and biological processes which influence the chemistry of the leachate. During the acetogenic phase, fermentative and acetogenic bacteria generate carboxylic acids, carbon dioxide, hydrogen and alcohols from proteins, carbohydrates and lipids occurring mainly in putrescible wastes (Rees, 1980, Christensen & Kjeldsen, 1989). For the acetogenic phase and following initial disposal into landfill, BOD and COD levels are greatest (Doedens & Cord-Landwehr, 1989) and the acidity of the leachate can reduce pH to 4.5 (Ehrig, 1983). In UK climatic conditions, transition from the acetogenesis phase to a stable methanogenic phase may not have occurred until after three years (Robinson, 2005).

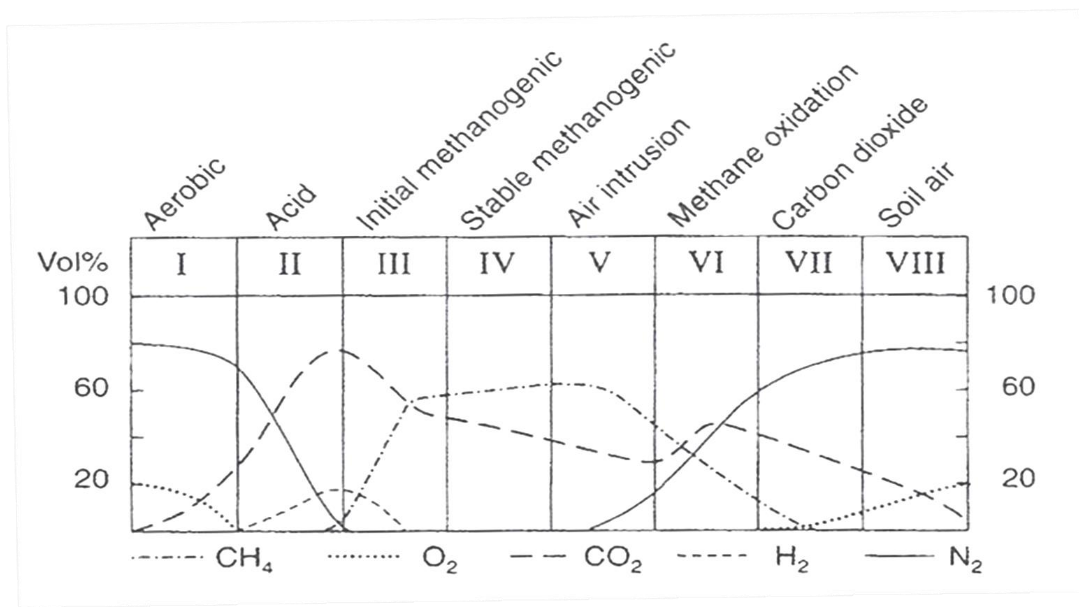


Figure 6.1: Decomposition phasing with gases generated. Figure reproduced from Christensen et al. (1996 p.31).

This phase is followed by the methanogenic phase where solid organic materials are microbially degraded producing methane as a significant product. Leachate pH typically increases above neutral to an average value of 8 (Ehrig, 1983).

Leachate generation is a consequence of water infiltration. The presence of water is necessary for the degradation process both as a reactant (equations 5.2 – 5.6) and, in a physical way, a liquid transport medium. For engineered landfills this transport medium allows the designed removal of contaminants dissolved in landfill leachate and facilitates gas generation, also collected, preventing their uncontrolled release into the environment (Cossu, 2019; Cossu et al., 2019). Historic landfills do not have these measures and, for this reason, are sources of environmental contamination. Figure 6.2 presents a simplified representation of carbon breakdown first as a result of hydrolysis and then acidification.

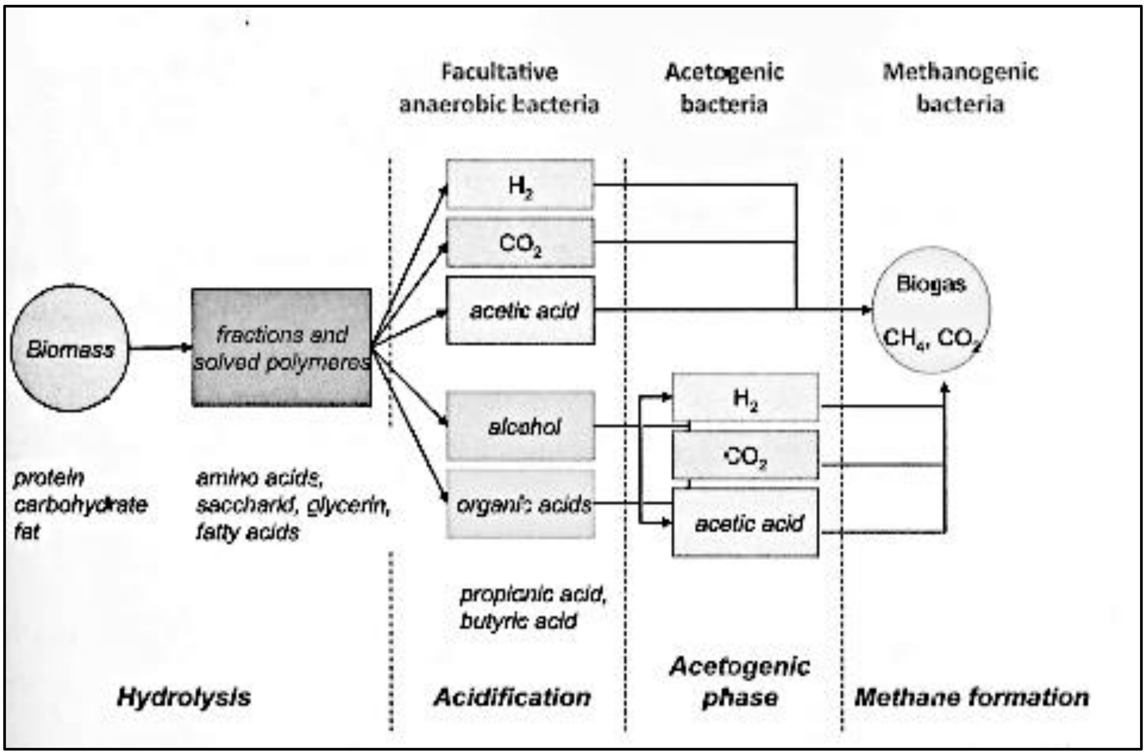


Figure 6.2: Simplified landfill degradation process. Figure reproduced from Cossu et al. (2019 p.97).

Biodegradable organic materials (carbohydrates, fats and proteins) in putrescible MSW waste streams provide carbon and energy for a range of bacteria which, under anaerobic conditions undergo a series of hydrolytic reactions:

$$Carbohydrates + nH_2O \rightarrow C_6 H_{12} O_6 \qquad (6.2)$$

$$Fats + nH_2O \rightarrow C_3 H_5(OH)_3 + R - COOH \qquad (6.3)$$

$$Proteins + nH_2O \rightarrow nR + CNH_2 - COOH \quad (6.4)$$

$$nR + CNH_2 - COOH \rightarrow nRCOO^- + nNH_4^+ \quad (6.5)$$

In the degradation process this is the rate limiting step, slowing the overall rate of degradation and can be considered the primary reason for the EU's objective for the removal of putrescible carbon from MSW streams. Many reactions occur, in particular from the fermentation of glucose which result in the generation of carbon dioxide and the production of a range of organic acids including acetic acid (6.6), butyric acid (6.7) and alcohol (6.8). Whilst these acids are not specifically tested for during leachate monitoring they create a reduced pH environment, hence acetogenesis, which is monitored:

$$C_6H_{12}O_6 + H_2O \rightarrow 2CH_3COOH + 3\mathrm{H}_2 + CO_2 \quad (6.6)$$

$$C_6H_{12}O_6 \rightarrow C_3H_7COOH + 2H_2 + 2CO_2 \quad (6.7)$$

$$C_6H_{12}O_6 \rightarrow 2CH_3CH_2OH + CO_2 \quad (6.8)$$

These reactions reflect only part of what is taking place in a decomposing landfill. The landfill matrix should be considered to comprise a large number of microenvironments however, it is likely the acidic environment inhibits the development of methanogenic bacteria (Christensen and Kjeldsen, 1989).

6.3.2 Biochemical indicators for acetogenesis and methanogenesis

Where landfills receive domestic and commercial MSW, the landfilling practices employed and the chemical, physical and biological processes that result have led to leachates comprising four groups of components: heavy metals, inorganic macrocomponents, dissolved organic matter and xenobiotic organic compounds (Christensen et al., 1994). Established key process indicators in landfill leachate are pH, BOD5/COD (ratio) and sampled concentrations in mg/litre for sulphates, calcium, magnesium, manganese, iron and zinc (Ehrig, 1983). Identification of a particular phase of landfill degradation relies primarily upon comparisons made between leachate samples with concentrations lying within specific ranges which are then determined as being indicative of either the acetogenic or methanogenic phases (Ehrig, 1988; Kjeldsen et al., 2002). Their usefulness as indicators relies upon the fact that respective concentrations change significantly during the landfill decomposition process resulting from the two contrasting chemical environments occurring first during acetogenesis and second, during methanogenesis. Those components that do not change, which include the inorganic fractions of ammonium and the chloride ion content, have been excluded by Ehrig. Ehrig's published concentrations are contained in Table 6.1. The sample average and sample range are provided.

Table 6.1: Sample concentrations for 10 key indicators in the acetogenic and methanogenic phases. All data values in mg/l with the exception of pH and BOD5/COD. Table reproduced from Ehrig (1988 p.31).

Key indicator	Acetogenic phase		Methanogenic phase	
	Average	Range	Average	Range
pH	6.1	4.5 – 7.5	8	7.5 - 9
BOD_5	13000	4000 – 40000	180	20 - 550
COD	22000	6000 – 60000	3000	500 - 4500
BOD_5/COD	0.58		0.06	
Sulphates	500	70 – 1750	80	10 - 420
Calcium	1200	10 – 2500	60	20 – 600
Magnesium	470	50 – 1150	180	40 – 350
Iron	780	20 – 2100	15	3 – 280
Manganese	25	0.3 - 65	0.7	0.03 - 45
Zinc	5	0.1 – 120	0.6	0.03 - 4

6.3.3 Changes in UK waste composition and disposal policy

For the UK, the Landfill Directive 31/1999 represented a major shift in waste disposal strategy. Waste policy and implementation at the local level now required that many fractions from municipal solid waste (MSW) streams should now be regarded as recyclable and were required to be diverted away from landfills. For the greater part, MSW can be categorized as waste collected at the kerbside and includes household and similar wastes generated by commercial, educational and governmental organisations (Burnley 2006). Reported statistics issued by the Department for Environment, Food and Rural Affairs (DEFRA) identify MSW as comprising food and garden waste, paper and packaging, wood, metal, WEEE, furniture and sanitary (solid) waste (DEFRA, 2009). Many of these wastes are classified as biodegradable however, respective degradation periods are substantially different with some having significant durations when compared to EU defined BW which occurs within a one to three-year period.

Data published by EUROSTAT reflects the United Kingdom's (UK) obligation under the Landfill Directive to redirect wastes away from landfill noticeably the diversion of biodegradable municipal waste [BMW] containing wastes that have led to food and garden wastes becoming a feedstock for energy production via digestion and composting to the point where weights of materials recycled now exceed those landfilled. Waste quantities reported to EUROSTAT (2017a) recorded a reduction in BMW to landfill from 29,030,00 tonnes, a theoretically derived figure, in 1995 to 10,339,000 tonnes in 2010 to 6,843,00 tonnes in 2014. Figure 6.3 compares MSW generation and evolving disposal strategies over the period 1995 – 2015.

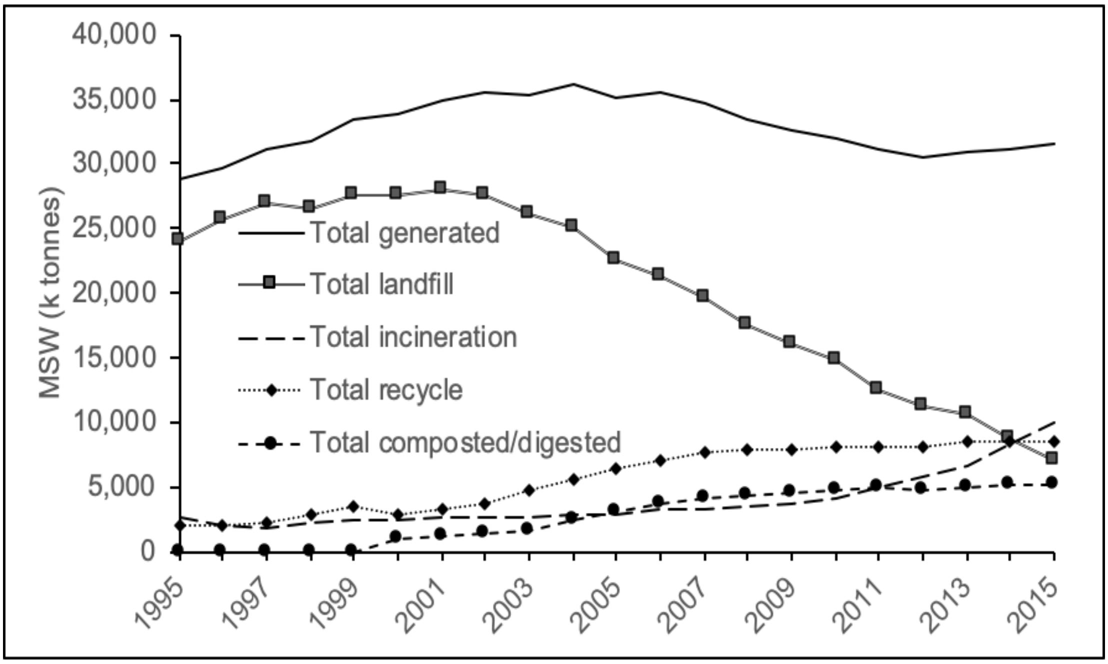


Figure 6.3. Comparison of changing UK MSW disposal methods 1995 – 2015. (Data from EUROSTAT 2017b).

6.4 Methodology and data sources

6.4.1 Methodology

The method adopted to analyse leachate data was a graphical approach using schematic boxplots. To plot the boxplots the statistical package Minitab has been used throughout this study also to analyse data. Boxplots separate data into quartiles with the two ‘inner’ quartiles forming the box. The lower and upper quartiles are represented by two tails or ‘whiskers’. Minitab defines outliers as those observed values that exceed one and a half the interquartile range, with these being denoted with a star (*). For each variable, the median is indicated by the (selectable) horizontal line within each interquartile range. The analysis and boxplots are included in Section 5.5.

6.4.2 Data sources

Leachate data has been provided by the UK waste management company, Viridor Waste Management Ltd [Viridor]. It should be noted that Viridor provided significant data which is currently being used for a wider study and the site and data labelling reflects that used in the wider study. All data have been taken from engineered landfill sites where waste disposal operations commenced in 2005 (identified as Site C) and 2007 (identified as Site B). Viridor provided a range of data which includes individual sampling point or well-point data (B11, B12, C2 and C3), composite leachate data together with deposited waste records. Leachate composition sampling occurs monthly and, following analysis, are submitted to the UK Environment Agency on a 3-monthly basis. Testing and analyses specifications are identified in Section 6.4.3. (Table 6.3) with individual site detail included in Section 6.4.4. The test house is accredited with the National Measurement Accreditation Service.

These data are compared to Ehrig’s key indicators together with data from two homologous and established engineered landfill sites, Sites E and Site F. Samples from these sites are composite leachates which firmly reflect the methanogenic phase of degradation when compared to Ehrig’s values. Table 6.2 summarises the operational commencement dates, primary waste source, the sample range and current site status for the case-study landfills. The four sites are located in the South West Region of England. Herein the South West region comprises the UK counties of Cornwall, Devon, Dorset, Gloucestershire, Somerset and Wiltshire. The South West region was categorised by the 2011 census as being mainly rural in respect of rural-urban typology. At the 30-kilometre cell level recorded dwellings per hectare were below 0.83 with Bristol and its surrounding area recording 1.90 – 3.41. (Bibby and Brindley, 2013).

Sites B and C have been chosen specifically because old waste was not present at either site. This removes the possibility of new leachates passing through old waste layers. Where underlying waste layers are in the methanogenic phase, leachates generated from overlying new wastes, that pass through these methanogenic wastes, will reflect the characteristics of the older (methanogenic) leachates (Kjeldsen et al.,1998; Assmuth, 1992).

Table 6.2: Operational commencement, waste source and date range for the 4 Viridor sites.

Site	Operation commenced	Waste source	Sample range	Status
Site B	2007	urban	2007 2015	open
Site C	2005[1]	rural	2006 2015	open
Site E	1988	urban	1995 2015	closed[2]
Site F	1994	rural	2004 2015	closed[3]

Notes: 1) Waste deposition commenced 2006 and was only 10,157 tonnes. 2) Year of final waste import – MSW 2009, cover materials 2012. 3) Year of final waste import – MSW 2010, cover materials 2010.

6.4.3 Leachate samples

The leachate data provided forms part of the Viridor's leachate sampling and management systems as regulated by Schedule 2, Paragraph 2(1)(c) of the Landfill Regulations (2002). As such, the sample sizes are small being dictated by the requirements of Schedule 2, Paragraph 2(1)(c) and the company's testing regime. A further point, and a necessary one, is the fact that landfilling operations have only recently commenced at both sites thus providing a unique opportunity to examine the impact of diverting biodegradable wastes away from landfills.

Leachates were extracted from individual monitoring well-points. A list of the determinants used in this study and relevant analytical procedures is provided in Table 6.3

Table 6.3: Analysis methods and reporting limits for leachate quality at the four Viridor sites.

Determinant	Method Description	Reporting Limit	Reporting Units
pH	WAS039 pH/ EC in Water by Electrode	1	pH units
BOD + ATU (5 day)	WAS001 BOD in Water	1	mg/l
COD (Total)	WAS040 COD in Water by Colorimetry	11	mg/l
Sulphate as SO_4	WAS036 Anions by Colorimetry	4.4	mg/l
Calcium, Total as Ca	WAS049 Metals in Water by ICP-OES	0.38	mg/l
Magnesium, Total as Mg	WAS049 Metals in Water by ICP-OES	0.6	mg/l
Manganese, Total as Mn	WAS049 Metals in Water by ICP-OES	0.007	mg/l
Iron, Total as Fe	WAS049 Metals in Water by ICP-OES	0.23	mg/l
Zinc, Total as Zn	WAS049 Metals in Water by ICP-OES	18	mg/l

6.4.4 Site descriptions

SITE B: Site Design and Construction

The Northern Extension has been operational since 2007 and is a fully engineered landfill site. The 32-hectare site currently comprises a total of 10 sub-phases developed on the principal of engineered containment, with a basal lining system comprising an artificial liner comprising a geosynthetic clay liner (GCL) and a reworked and natural geological barrier of in situ Alluvium. The phases are subdivided into elongate sub-cells by permanent clay bunds of 2m height. Each sub cell is filled using a series of elongate ‘tipping areas’. It is proposed to cap the Northern Extension with a LLDPE geomembrane, overlain by 800mm alluvium and 200mm topsoil. The design principles of the Northern Extension are based around both engineered and hydraulic containment of leachate within the landfill site during the operational and aftercare phases of the site life. During the site construction and development, groundwater underdrainage beneath the site will locally influence the naturally occurring groundwater regime that was present before site development, which will re-establish once dewatering operations have ceased. The average waste depth is 13.0 metres with waste deposition undertaken in 2.5-metre layers.

SITE B: Leachate Management

Leachate collection and control measures have been installed within each cell in the form of leachate monitoring and abstraction boreholes and aggregate drainage system. Leachate from both areas of the landfill is pumped to the raw leachate balance tank for on-site treatment. Each cell has a unique leachate sampling/extraction point identified on the site plan. Only leachate samples extracted from those well points indicated were used in the study. Furthermore, leachate recirculation was not installed at the time of sampling (2007 – 2011). Figure 6.4 identifies the leachate sampling points for Site B.

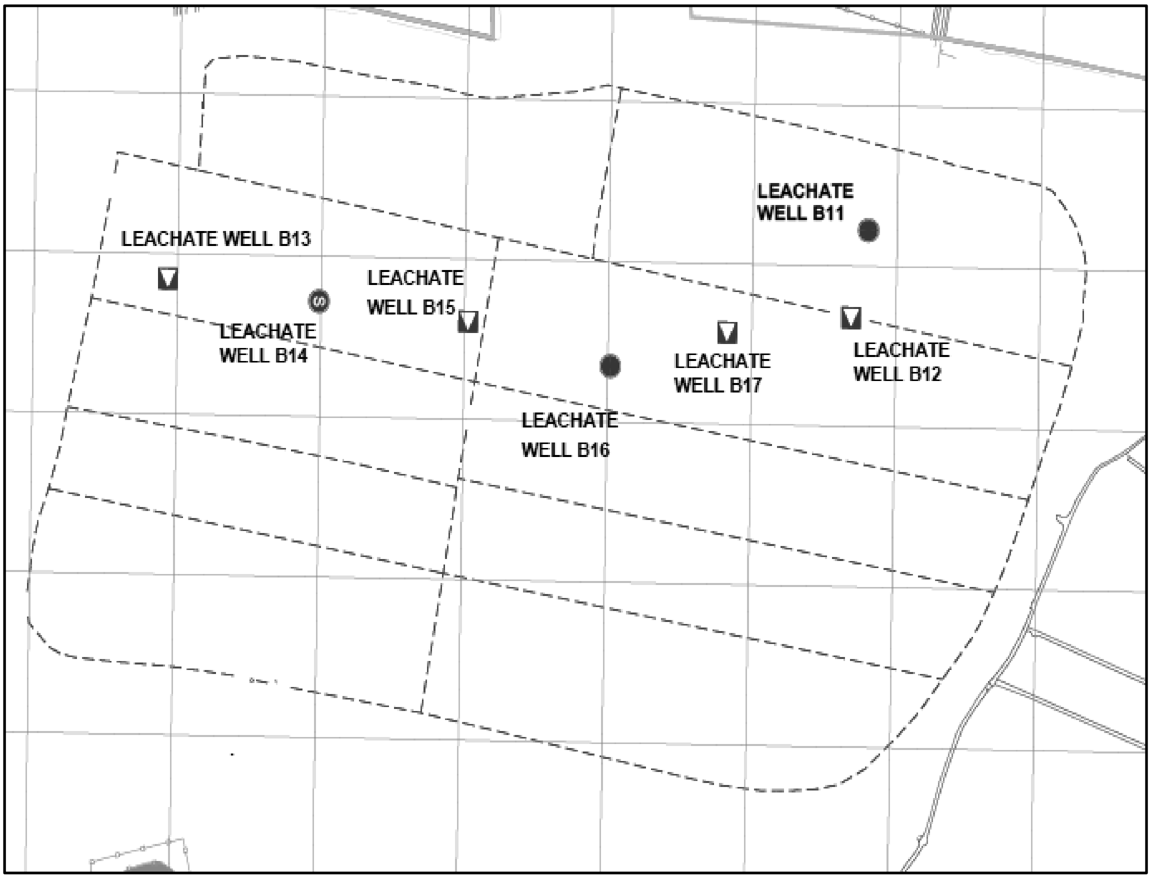


Figure 6.4: Site B Northern Extension Plan. Individual landfill cells and site perimeter are indicated by the broken lines. Leachate sampling points/locations used in the study are included. The grid lines are at 100 metre spacings.

SITE B: Gas Collection System (GCS)

The current GCS comprises vertical gas wells connected to gas mains and manifolds. Wells have typically been installed on twenty to forty metres spacing. Extraction is generally provided by a manifold system with wells connecting individually into inlets on the manifolds. Wells have generally been drilled to 375mm-450mm diameter and installed with a suitable standoff from the pit base.

SITE C: Site Design and Construction

Site C has been operational since 2006 as a fully engineered landfill site. This 26-hectare site currently comprises eleven engineered containment cells (Cells A to L). Cells A to E have been developed utilising 0.75m of low permeability clay plus a 0.25m protection layer, overlying in-situ Lias Clay. Cells E to J have been developed utilising 1m of low permeability clay with a maximum permeability of 2x10-10m/s. Where limestone bands have been encountered within a metre of the base of cells, then the upper 0.5m of Lias Clay has been replaced with clay that achieves a maximum permeability of 5x10-10m/s. The average waste depth is 10.0 metres with waste deposition undertaken in 2.5-metre layers.

SITE C: Leachate Management

Leachate collection and control measures have been installed within each cell in the form of leachate monitoring and abstraction. An installed drainage system delivers raw leachate to the remote sampling points before being pumped to the raw leachate balance tank for on-site treatment at the leachate treatment plant. Only leachate samples extracted from sampling points C1, C2 and C3 were used in the study. Composite leachate samples were not included. Furthermore, leachate recirculation was not installed at the time of sampling (2006 – 2011).

SITE C: Gas Collection System (GCS)

The current GCS comprises vertical gas wells connected to gas mains and manifolds. Wells have typically been installed on twenty to forty metres spacing. Extraction is generally provided by a manifold system with wells connecting individually into inlets on the manifolds. Wells have generally been drilled to 375mm - 450mm diameter and installed with a suitable standoff from the pit base. The general site layout is given in Figure 6.5.

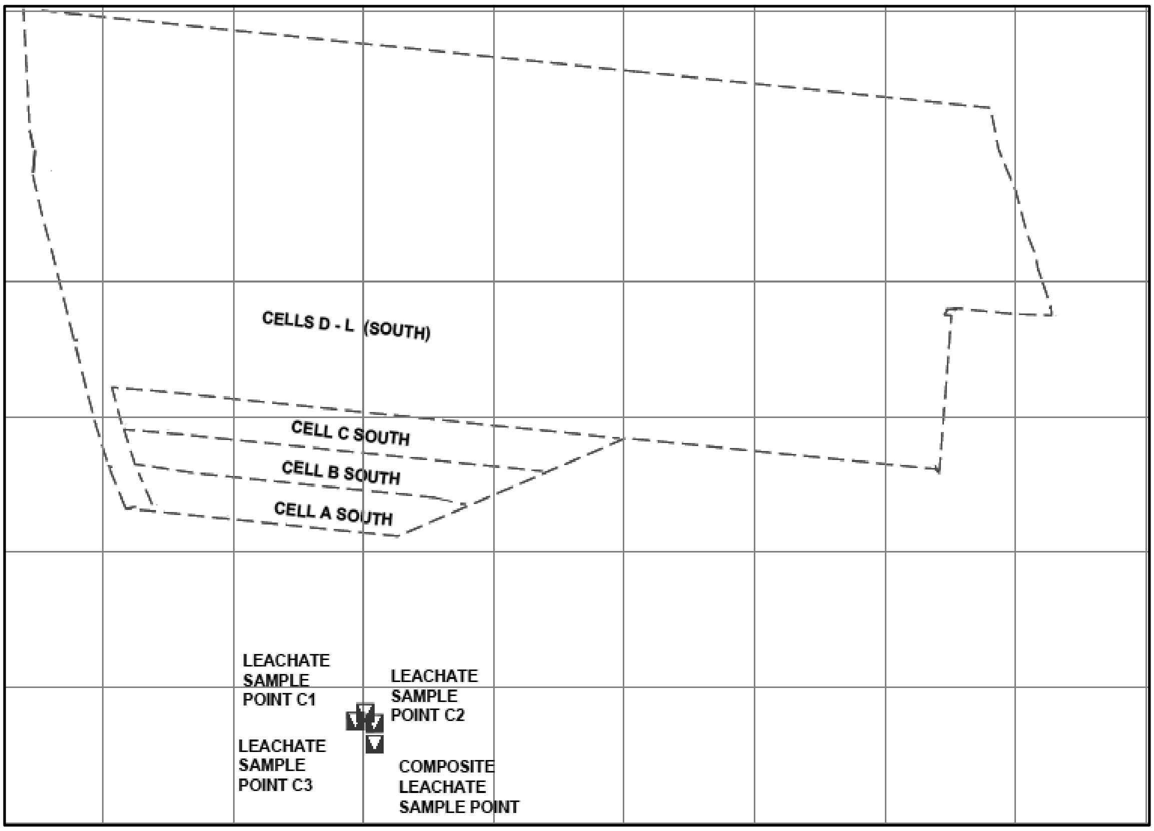


Figure 6.5 Site C Plan. Individual landfill cells and site perimeter are indicated by the broken lines. The cell boundaries for cells D, E, F, G, H, I and L have been excluded to improve the plan's reproducibility. Leachate sampling points/locations used in the study are included. The grid lines are at 100 metre spacings.

6.4.5 Waste types imported into the four study sites

Wastes imported into Sites B, C, E and F are summarized in Table 6.3. Imported waste data for all sites commences from 2005. Viridor categorise imported wastes using the List of Waste codes identified in Technical Guidance WM3 (EA 2015) having (first) separated waste imports into those high-level categories identified within the table. Waste imports are recorded in tonnes.

Waste imported into each of the study sites can be classified as: cover materials, for the most part soils and transfer station fines; MSW comprising mixed domestic and non-domestic wastes, transfer station wastes and non-special clinical wastes; difficult wastes, which include sewage works screenings, printer toner waste and green wastes; sludges comprising mainly septic tank and sewage sludge residues; and non-hazardous wastes comprising contaminated (non-hazardous) soils. Site B was also used to dispose approximately forty-six thousand tonnes of asbestos-containing materials which for the most part was contained within construction wastes.

Table 6.4: Summary of waste sources for the four Viridor sites.

	Site B tonnes '000 (%)	Site C tonnes '000 (%)	Site E tonnes '000 (%)	Site F tonnes '000 (%)
Total waste imported	2,077.10	1,264.00	1,136.20	58.2
Cover materials	682.9 (32.9%)	142.7 (11.3%)	317.2 (27.9%)	11.9 (20.4)
MSW - domestic & non-domestic	1012.0 (48.7%)	983. (77.8%)	450.8 (39.7)	35.8 (61.6%)
Difficult wastes	1.4 (0.1%)	6.2 (0.5%)	1.6 (0.1%)	-
Sludges	46 (2.2%)	0.97 (-)	1.01 (0.1%)	9.6 (16.4%)
Non-hazardous waste	16.2 (7.7%)	5.8 (0.5%)	-	-
Other waste	174.6 (8.4%)	126.1 (10.0%)	365.6 (32.2%)	0.9 (1.5%)

6.5 Results and discussion

6.5.1 Analysis of leachate samples from Sites B and C

The leachate samples are presented as a series of boxplots. Specific chemical markers (the key indicators) pH, BOD, COD, BOD/COD together with the sulphate and calcium concentrations are compared to Ehrig's ranges for both the acetogenic and methanogenic phases. Each boxplot represents the collated annual sampling data taken from individual well-points.

Metal concentrations for both sites were very low and are not included as boxplots. Mean values for Mg, Mn, Fe and Zn for Site B were 140 mg/l, 0.53 mg/l 2.97 mg/l and 0.11 mg/l respectively. Mean values for Site C, in the same order, were 125 mg/l, 0.37 mg/l, 2.28 mg/l and 0.10 mg/l. Metal concentration in leachate increases with decreasing pH (Aucott, 2006). Concentrations at these low levels seen in the data are typical of the methanogenic phase where the pH is in excess of 7.5 (Ehrig, 1988; Kjeldsen et al., 2002) and reinforces the proposal that acetogenesis is not occurring. Figure 6.5 (a and b) compare combined pH data for both sites where the median pH exceeds 7 for all but one sample set.

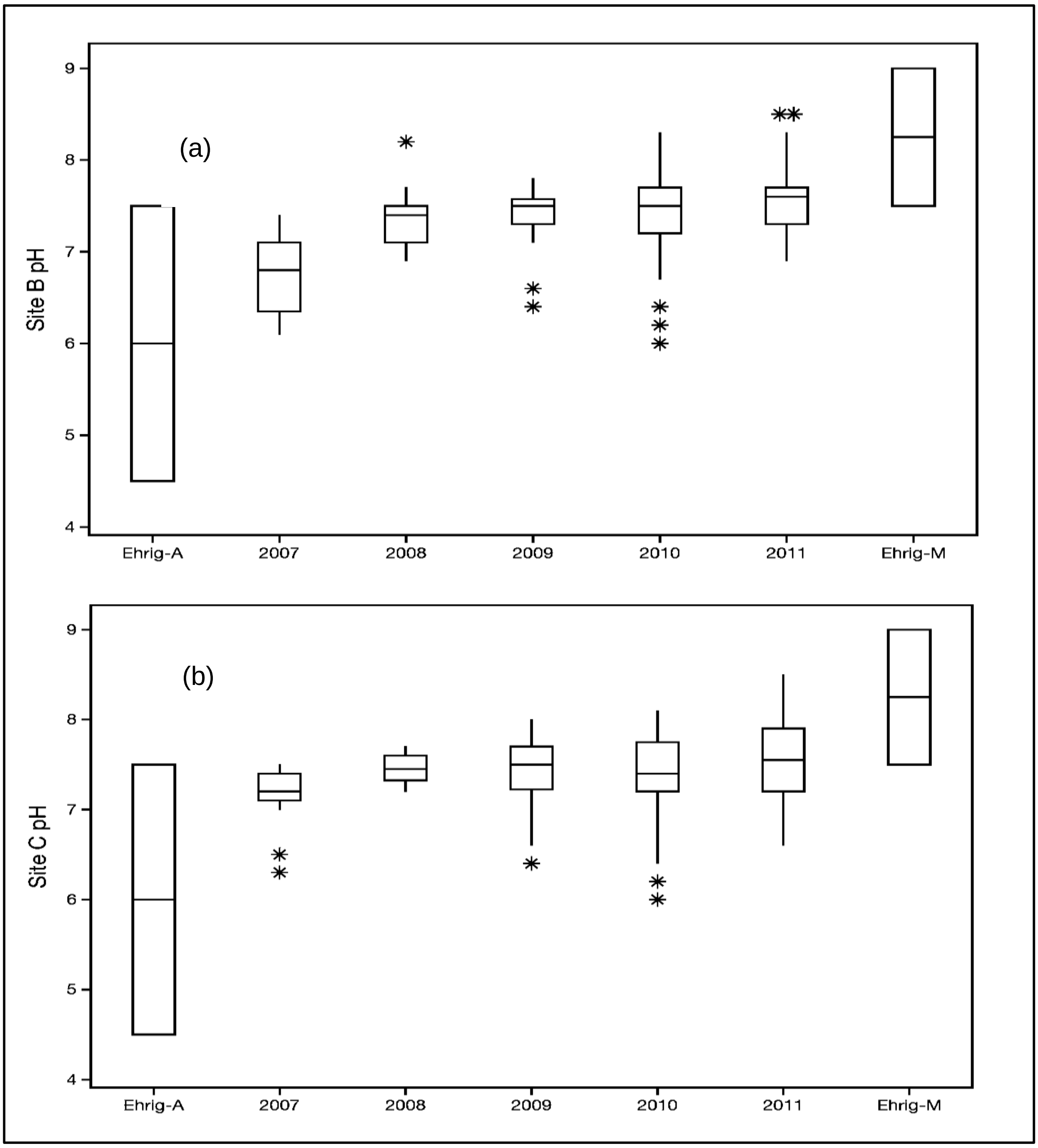


Figure 6.6 (a & b): pH for Sites B & C from 2007 to 2011 compared to Ehrig's data in the acetogenic and methanogenic phases. The pH for each site at this point in MSW decomposition is expected to reflect weak acidic conditions (pH 4.5 – 7.0) where the mean values tend to weak basic condition.

For these new waste deposits and the conditions prevailing in the UK, the pH of leachate samples should reflect weak acidic conditions (pH 4.5 – 7.0) throughout the transition of acetogenesis to the establishment of methanogenesis, approximately 4 years (Robinson, 2005). Figure 6.7 a and b presents Individual wellpoint (B11) pH data for Site B where the sampled data highlights the initial period where pH values are less than neutral, 6.1, 6.4 and 6.6 for March, April and May 2007, respectively. By September 2007 and following an initial period of reduced (slightly acidic) pH during 2007 all sampled values lie above 7 and generally reflect values expected during the methanogenic phase (basic conditions) which are expected to occur after 4-years of decomposition.

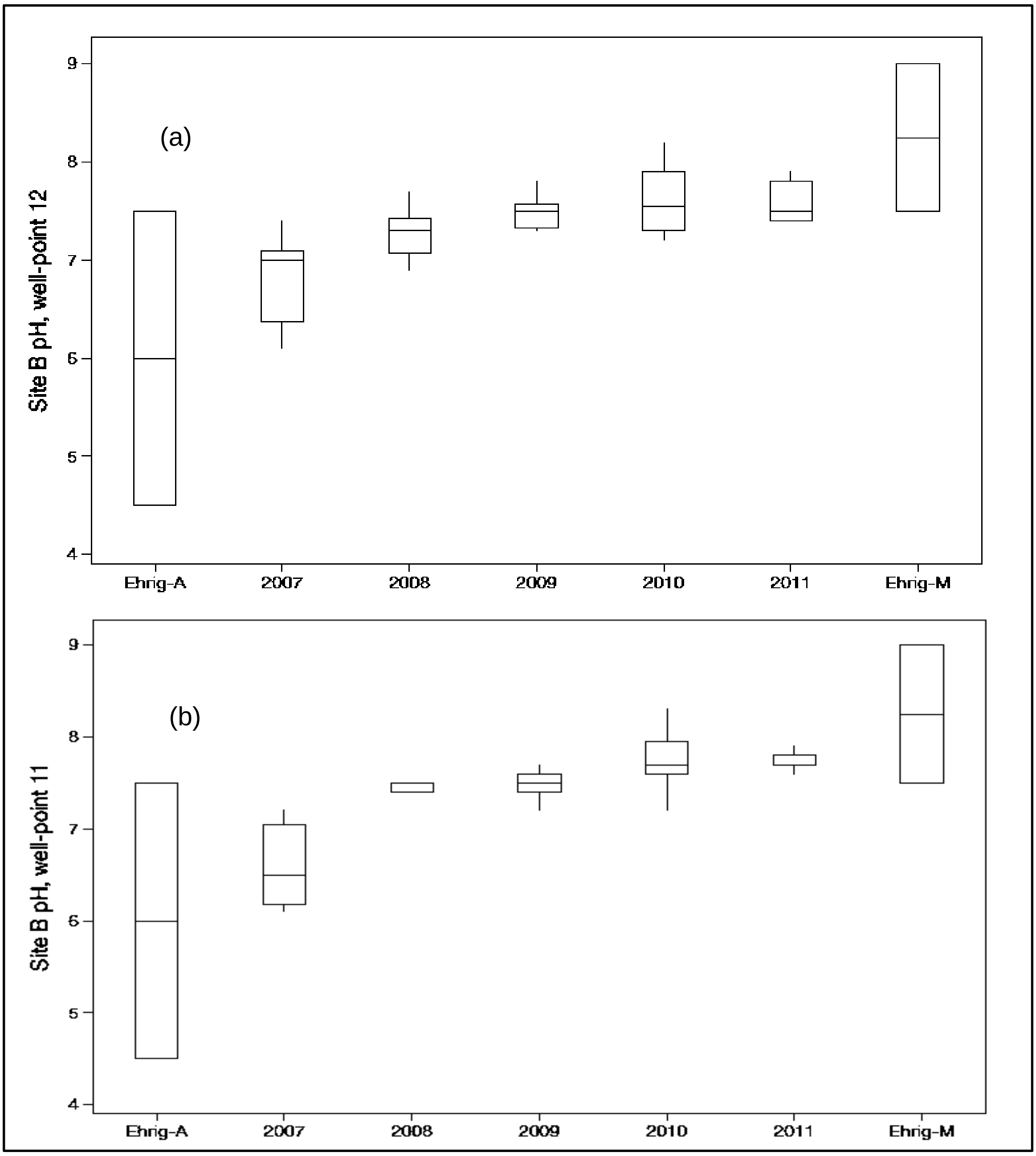


Figure 6.7 (a & b): pH for wellpoints 11 & 12, Site B from 2007 to 2011 compared to Ehrig's data in the acetogenic and methanogenic phases. After an initial reduced pH condition (during 2007) all values lie above 7 and generally reflect values expected during the methanogenic phase.

For the remaining key indicators, the leachate samples continue to reflect the methanogenic range identified by Ehrig and are presented as Figures 6.8 – 6.10. The BOD_5 and COD test results (Figure 6.8) are highly methanogenic and would

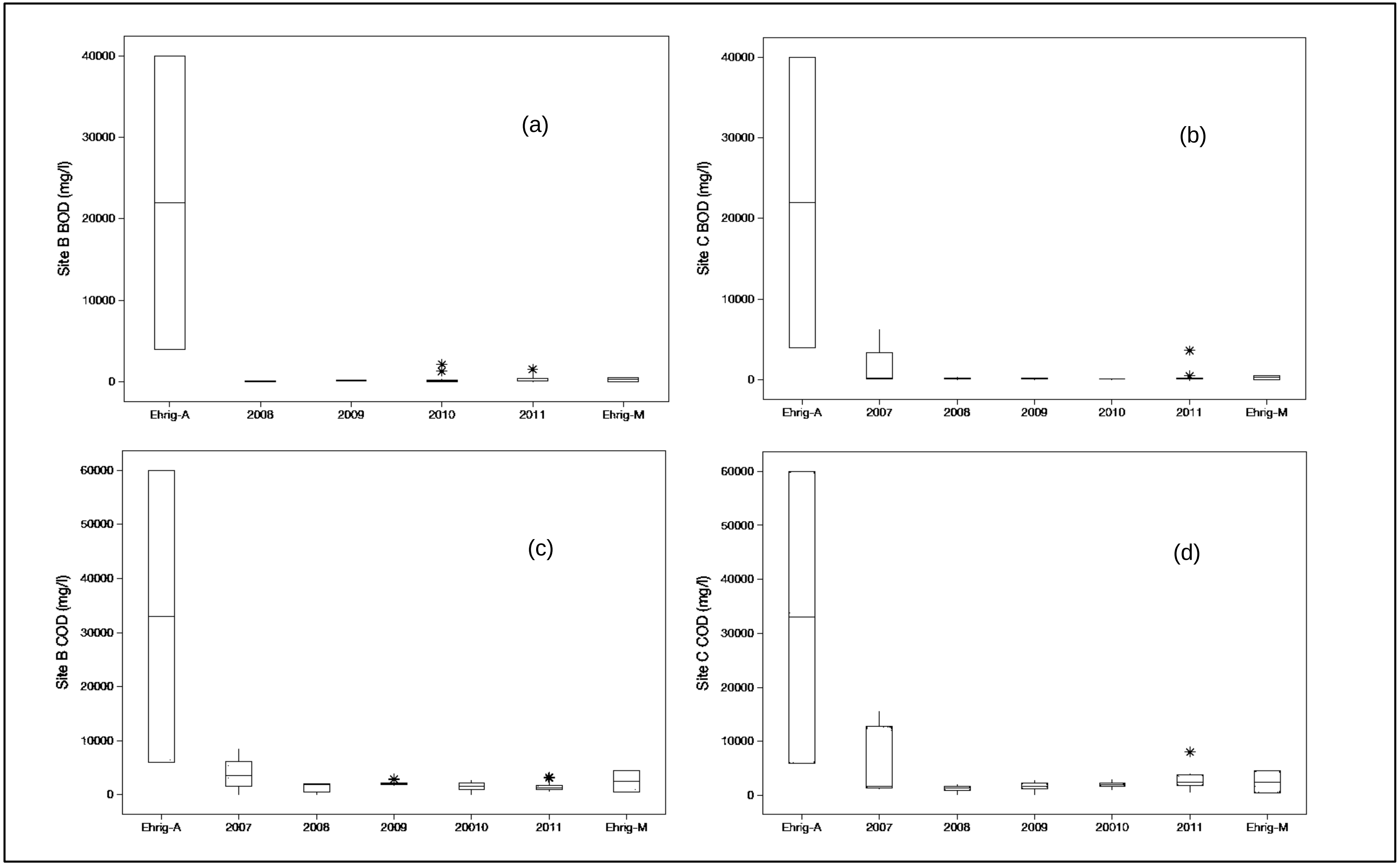


Figure 6.8(a – d): BOD_5 and COD for Sites B & C from 2007 to 2011 compared to Ehrig's data in the acetogenic and methanogenic phases. Test results are highly methanogenic and would be expected to indicate average values of 13,000 mg/l for BOD_5 and 22,000 mg/l for COD during the acetogenic phase. Average methanogenic values are reduced, significantly, equating to 180 mg/l for BOD and 3,000 mg/l for COD.

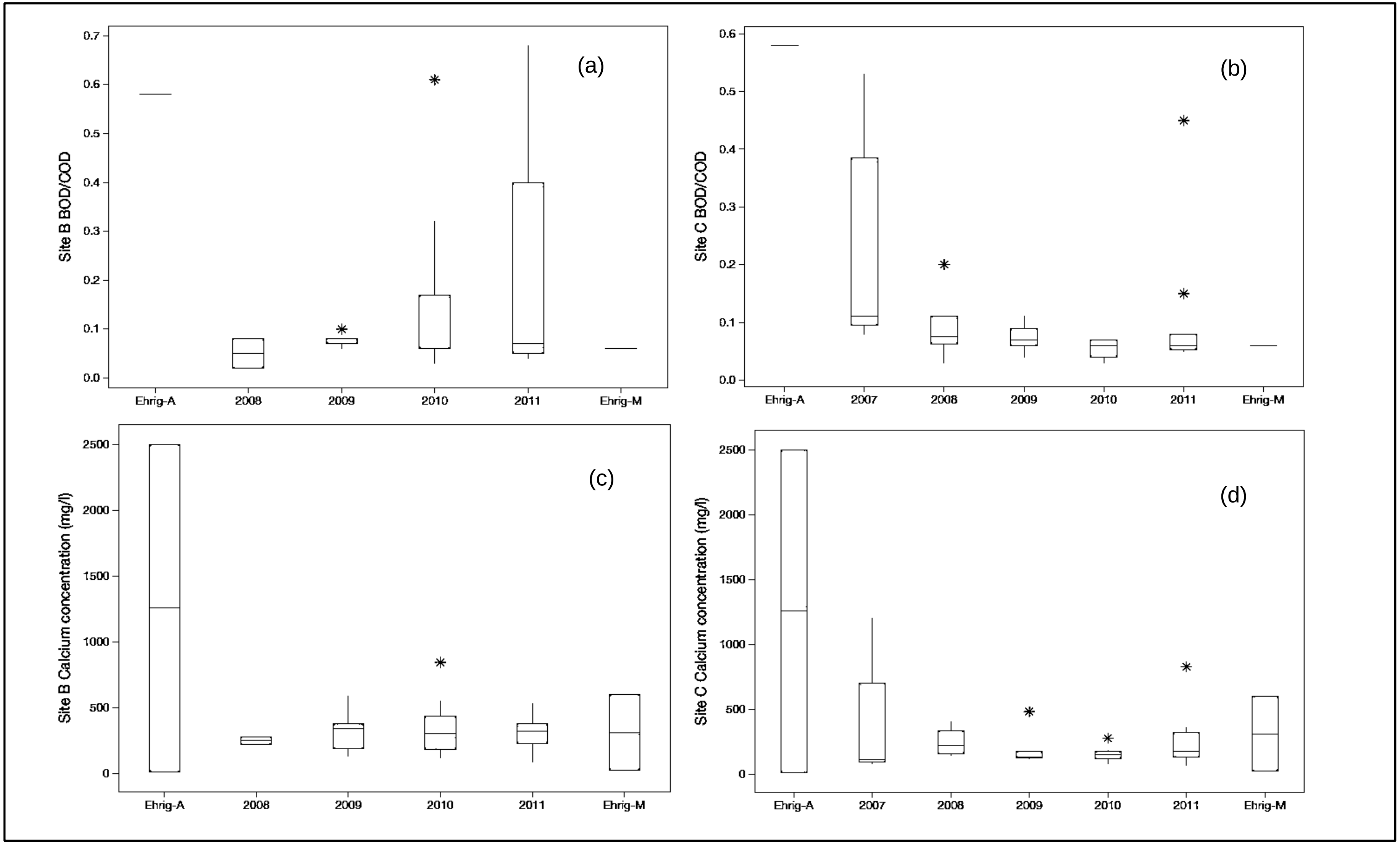
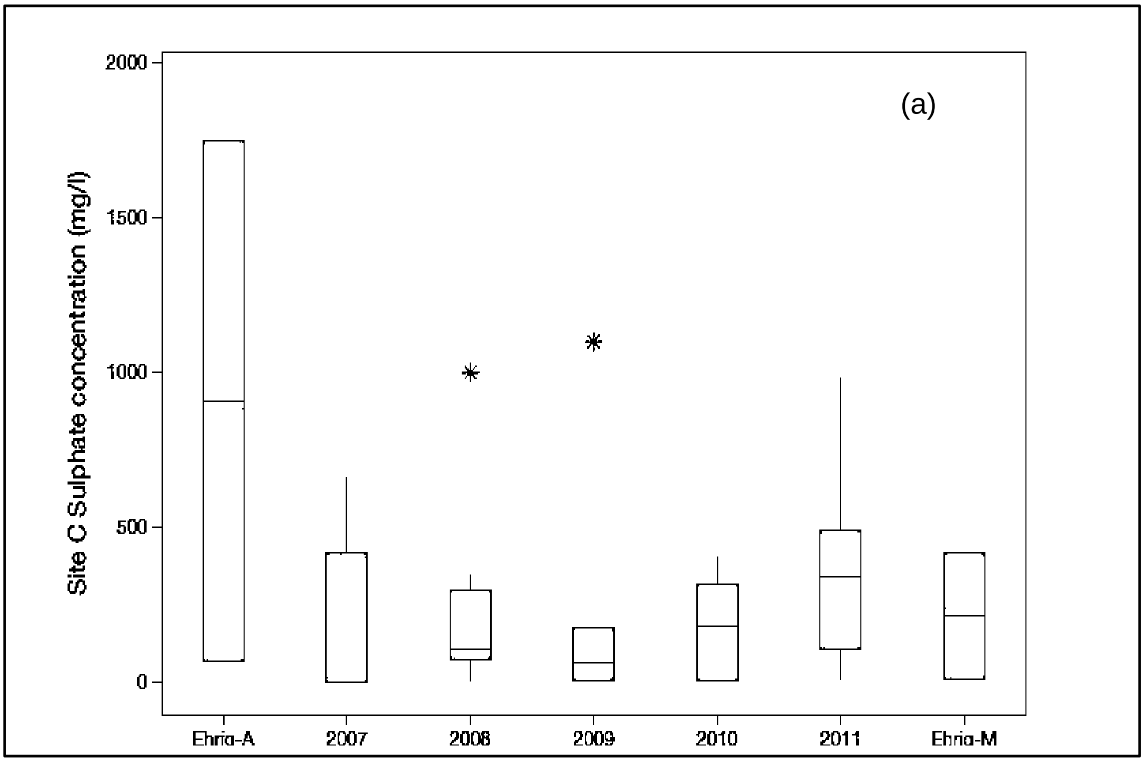


Figure 6.9(a – d): BOD_5/COD ratio and calcium concentration for Sites B & C from 2007 to 2011 compared to Ehrig's data in the acetogenic and methanogenic phases. Average BOD_5/COD for the acetogenic phase equates to 0.58 however, the median value for each box plot is close to the methanogenic value of 0.06. This applies to the sampled values for calcium where average concentrations for the acetogenic phase are 1200 mg/l and for the methanogenic phase 60 mg/l. Median values in all sets of data are representative of the methanogenic phase.

be expected to indicate average values of 13,000 mg/l for BOD_5 and 22,000 mg/l for COD during the acetogenic phase. Average methanogenic values are reduced, significantly, equating to 180 mg/l for BOD and 3,000 mg/l for COD. Whilst BOD_5 fully reflects the methanogenic range for Sites B and C some inconsistencies occur in the data sets for sulphates, COD values for Site C and BOD/COD for Site B. For the most part, COD sample values for Site C similarly reflect the methanogenic range with one exception. This exception applies to one data set only where COD is consistent for both sampled wellpoints (C2 and C3) at 1670 mg/litre, 1420 mg/litre, 1620 mg/litre and 1240 mg/litre however a single value of 11800 mg/litre for wellpoint C3 skews these lower values. BOD_5 for this sample is similarly high and equates to 6250 mg/litre, which has a similar effect. The BOD/COD ratios are conditional upon each parameter having been analysed from the same sample and this has reduced the number of observations. Furthermore, respective leachate samples have both low BOD_5 and COD concentrations and these impact on their ratio having a greater value.

Figure 6.10 (a&b) presents sulphate concentrations for the 2 sites where the acetogenic and methanogenic phases can overlap with the range for the former extending from 70 mg/l to 1750 mg/l with methanogenic concentrations lying in the range 10 – 420 mg/l. Sulphate concentrations at each site remain within



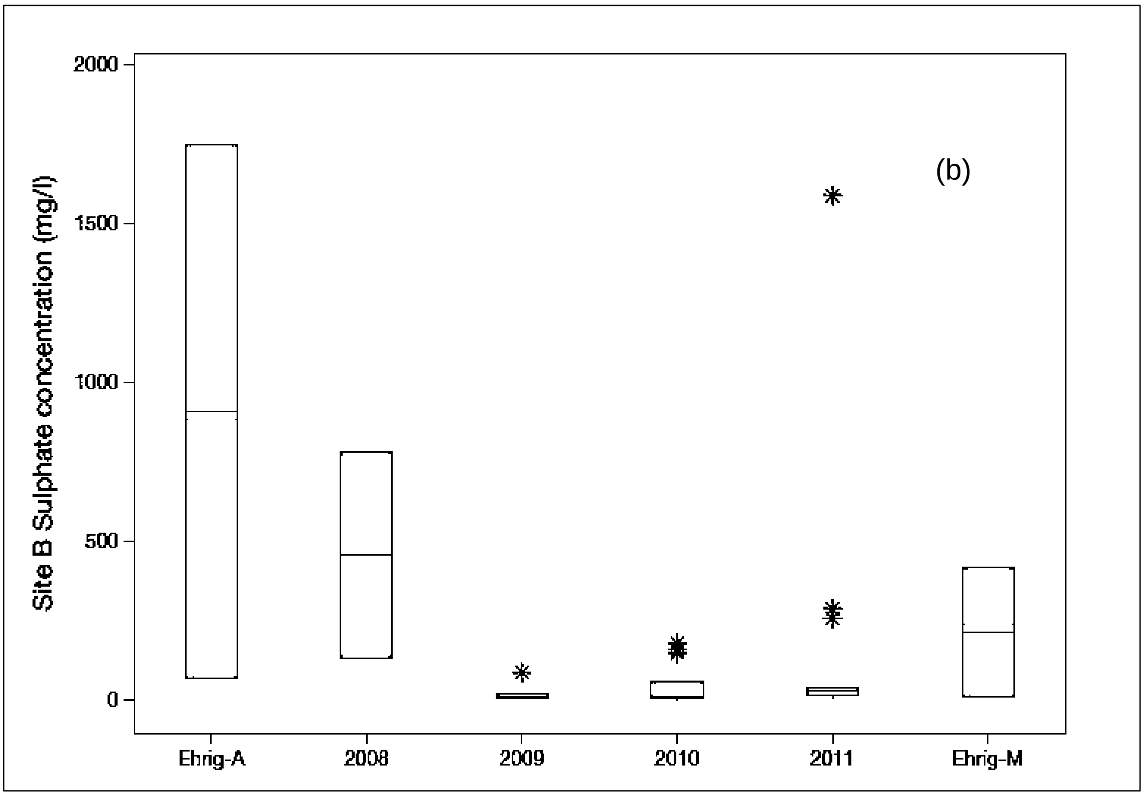


Figure 6.10 (a & b) Sulphate concentrations for Sites B & C from 2007 to 2011 compared to Ehrig's data in the acetogenic and methanogenic phases. Sulphate concentrations tend to overlap with the range for the acetogenic phase extending from 70 mg/l to 1750 mg/l and methanogenic concentrations lying in the range 10 – 420 mg/l. Whilst concentrations are elevated for Site B for the first year of operation (2008) they remain within the methanogenic range overall.

the methanogenic range overall with the exception of Site B, 2008, where the median value is 480 mg/l and the interquartile range extends to 810 mg/l. Cossu et al. (2019) confirm that sulphate concentrations in landfills are low, particularly in the methanogenic phase, which is the situation at both Sites B and C.

Landfill leachate samples from Sites B and C generally display the chemistry typical of mature wastes in the methanogenic phase. In addition, leachate recirculation does not occur at any of the sites examined in this study therefore any likely effect due to recirculation can be discounted.

6.5.2 Comparison with leachate samples from Sites E and F

Figure 6.11 provides a direct comparison of the established and methanogenic Sites E and F for the key parameters pH, BOD_5, and COD for Sites B and C. The prevailing climatic conditions and waste types deposited share identical backgrounds. Whilst sampled leachate pH values for all sites are similar, leachates generated at Sites B and C are from wastes not exceeding 4 years and would be expected to be weakly acidic.

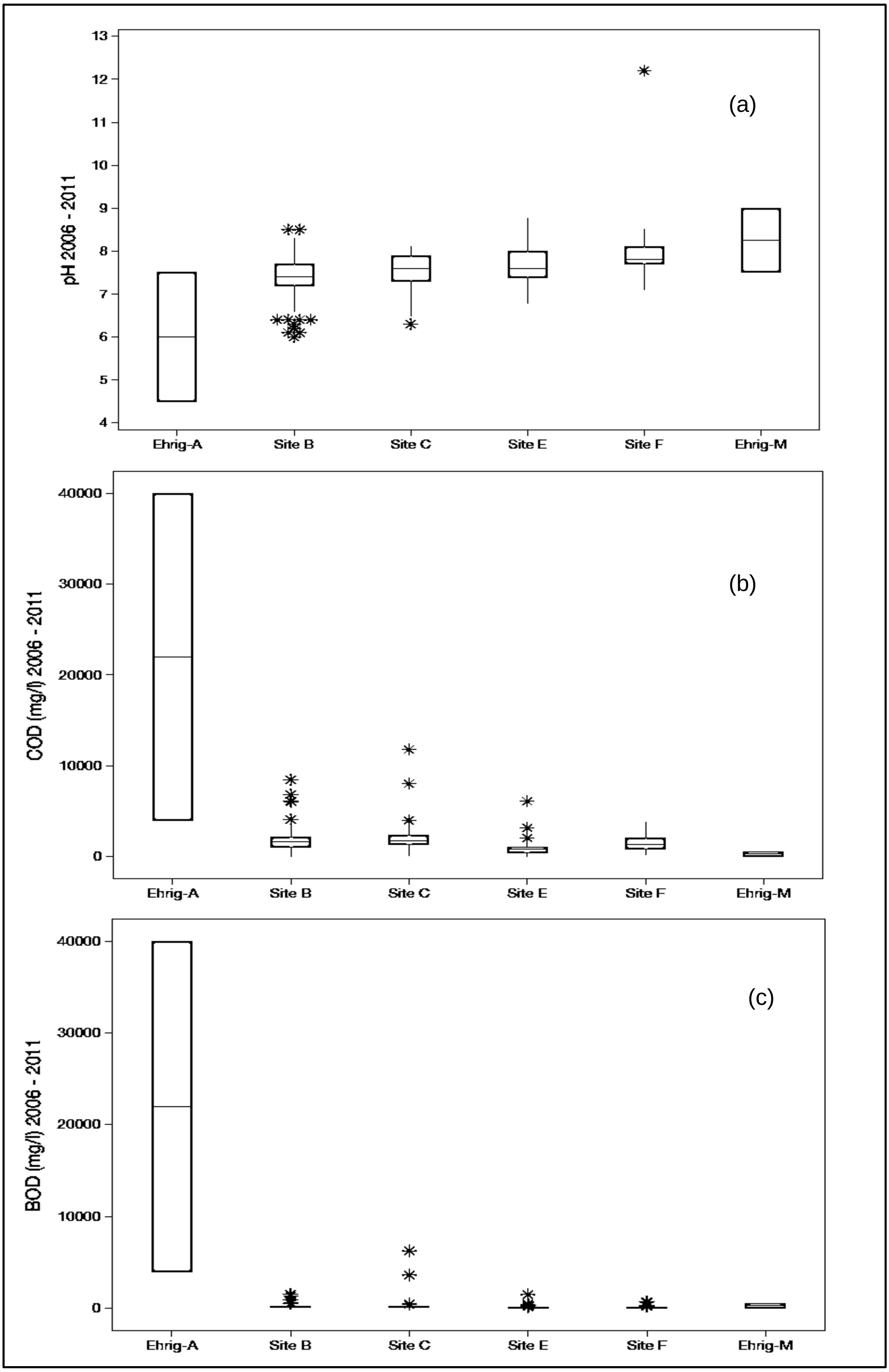


Figure 6.11: pH, BOD_5 and COD for Sites E & F from 2006 to 2011 compared to Ehrig's data in the acetogenic and methanogenic phases and Sites B & C. Sites E and F are established sites containing methanogenic wastes where key indicators compare directly to those of Sites B & C which should demonstrate acetogenic characteristics.

BOD5, and COD sample values are also similar and reflect leachates from mature wastes as opposed to decomposing wastes in the acetogenic phase.

6.5.3 MSW diversion: its impact on MSW imports into Sites B and C

The implementation of the Landfill Directive increased household recycling from 11.2% of total MSW disposal in 2000/01 to 32% in 2006/07 and then to 44% in 2014/15 (Environment Agency, 2016b). Waste imports into Sites B and C were influenced by recycling strategies however recycling capture rates across England differ quite markedly. Records show South West recycling (the location of the case-study landfills) operations exceeded the national average by five percentage points during this period (Environment Agency, 2016b).

The UK objective in decreasing MSW to landfill (Figure 6.3) is consistent in both sites B and C as presented in Figure 6.12. MSW imports into Site B display a reducing trend after 2010. Domestic MSW deposition into Site C is reducing with non-domestic MSW decreasing and then remaining relatively constant. Whilst waste reduction is germane to the Landfill Directive, of critical importance is the redirection of BW into alternative waste streams, away from landfill thus impacting on leachate chemistry by the removal of the acetogenic drivers found in BW wastes (Rees 1980, Christensen & Kjeldsen, 1989).

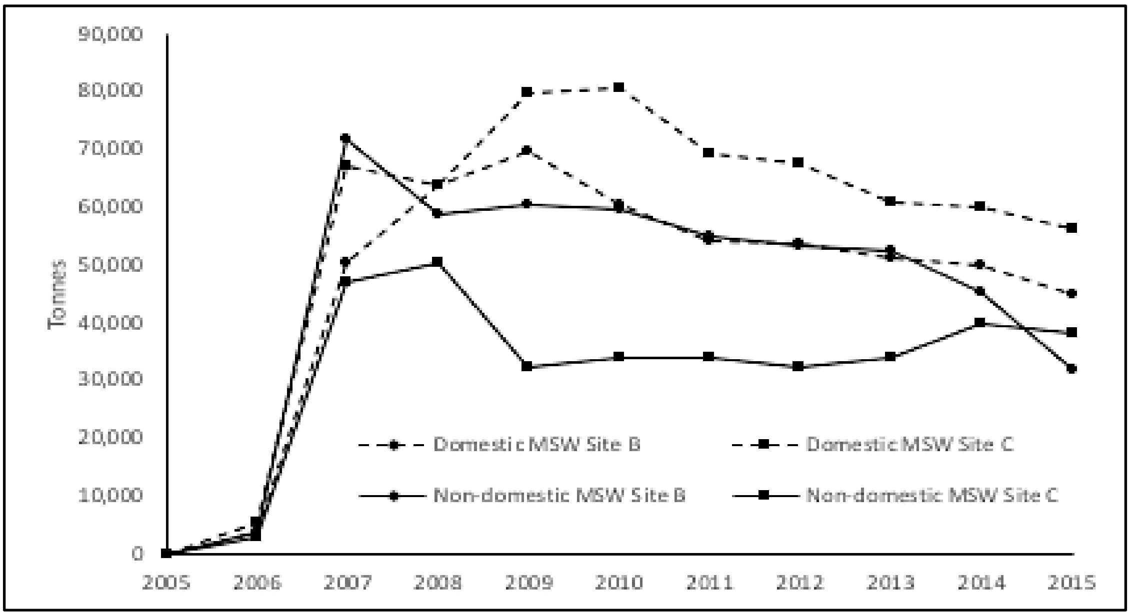


Figure 6.12: Imports of domestic and non-domestic MSW into sites B and C.

Wastes imported into Sites B and C (Table 6.4) identify MSW as forming a significant component of the imported waste. MSW deposits at each site include both rapidly degrading biodegradable waste [BW] and slowly degrading organic wastes ranging from wood to packaging – plastic and otherwise through to mattresses and furniture which are shown to exist unchanged many years after deposition (Rathje & Murphy, 2001). However, the BW fraction is neither separately identified nor recorded.

Although the implementation of alternative waste strategies is having a significant impact on landfilling, it would be incorrect to suggest all rapidly degrading organic wastes have been diverted away from the study landfill sites. At the meso-scale of any landfill, pockets, possibly significant, of BW containing wastes will exist because of less well-developed kerbside collections or inefficiencies in householders or transfer station processing. Furthermore, the EU waste codes (EWC) used to record waste imports are generic descriptions and, as such, often encompass multiple components in respect of both rapidly and slowly degrading components. This is particularly so for MSW (EWC 200301) where it is necessary to identify a number of components which vary with changing preferences in consumer tastes.

6.5.4 Biodegradable waste reduction in the UK

UK specific MSW and BW data for the period before 2010 is unavailable and, where it exists, is unclear or contradictory. This point is clearly evidenced in the 1995 baseline figure for BW content in MSW which exceeds the recorded quantity of MSW generated for that same year. Furthermore, waste composition analyses and respective BW estimates contain variations that have resulted in understated organic content in domestic MSW. Parfitt (2002) concluded the percentage of BW in domestic MSW was as high as 59%. Adjusting for EU defined BW this equates to some 41% (Parfitt, 2002). The inference drawn from these figures for Sites B and C would suggest for some 414,900 tonnes and 403,000 tonnes of BW or approximately 20 and 32 percent respectively of the entire landfill mass for each site for the MSW component alone.

A general indication of the quantities of BW in UK waste streams can be identified from MSW composition studies. As has been discussed, waste composition changes, quite significantly, over time. Figure 6.13 identifies the replacement of ash and clinker with organic materials consistent with the contemporary definition of biodegradable wastes which, at that point, included any organic material. Following World War II organic wastes represented only some 9% of a typical MSW waste stream (Higginson, 1960). By 2006/07 organic materials comprised over 80% of MSW (Resource Futures Appendix 4, 2008).

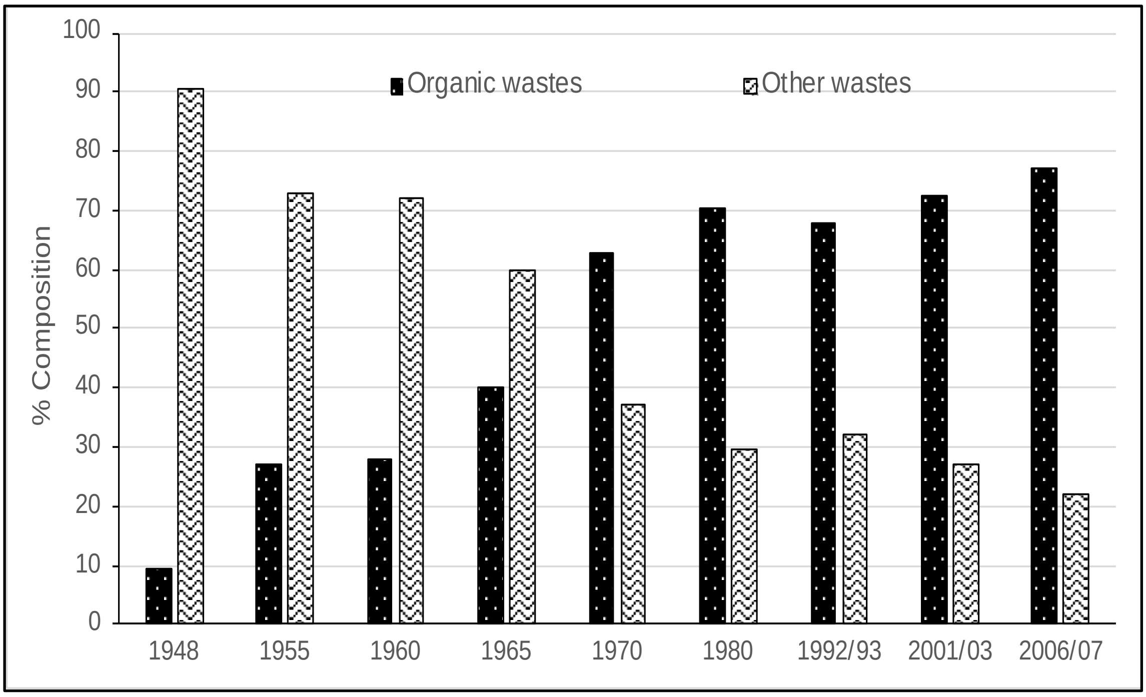


Figure 6.13: Progressive replacement of non-biodegradable wastes with organic materials in English MSW streams.

The European Commission redefined biodegradable wastes to include only garden and park waste, food and kitchen waste from households, restaurants and caterers, retail premises and food processing plants (EC, 2008). These wastes are separated from other organic components in Figure 6.14 where biodegradable waste approximated to 3% of MSW (Higginson, 1960) In 2006/07 biodegradable waste comprised over 40% of MSW making it the largest component (Resource Futures Appendix 4, 2008).

Evidence that BW deposition into landfill has reduced substantially is provided by the Environment Agency's UK Statistics on Waste (Environment Agency, 2016a). The Landfill Directive obligated the UK to determine a 1995 baseline figure for the mass of biodegradable materials in MSW. The figure accepted by the EU in 2010 as the biodegradable municipal waste (BMW) content in MSW was 29,030,000 tonnes (Environment Agency, 2016a) and is the figure subsequent landfilled BW is compared against. By 2010, recorded BW deposits into landfill

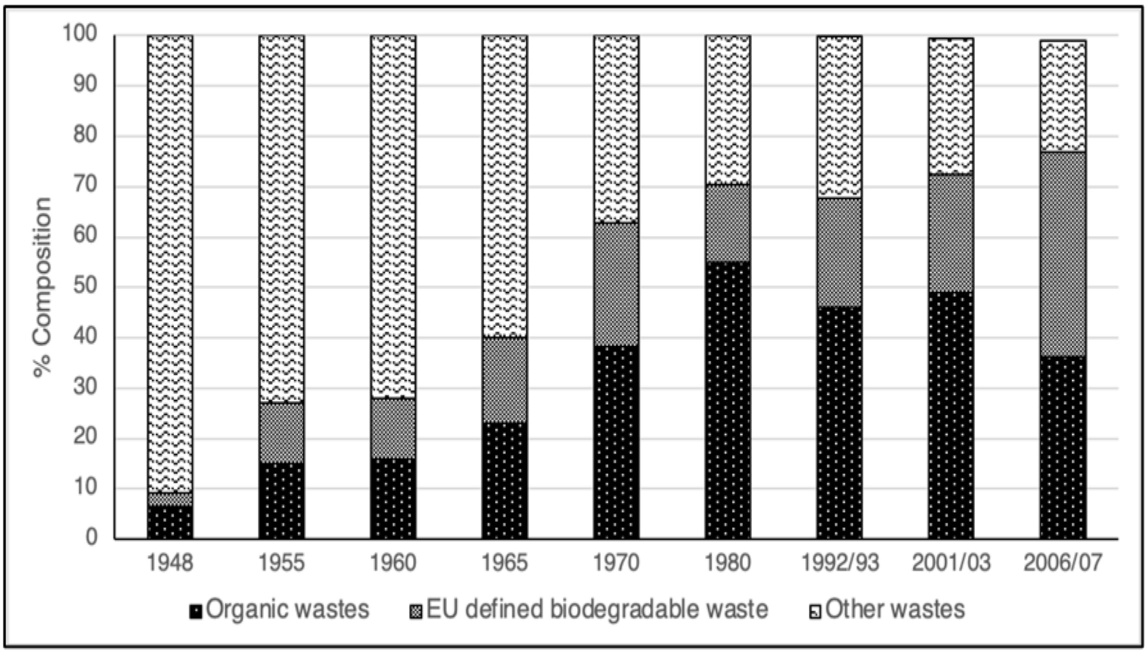


Figure 6.14: Progressive replacement of non-biodegradable wastes with organic materials in the English MSW waste streams.

had reduced significantly to 10,339,000 tonnes (35.6% of the 1995 figure) further reducing to 6,843,000 tonnes in 2014 (23.6% of the 1995 figure). Recorded MSW generated in 1995 was 28,900,000 tonnes and for 2010 was 31,955,000 tonnes (EUROSTAT, 2017b).

It is important to establish quantitatively the decrease in BW going into landfills B and C so that this can be correlated with the changes in evolution of leachate chemistry, to answer the following question: what reduction in BW is required to impact the evolution of landfill chemistry? An analysis of waste imports into the case-study sites is required to estimate the likely BW content of the pre-directive as compared to post-directive landfills. This is achieved by reviewing national data and targets with recorded waste imports at the sites.

6.5.5 MSW imports and estimates of BW by fraction for Sites B and C

To estimate the BW content of wastes imported into the case-study landfills from the recorded tonnages the following methodology is adopted: (i) Identification/separation of the imported tonnages of wastes that contain biodegradable components from their EWC listings (summarised in Figures 6.16 and 6.17), (ii) Component analyses of these wastes to determine those with rapidly degrading fractions (Table 6.5) and (iii) An estimation of the proportion of BMW and BW in each fraction using best available data (Table 6.6).

For (i), recorded imports for respective EWCs was provided by Viridor for each site. Each EWC was then separated into either BMW, BW or non-degradable waste. BMW/BW containing wastes, where those components exceed 1000 tonnes annually, are summarised as Figures 6.15 and 6.16. MSW, domestic and non-domestic are, by far, the most significant waste imports.

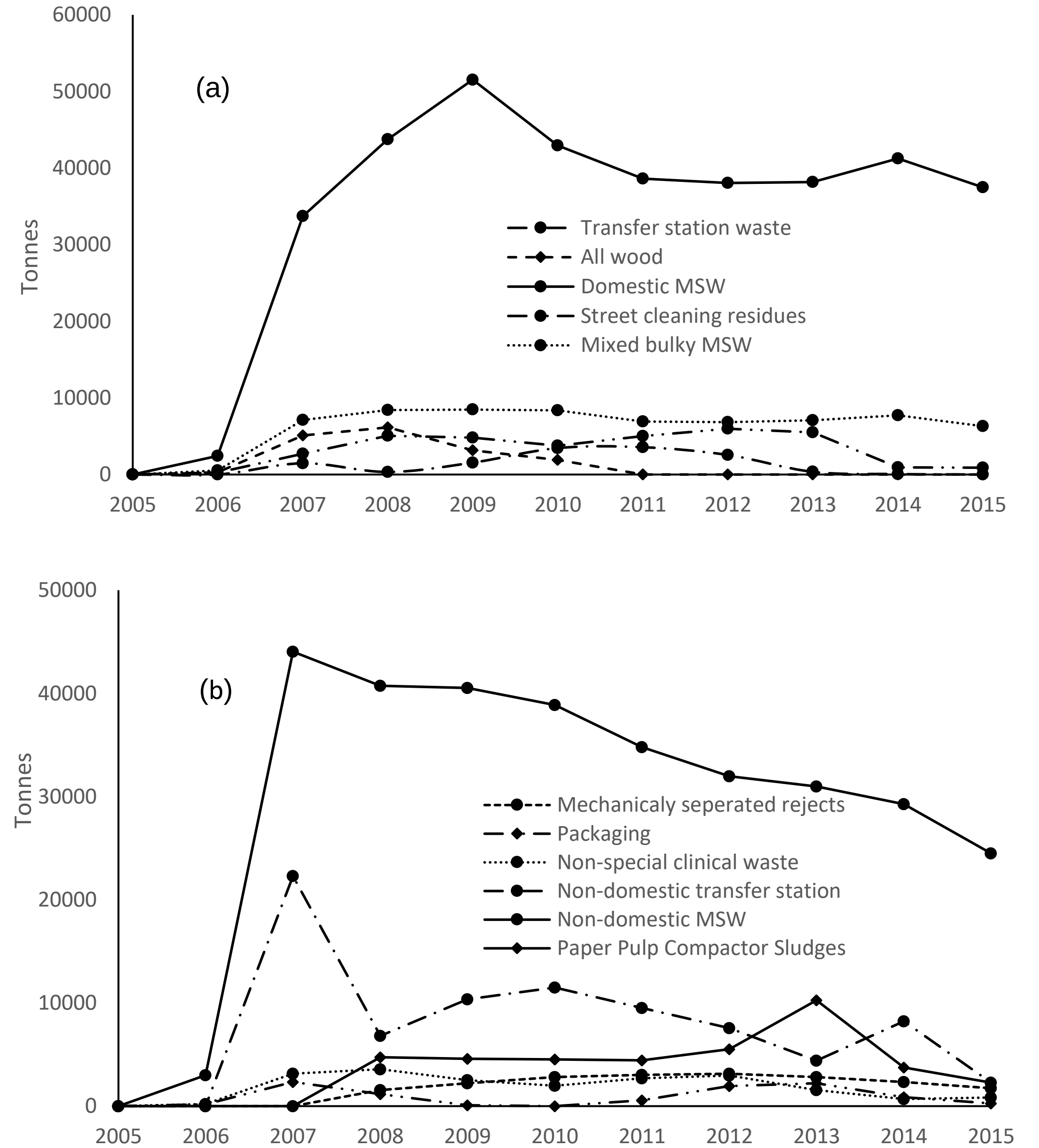


Figure 6.15 a & b: Site B domestic and non-domestic BMW containing wastes 2005 – 2015. Waste import recording and reporting is mandatory whilst sampling was undertaken using European Waste Codes. The same recording methods and classifications were used throughout the period 2005 – 2015.

Whilst the EWCs allow for the separation required for (i) above, there are no established practices for the final two steps and, as a result, different possible data sources are reviewed. For the component analysis required for step (ii), four compositional studies (Table 6.5) from the literature were reviewed as possible estimators from which to establish BW content for MSW imported into Site B and Site C.

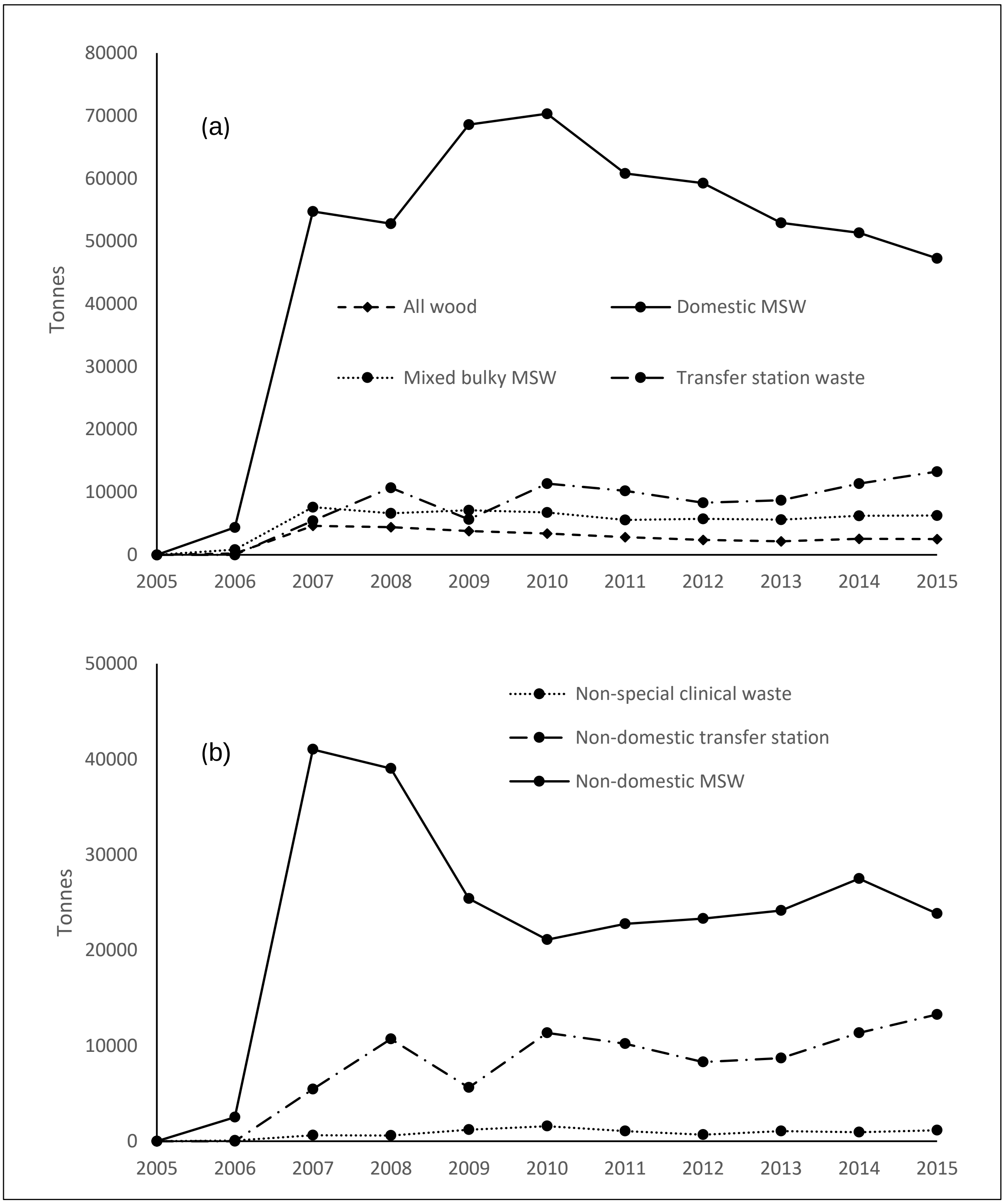


Figure 6.16 a & b: Site C domestic and non-domestic BMW containing wastes 2005 – 2015. Waste import recording and reporting is mandatory whilst sampling was undertaken using European Waste Codes. The same recording methods and classifications were used throughout the period 2005 – 2015.

The NHWAP study was undertaken before the introduction of the Landfill Directive and is considered as incomplete (Parfitt & Flowerdew, 1997) whilst the 2001/03 Burnley study is included for comparison purposes only. The later DEFRA study, undertaken in 2010/11 would not reflect the wastes disposed during the period 2005 – 2009. Therefore, for the purposes of this paper the DEFRA (2006/07) review is considered the most relevant for domestic and non-domestic MSW and includes both regional and BMW analyses (Resource Futures Appendix 4, 2008).

Table 6.5: Compositional assessments for MSW in the UK 1991 – 2010/11

MSW % composition	NHWAP[1] (1991)	Burnley et al.[2] (2001/03)	Defra[3] (2006/07) Domestic	Defra[3] (2006/07) Non-domestic	Defra[4] (2010/11)
Paper and card	34.4	23.6	22.69	39.3	19.2
Kitchen and garden waste	20	35.1	33.65	18.3	33.3
Textiles	2.4	2.4	2.83	2.0	2.9
Plastics	10.9	10.2	9.99	14.4	3.8
Misc. combustible	3.7	4.6	2.37	2.6	-
Disposable nappies	4.2	3.6	2.51	-	-
Fines	6.7	0.6	0.53	1.8	-
Wood	-	4.6	1.66	3.8	-
Furniture and mattresses	-	-	3.73	-	-
Sanitary	-	-	-	1.6	-

Notes: 1) The 1991 data was collected as part of the National Household Waste Analysis Programme (NHWAP). 2. The 2001-2003 data resulted from a study undertaken by Burnley et al. (2006) on behalf of the Welsh Government. 3. The Defra study relied on a range of data most notably Resource Futures. 4. Defra (2015).

Step (iii) entailed the quantification of the BMW/BW content for respective EWCs. This was achieved by, first, determining the BMW content obtained from the DEFRA publication WR0119: A Review of Municipal Waste Component Analyses – APPENDIX 4 to each EWC and second, allocating the BW content as defined by the European Commission in EC2008. BMW and BW content for each waste category is summarized in Table 6.6 which similarly includes the estimated percentages of BW contained within the wastes imported into Sites B and C. Confirmation of the BMW data is provided by Appendix 3 of the Resource Futures report WR1003 (DEFRA, 2012). Furthermore, the biodegradability multipliers applied in Appendix 3 were used to complete gaps in the WR0119 data.

Table 6.6: Estimated biodegradability of major MSW components disposed into Sites B & C. Data sources: BMW - Defra WR0119: A Review of Municipal Waste Component Analyses – APPENDIX 4. BW – EC2008.

Waste category	DEFRA WR0119 BW estimate (%)	EU BW estimate (%)	Sites B & C BW estimate (%)
Food and kitchen waste	100	100	100
Garden Waste	100	100	100
Other inorganic (pet bedding + excrement, unidentifiable putrescibles	100	Nil	50
Paper	100	Nil	Nil
Card	100	Nil	Nil
Glass metals and plastics	Nil	Nil	Nil
Wood	100	Nil	Nil
Textiles	50	Nil	Nil
Sanitary (nappies and clinical)	50	Nil	Nil
Mattresses and furniture	50	Nil	Nil
Miscellaneous combustibles	50	Nil	Nil
Miscellaneous non-combustibles	Nil	Nil	Nil
Soils, builders waste and asbestos	Nil	Nil	Nil
Fines (typically >20mm)	50	Nil	Nil

For a number of EWCs determining the BMW or BW content was straightforward. For example, where an EWC comprised a single waste with identifiable organic content and the imported quantity was low, or waste imports were naturally, rapidly biodegradable for example EWC 190503, off-specification compost, then these were treated as 100% BW. Furthermore, many waste imports at both sites are inert cover materials or classed similarly and have no BMW or BW content. More challenging were those MSW components within Table 6.5 that comprise composite items and contain a mixture of both slowly and rapidly degrading wastes along with wastes that are excluded by EC2008 but nevertheless comprise rapidly degrading components. In defining BW, within EC2008, the European Commission removed a number of BMW components found in MSW, as such, paper and card, wood, textiles, furniture and mattresses have no BW content (European Commission, 2008 and reaffirmed in European Commission, 2016). Other excluded wastes, for example, disposable nappies and components of non-special clinical wastes which include hygiene waste and incontinence pads (Environment Agency. 2015) these, whilst composed of cellulose, some plastics and rubber, absorbent polymers and paper tissue (Rathje & Murphy, 2001, DEFRA, 2008) and considered to biodegrade extremely slowly, differ from their soiled content which comprise rapidly degrading organic constituents that need to be included in the BW estimate. Within the UK, the composition of disposable nappies is very similar (Department for Environment, Food and Rural Affairs, 2008) and it is assumed this similarity applies to some clinical wastes. The Waste and Resources Action Programme (WRAP) surveyed the excreta content in disposable nappies and estimated the weight of the soiled contents to be 727kg based upon a two-and-a-half-year use by a single infant (Department for Environment, Food and Rural Affairs, 2008). For this reason, here some fifty percent of the mass of sanitary waste is included as BW as it is considered better to overestimate than underestimate. Miscellaneous combustibles contain carpets and underlay, rubber and unclassified combustibles. For these latter two items and due mainly to classification differences these remain unchanged that is 50% BMW but no BW content (Department for Environment, Food and Rural Affairs, WR0705, 2008).

As a result of the diversion of kitchen and garden, paper and card waste and the wide-ranging definition of MSW, for both domestic and non-domestic MSW, the component percentages are required to be renormalized to obtain the increased quantities of remaining MSW components still being landfilled. The given percentages in Table 6.6 are stated before any diversion occurs. As a result of diversion, other MSW components will increase as a result of the diversion strategy.

6.5.6 Estimated BW materials at Sites B and C

The total quantity of waste imported into Sites B and C together with respective MSW imports are presented in Table 6.7. MSW forms the major source of biodegradable materials imported into each site, for Site B a further 33% of imports comprise inert cover materials. In addition, Table 6.7 presents a series of estimated scenarios for the both the total BMW and likely BW content for different diversion capture rates for each site. The BMW estimate is included for comparison to Parfitt's 2002 published estimate (section 4.4), although this is included as a percentage of total waste imported. For the three BW estimates, the first is a reconstruction of the no-recycle condition expected to exist either during the pre-Landfill Directive period or, had similar waste disposal practices continued, where the bulk of domestic and non-domestic MSW was discharged directly to landfill. The second estimates the situation occurring post 2005 where alternative waste strategies divert recyclable components in MSW into more favoured options – here the 50% estimate being representative of the minimum recycle rate for those components in the South West Region. The third is the estimated BW content where the maximum Department for Environment, Food and Rural Affairs diversion rate for kitchen and garden waste applies. The latter two scenarios provide a feasible range of diversion from which the reduction in BW can be estimated. However, for scenarios two and three, it should be noted that the renormalisation of remaining wastes increases their mass when compared to those in the zero-diversion scenario. This accounts for the fifty and seventy-five percent diversion rates not equating to that fraction of the BW content in the zero-diversion scenario. The waste imports used to estimate the BMW and BW content were restricted to those years encompassed by the 2006/7

DEFRA composition analysis. Furthermore, it is these wastes that will determine the resultant leachate chemistry.

Table 6.7: Estimated BMW and BW content as weights and percentage of total waste deposits and MSW for landfill Sites B and C.

		Estimated total imported BMW & BW as a proportion of total waste			
	Total imports 2006-9 in tonnes	BMW content in tonnes (%)	BW - zero diversion tonnes (%)	BW - 50% diversion tonnes (%)	BW - 75% diversion tonnes (%)
Site B	555,158	268,909 (48.4%)	115,809 (20.9%)	78,136 (14.1%)	57,455 (10.3%)
Site C	430,050	239,601 (55.7%)	103,298 (24%)	63,359 (14.7%)	43,388 (10.1%)
		Estimated MSW BMW & BW as a proportion of total MSW			
	MSW content in tonnes	BMW content in tonnes (%)	BW - zero diversion tonnes (%)	BW - 50% diversion tonnes (%)	BW - 75% diversion tonnes (%)
Site B	330,423	213,614 (64.6%)	100,942 (30.5%)	63,269 (19.1%)	42,587 (12.9%)
Site C	332,949	214,973 (64.6%)	101,937 (30.6%)	61,998 (18.6%)	42,027 (12.6%)

The literature identifies the completion of compositional analyses as providing a number of “challenges” (Resource Futures Appendix 4, 2008). For this analysis a percentage of MSW remains unclassified. From other MSW compositional studies these unclassified materials comprise glass, metals, WEEE, some hazardous wastes including batteries together with bricks, plaster soils and other building materials and are not specifically identified in the 2006/7 analysis. These wastes have no BMW/BW content (Resource Futures Appendix 4, 2008; Department for Environment, Food and Rural Affairs, 2012). The fine fraction is accounted for. Despite these challenges, the data provided is the best available for the period in question.

Given that MSW forms the major source of biodegradable waste at each site, the estimated range for landfilled BW, resulting from the 50% and 75% diversion scenarios, relates to a reduction when compared to the Parfitt’s (adjusted) estimate of 41% identified in section 4.4. Allowing for variations in methodology between Parfitt's estimate and this approach this reduction is significant and is reflected in the chemistry of the leachates arising.

6.6 Conclusion

Directive 1999/31/EC required member states to divert biodegradable wastes contained in MSW away from landfills. By invoking this policy change it was anticipated that an evolution in decomposition would: i) manifest itself in leachate chemistry and ii) speed the development of methanogenesis by improving the physiochemical environment in which methanogenic bacteria developed. Two study sites (labelled B and C) have offered a rare opportunity to: (1) review and compare leachate samples from the post-Landfill Directive period to established research, undertaken before the Landfill Directives publication and (2) Estimate scenarios for the reduction in biodegradable materials deposited into landfills since the implementation of the Landfill Directive.

Ehrig (1988) published a series of key indicators for leachates generated during 2 phases of MSW decomposition: i) acetogenesis (early-stage decomposition) where immature wastes generate weakly acidic leachates and ii) methanogenesis (decomposition extending from 5 to 15 years) where leachate pH is basic. Identification of a particular phase of landfill degradation relies, primarily, upon comparisons made between leachate samples with concentrations lying within specific ranges which are then determined as being indicative of either the acetogenic or methanogenic phases (Ehrig, 1988; Kjeldsen et al., 2002). For the initial period where pH values are weakly acidic and expected to lie in the range 4.5 – 7.0, the lowest pH values were 6.1, 6.4 and 6.6 for March, April and May 2007, respectively a period . By September 2007 and, following an initial period of reduced (slightly acidic) pH during 2007 all sampled values lie above 7 and generally reflect values expected during the methanogenic phase. Furthermore, BOD_5, COD, BOD_5/COD and dissolved calcium and sulphates lie with the range associated with the methanogenic phase.

Results from leachate sampling corroborate the sites have circumvented the "classical" acetogenic phase. The leachate samples, for the most part, when compared to Ehrig's data are representative of a site in the methanogenic phase and closely resemble Ehrig's data values for this phase. The consistency or lack of variation in respective samples further reinforces this assessment.

The redirection of BW waste from landfills removes materials able to decompose rapidly. The lack of credible UK records for waste deposits, particularly in respect of the central biodegradable content within MSW waste streams, has led to the estimation of its likely content. Four scenarios are presented to explain the observed change in leachate chemistry. Whilst the estimate for total biodegradable materials (BMW) represents some 49 – 56% of the materials deposited into the two study sites, the pre-Landfill Directive BW content represents 21 – 24% of deposits. Redirection to the end of 2009 has reduced this to approximately 10% with the concomitant change in leachates produced. It is noticeable that recycling strategies have continued to develop across the UK since this period.

Chapter 7: Conclusions, contribution to the knowledge base and proposals for further work

7.1 Conclusions

7.1.1 General conclusion

Landfilling as an adopted strategy for the disposal of municipal solid wastes since 1945 has created a legacy where decomposing MSW creates a series of biogeochemical environments which release harmful gaseous and liquid emissions. It is proposed that an achievable solution is the removal of these wastes from landfills. However, this study has demonstrated that considerable quantities of soils and fines will require to be excavated to achieve this outcome.

Before such an engineering project can be actioned, knowledge of an individual landfill's content is necessary. Engineering projects usually employ intrusive investigation with trial pits or boreholes. However, this is expensive and due to the heterogeneity of MSW, will carry a high degree of uncertainty. These issues led to the framing of research questions 1 and 2 specifically: i) can historical and geographic data provide a means of predicting landfill mass content? and ii) do significant correlations between sampling data and MSW content exist? It is well understood that recording of waste data in the UK was limited until the introduction of WasteDataFlow (2004). Assuming a positive outcome from these research questions, Objective 4 of this study proposed to develop a model for the identification of materials landfilled during the period 1945 – 2007 so as to benefit decision taking.

Chapter 3 considered data published by English waste collection and disposal authorities and proposed a near linear series of daily per capita MSW generation rates for England based on mean values which extend from 0.80 kg for the 1950s to 1.55 kg for 2005/06. In addition, the data would also allow a regional, district or more localised evaluation to be developed and utilised. Furthermore, the chapter examined waste disposal data to confirm that 85 – 91% of waste was disposed to MSW landfills from the early 1950s through to the mid/late 1990s. Whilst landfilling was generally accepted to form the disposal option for many local authorities, it does not apply to all and for the accurate application of the proposed model at the district or individual authority level the study established:

(i) how the remaining percentage of waste was disposed and by which local authorities (ii) how different local authorities waste strategies evolved over time and iii) how different data sets confirmed or contradicted others.

Chapter 4 identified the difficulties with forecasting MSW composition where variations can be attributed to: i) consumer purchasing preferences, ii) temporal variations and iii) socio-economic background. In the absence of regulation these variations in waste composition determine the materials flowing into a landfill. As such, these variations and the developing regulatory framework have a significant impact on the materials flowing into landfills are considered within the framework of the proposed epochs. For the first 2 epochs waste composition is the primary determinant with regulation becoming the ultimate determinant by the 4th epoch. Filling gaps with surrogate data is problematic and attention is required in respect of the socio-economic factors prevailing and the changing nature of individual samples. In defining epochs, data substitution offers an opportunity to fill gaps via interpolation as opposed to extrapolation.

In respect of how this study answered research questions 1 and 2, Chapter 3 framed a methodology to answer the second question by determining there was significant and representative data available in respect of sample size for those English waste collection and disposal authorities weighing 50% or more of the MSW generated. In respect of the first question, Chapters 3 and 4 present a comprehensive package of historical data that allows a positive response to the question and furthermore, provides sufficient data to meet the requirements of research objectives 3, 4 and 7.

MSW waste composition studies classify material content into 11 to 13 quite general categories by grouping wastes composed of similar materials. This study proposes the value materials buried in landfills namely, combustible materials, glass and more importantly metals were relatively consistent over each epoch. Data also show the limited post-war impact of material salvage on flows into landfills before its near demise during the mid-late 1960s. Material flows remained undisturbed until the implementation of recycling which had little impact until 2002/03. From this point more detailed recycling data began to be collected and published. Chapter 4 also consider data in respect of waste flows from industrial generators. Evidence given to the Working Party on Refuse Disposal

(Department for the Environment, 1971) suggested that 90% of landfills accepting these wastes were under ownership of the generators. Furthermore, EURELCO identified that 80% of European landfills are MSW landfills.

Chapter 5 proposes a system dynamics model to estimate the material content in historical MSW landfills. The model was structured with 3 sub-units and 2 tiers which generated MSW generation data, disposal data and landfill content data over the study period. The model has the capability to estimate material content in individual landfills, landfills servicing a specific conurbation or region along with generating output at the national level. Output from the modelling produced a range of material quantities determined by the percentage fraction of waste categories entering landfills. These fractions were termed low, average and high and reflected the percentage range of a particular material likely to be stored within a landfill.

Methods for verification and validation of the model are proposed and implemented. To verify the Tier 1 model's output a mass balance exercise was undertaken using the same LA collected data set as inputs. The figures generated by the mass balance were identical to the model output although the approach was time consuming and required many spreadsheets. Validation of the Tier 2 output focused on comparisons to landfill mining and specific site sampling data where the model was able generate comparable results against those published. The Chapter allowed the successful completion of research objectives 4 and 7 where a low/high estimate for the value materials existing in English landfills is presented.

In respect of the 3rd research question and research objectives 5 and 6, Chapter 6 presents 2 study sites that offered a rare opportunity to: (i) review and compare leachate samples from the post-Landfill Directive period to established research, undertaken before the Landfill Directives publication and (ii) Estimate scenarios for the reduction in biodegradable materials deposited into landfills since the implementation of the Landfill Directive. The redirection of biodegradable waste from landfills removes materials able to decompose rapidly. Due to the paucity of credible UK waste records which includes the biodegradable content within MSW streams, a method of estimation is proposed. Whilst the estimate for total biodegradable materials represents some 49 – 56% of the materials deposited into the two study sites, the pre-Landfill Directive biodegradable content represents 21 – 24% of deposits. Redirection to the end of 2009 has reduced this to approximately 10% with the concomitant change in leachates produced. Results from leachate sampling corroborate the sites have circumvented the "classical" acetogenic phase. The leachate samples, for the most part, when compared to Ehrig's data are representative of a site in the methanogenic phase and closely resemble Ehrig's data values for this phase.

7.1.2 Historical data – dealing with uncertainty

This study collected and in Chapters 3 and 4 examined and reviewed: i) historical data that would lead to waste generation, ii) historical data identifying how and where MSW was disposed iii) historical municipal waste composition data in order to estimate the material content in landfills. Research presented in Section 2.4 of Chapter 2 identifies associated risks attached to the significant uncertainty surrounding a landfill's content. This study presents an approach to reduce this uncertainty which, in one respect, relies on a basic foundation and that is to identify and quantify those materials flowing into landfills.

The approach relies upon data collected and reported by English local authorities in respect of MSW generation and disposal with MSW composition sampling data collected on either an authority's behalf, by acknowledged waste professionals, or taken from peer reviewed academic studies. However, whilst the data reflect a series of snapshots of contemporary waste generation and composition some uncertainty with the resulting statistical parameters will occur. Webster and Lark (2013) accept a level of precision is required and that should be determined by the nature of the problem and the approach taken to sampling however, for this study sampling methods were determined by others.

Chapter 3 has shown sample size to be sufficient and the results arising from sampling were reviewed. Resulting from that review, only local authority data reported by those authorities weighing 50% or more of generated MSW were selected for further analysis or modelling purposes. By excluding data with fewer or only test weighings sample means and medians were very similar and for sampled data, reflect mainly normal distributions. However, simply using measures of central tendency is insufficient (Caers, 2011). The interquartile ranges and standard deviations of respective data sets were small reflecting the similarity in respect of MSW generation data and its composition. What this means is that waste generation for a large sample of the population is similar and where MSW generation statistics were missing (and these are numerous), these gaps could be reliably filled with estimated statistics that reflected contemporary trends. In respect of MSW composition, differences in composition exist however, these are accepted and taken into the modelling framework as a range from which a series of material expectations arise. Modelled material content, when compared with arisings from excavated landfills compare favourably and give credibility to the adopted methodology.

7.2 Contribution to the knowledge base

7.2.1 Overview of contributions

This study's contributions to the knowledge base include inventories of MSW data and landfill content together with an inventory of waste repositories in England. A system dynamics model to estimate waste flows and the material content in landfills. In addition, this study provides confirmation the diversion of biodegradable MSW away from landfills impacts on the chemical environment within landfills.

7.2.2 Inventories

This study has produced:

- An inventory of MSW disposal sites for England.
- A compendium of historical and contemporary waste data.
- 4 material inventories for materials contained in landfills based upon this study's defined epochs (Chapter 5, Section 5.8).

7.2.3 Landfill content modelling

For most historical landfills, any record of MSW imports is simply not available. An approach to ascertaining landfill content is by modelling. The literature search reported in Chapter 2 identified 2 publications that proposed different approaches to modelling the content of landfills: (i) a mathematical method (Lyons et al., 2010) and (ii) a data model based on historical site records and decomposition modelling (Wolfsberger et al., 2015a). Many other models have been produced but they focus upon waste management and not landfill content.

This study's contribution include:

- Identification of many positive MSW data elements where previously, MSW data was viewed against a background of negative anecdotal feedback (Chapters 3 and 4).
- From that data, has demonstrated them to be effective in the generation of annual and cumulative MSW quantities for disposal to landfill and incineration (Chapters 3 and 4).
- Applied those data as model inputs where they are effective in predicting landfill mass content (Chapter 5).

This study has developed, verified and validated:

- A model that captures the dynamics associated with MSW and regulatory changes since the termination of World War II (Chapter 5).
- A modelling approach applicable to a national, regional or local setting for the estimation of material content in landfills (Chapters 3, 4 and 5).

The study has established and produced:

- 4 epochs of MSW generation and disposal.
- A methodology for estimating missing data (Chapters 4 and 5).
- Estimates for the quantity of MSW generated in England since the termination of World War II where they did not previously exist (Chapter 5, Section 5.4.5).
- Estimates for the quantity of MSW disposed into landfill and incinerated in England since the termination of World War II where they did not previously exist (Chapter 5, Section 5.5.5).
- Low and high estimates for the quantities of value materials together with soil cover that exist in English landfills (Chapter 5, Sections 5.6.6 and 5.7.3).

7.2.4 Landfill chemistry evolution

European Council Directive 1999/31/EC required member states to divert rapidly degrading organic materials contained in MSW from entering landfills. From a theoretical perspective this would allow a more rapid development of methanogenic bacteria by preventing release of organic acids resulting from the decay of putrescible (food and garden) wastes. Testing to assess the effectiveness of the proposal is difficult as liquid emissions that flow through older wastes assume the characteristics of that waste. This study was able to use 2 new landfills, where there would be no impact from older wastes, to demonstrate the Directive was occurring. These findings were published in Warwick et al., (2018). Altered chemical evolution in landfill leachate post implementation of biodegradable waste diversion. *Waste Management & Research,* 36 (9) pp. 857 - 868.

7.3 Proposals for further work and validation

7.3.1 Missing data

This study identified characteristics occurring within 4 specific epochs as being similar and using this to fill gaps in data sets however this was a somewhat 'broad brush' approach. Chapter 3 proposed searching for similarities within population sub-groups or intra-relational connections. These fall into two areas: i) rates of MSW generation and ii) material components in MSW. For the first, Hoornweg and Bhada-Tata (2012) identify the differences in generation rates between rural and urban areas. However, The Ministry of Housing (1967) reported that for the UK rural areas generated, on average, amounts equal to or more than urban areas whilst for urban areas there were similar generation rates. This study identified similarities in the rate of MSW generation by certain densities of population, but this requires further work to establish definite correlation(s) between population density and rates of MSW generation. For the second area, precision at the local level could be increased by the incorporation of socio-economic differences associated with MSW generation in the modelling. This study has reported these differences in composition (Chapter 4, Section 4.3 and Chapter 5, Section 5.7.4) and qualifies the approach taken here by the adoption of a range of material content.

7.3.2 Expanding the material components in MSW

MSW composition studies traditionally were limited to 10 – 12 categories. From the mid-1980s (Rufford, 1984) these have been increased. Furthermore, waste sampling practice has sought to separate chemically different fractions existing within each category as these are treated differently when recycled. For the historical MSW sampling further separation could be undertaken although this would require research into the archives of economic history and the work of social historians, which are available and have been accessed, in part, to aid the development of this study. Further work would be useful for metals and plastics in terms of their realizable values and household chemicals in respect of their future contamination potential.

7.3.3 Applying this model to individual landfills

Objective 2 of this study produced an inventory of potential MSW resource repositories based upon the Environment Agency's *Not in my back yard* web pages. Whilst the 19,763 historical sites included could be located by their grid reference, rapid sorting was impossible as fewer than 6,000 sites contained the necessary site address with many identified by the operator's head office address. The model proposed by this study along with different scenario matrices (with the exception of Manchester, these are yet to be compiled from the data available) can be used to determine the resource content within individual landfill sites. This study has used the model to identify limits for the resource content for all MSW landfill sites in England.

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Appendix 2.1 - Box plot descriptive statistics

Variable	N	Mean	Mean Standard Error	Standard Deviation	Variance	Median	Skewness	Kurtosis
Fines	54	50.67	2.34	17.17	294.90	51.70	0.15	-0.19
Organic	54	0.43	0.26	1.93	3.74	0.00	4.97	26.24
Wood	53	8.33	0.72	5.24	27.41	7.60	0.57	-0.36
PPC	47	6.14	0.75	5.17	26.77	5.00	2.16	6.84
G & C	29	3.61	0.76	4.09	16.75	2.00	2.35	7.32
Metal	49	4.08	0.55	3.87	15.01	3.30	2.25	6.32
Plastic	51	19.76	1.97	14.06	197.66	18.30	1.13	1.83
Textile	38	4.73	0.49	3.01	9.08	4.09	0.94	0.23
Other	48	6.32	0.92	6.35	40.36	3.00	0.97	-0.16

	N	Minimum	Maximum	Q1	Q3	IQR	Mode	N for Mode
Fines	54	14	89.70	35.70	62.40	26.70	35.7, 45.0	2
Organic	54	0	12.00	0.00	0.00	0.00	0.00	51
Wood	53	0	20.10	4.33	10.85	6.53	10.00	3
PPC	47	0	28.70	2.60	7.00	4.40	6.00	3
G & C	29	0.1	19.50	0.75	5.29	4.54	0.50	4
Metal	49	0	20.00	1.32	5.10	3.79	1.10	3
Plastic	51	0	70.00	7.90	25.00	17.10	34.30	2
Textile	38	0	11.70	2.86	6.18	3.32	3.00	3
Other	48	0	23.59	1.33	11.20	9.87	0.50	4

Appendix 3.1 – Primary data sources

Source	Document	Type	Dates	
Ministry of Housing and Local Government	Public Cleansing. Refuse collection and disposal; Street cleansing costing returns	pdf	1935/36, 1952/53, 1953/54, 1954/55, 1955/56, 1957/58, 1959/60, 1960/61, 1961/62, 1962/63, 1963/64, 1964/65, 1966/67	
Departmant of the Environment, Society of County Treasurers and County Surveyor's Society	Analysis of Local Authority Waste Disposal	pdf	1974/75	
Departmant of the Environment, Society of County Treasurers and County Surveyor's Society	English Local Authority Waste Disposal Statistics 1974/75 to 1977/78	pdf	1974/75, 1975/76, 1976/77, 1977/78	
Society of County Treasurers and County Surveyor's Society	Waste Disposal Statistics	pdf	1975/76	
Chartered Institute of Public Finance and Accountancy	Waste Collection and Disposal Statistics	pdf	1976/77, 1977/78, 1978/79, 1979/80, 1980/81, 1981/82, 1982/83, 1983/84, 1984/85 1985/86, 1986/87, 1987/88, 1991/92, 1992/93, 1994/95	https://www.cipfastats.net/environmental/wastemanagement/default.asp?view=commentary&year=1992-93&content_ref=12969
Chartered Institute of Public Finance and Accountancy	Waste Collection and Disposal Statistics	Excel	1995/96 - 2018/19	https://www.cipfastats.net/environmental/wastemanagement/default.asp?view=commentary&year=1992-93&content_ref=12969
Audit Commission	Local Authority Performance Indicators	pdf	1993/94, 1994/95, 1995/96, 1996/97, 1998/99, 2000/01	https://webarchive.nationalarchives.gov.uk/20150410163041/http://archive.audit-commission.gov.uk/auditcommission/aboutus/publications/pages/corporate-papers-archive.aspx.html

Appendix 3.2 – Waste composition sources

Publication year	Sample Date & (Number of analyses)	Author(s)	Publication and Source Detail
1929	1925/26 (1)	J.C. Dawes	Report of an Investigation into the Public Cleansing Service in the Administrative County of London
1952	National pre-war av. (1)	J.C. Dawes	Public Health Paper No 5 - The Storage, Collection and Disposal of Domestic Refuse.
1955	1948 (1), 1949 (1), 1950 (2), Average (1)	JC Wylie	Fertility from town's wastes.
1960	National pre-war av. (1) National post-war av. (1) 1937/38 (1) & 1958/59 (1)	A.E. Higginson	The salvage potential of domestic refuse. Reported to the Public Works and Municipal Services Conference
1961	1954 (1)	Ministry of Housing	Report on the Pollution of Water by Tipped Refuse
1965	National post-war av. (1)	F.L. Stirrup	Public Cleansing. Available at: https://www.sciencedirect.com/book/9780080104997/public-cleansing
1966	1954/55 (2), 1961 (2), 1962 (2), 1963 (9), 1964 (3) & 1965 (3)	A.E. Higginson	The Analysis of Domestic Refuse. Institute of Public Cleansing. 45 separate analyses for a range of towns undertaken for Summer, Autumn, Winter and Spring. These are also given as a national average and include up to 15 local authorities. Analyses are included as Appendices 1 to 6 (inclusive) in the publication.
1966	1954/55 (41)	P.D. Fairlie	A review of factors affecting the economic collection of refuse. Separate analyses for a range of towns undertaken for Summer, Autumn, Winter and Spring. Analyses are included as Appendices 5 and 6 in the publication.
1966	1964 (1) 1966 (1)	A.E. Barton & E.J. Ostie	Emissions from Incineration Chimneys. Reported to the Public Works and Municipal Services Conference. Survey undertaken in Birmingham
1967	1959 (4)	R.E. Bevan	Controlled Tipping. Table 8.4 Analysis of Refuse-Manchester by 4 property types: Old terraced/modern semi-detached/high rateable value/modern terrace
1969	1955 (1), 1960 (1) & 1965 (3)	F. Flintoff & R. Millard	Public Cleansing. Maclaren & Sons Chapter 3 Refuse-Definition & Analysis. Given as "typical of domestic refuse"

1971	1935/36 (1), 1967 (1) & 1968 (1)	Department for the Environment	Report of the Working Party on Refuse Disposal. Figures supplied by 15 LAs
1971	1971 (1)	PENECOL Engineering Consultants	Refuse Disposal Study for the Borough of Hove: Report for recommendations for future disposal strategy
1971	1966 (1)	H M Government	HM Royal Commission on Environmental Pollution 1971. https://webarchive.nationalarchives.gov.uk/20110322143936/http://www.rcep.org.uk/reports/01-first%20report/1971-01firstreport.pdf
1972	1972 (1)	PENECOL Engineering Consultants	Refuse Disposal Study for Mid-Sussex: Report for recommendations for future disposal strategy
1977	1971/73 (1) & 1980 (2 – projected)	H-C. Bailly & C. Tayart de Borms	European Commission ISBN 0 86010 080 4: Material Flows in the post-consumer waste stream of the EEC
1979	1934 (1), 1969 (1), 1972 (1) & 1980 (1 – projected)	A.W. Neal	1934, 1969 & 1980: Original data from: Birmingham Borough Council. 1972: Original data from: Coventry Borough Council
1979	1966/67 (1) & 1972/73 (1)	Greater London Council	GLC comprehensive analysis. Data from London Boroughs
1979	1979 (1)	D. Wilson	The uncertain costs of waste disposal and resource recovery. Resource Recovery and Conservation 1979 4 pp. 261 – 299
1979	1963 (1) 1969 (5)	J Skitt	Composition and analysis of Household Waste. In: Waste disposal and management practice
1981	1969 (1) 1970 (3) 1973 (1) 1974 (1) 1975 (1) 1981 (1)	D. C. Wilson	Waste Management: Planning, Evaluation, Technologies
1984	1982 (1)	N.M. Rufford	PhD Study: The analysis and prediction of the quantity and composition of household refuse in Birmingham. http://publications.aston.ac.uk/id/eprint/14241/
1986	1935 – 1979 (15)	A. V. Bridgewater	Refuse composition projections and recycling technology. Resources and Conservation: 12. Pp 159 – 174
1991	1989 – 1990 (1 – civic amenity waste)	C. Coggins et al.	Public awareness and use of civic amenity sites and recycling centres, Department of the Environment report CWM 024/90. Average figures and range given
1993	1993 (4)	WSL & Aspinwall	WSL & Aspinwall Civic Amenity Sites (1993)

1994	1992/3 (5)	Department for the Environment	National Household Waste Analysis Programme (NHWAP)
1995	1992/3 (2)	Richard Waite	Household Waste Recycling (1995). Sampled January
1996	1992/3 (2)	Jones et al.	Analysing household wastes: a new method for the analysis and estimation of household waste arisings,
2000	1996/97 (1)	Environment Agency	A Study of the Composition of Collected Household Waste in the United Kingdom - with Particular Reference to Packaging Waste R&D Technical Report P347 (University of East Anglia Study)
2002	1999 – 2002 (2)	J. Parfitt (WRAP)	Analysis of household waste composition and factors driving waste increases. https://www.researchgate.net/profile/Julian_Parfitt/publications
2003	1994 (2) 1999 (2) 2000 (2)	Emery et al.	An in-depth study of the effects of socio-economic conditions on household waste recycling practices: https://journals.sagepub.com/action/doSearch?AllField=landfill+mining&SeriesKey=wmra&content=articlesChapters&countTerms=true&target=default&sortBy=Ppub&pageSize=20&startPage=9
2004	2003/4 (4)	A. Poll	Variations in the composition of household collected waste, AEAT/ENV/R/1839, AEA Technology, Harwell, UK.
2005	2000 - 2005 (1)	S. Chackiath & P. Longhurst*	Waste Modelling for London: https://www.londoncouncils.gov.uk/download/file/fid/3795
2006	2001 – 2003 (1)	S. Burnley et al.	Assessing the composition of municipal solid waste in Wales. Resources, Conservation and Recycling. 49 pp. 264 - 283
2008	2001 – 2003 (2)	Resources Futures for DEFRA	Defra WR0119: Appendix 8 Compositional data evaluation criteria. http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=15133
2009	2006/07 (1)	Environment Statistics Service, Department for Environment, Food and Rural Affairs, Area 6E Ergon House, 17 Smith Square, London SW1P 3JR, 08459 33 55 77	Municipal Waste Composition: A Review of Municipal Waste Component Analyses

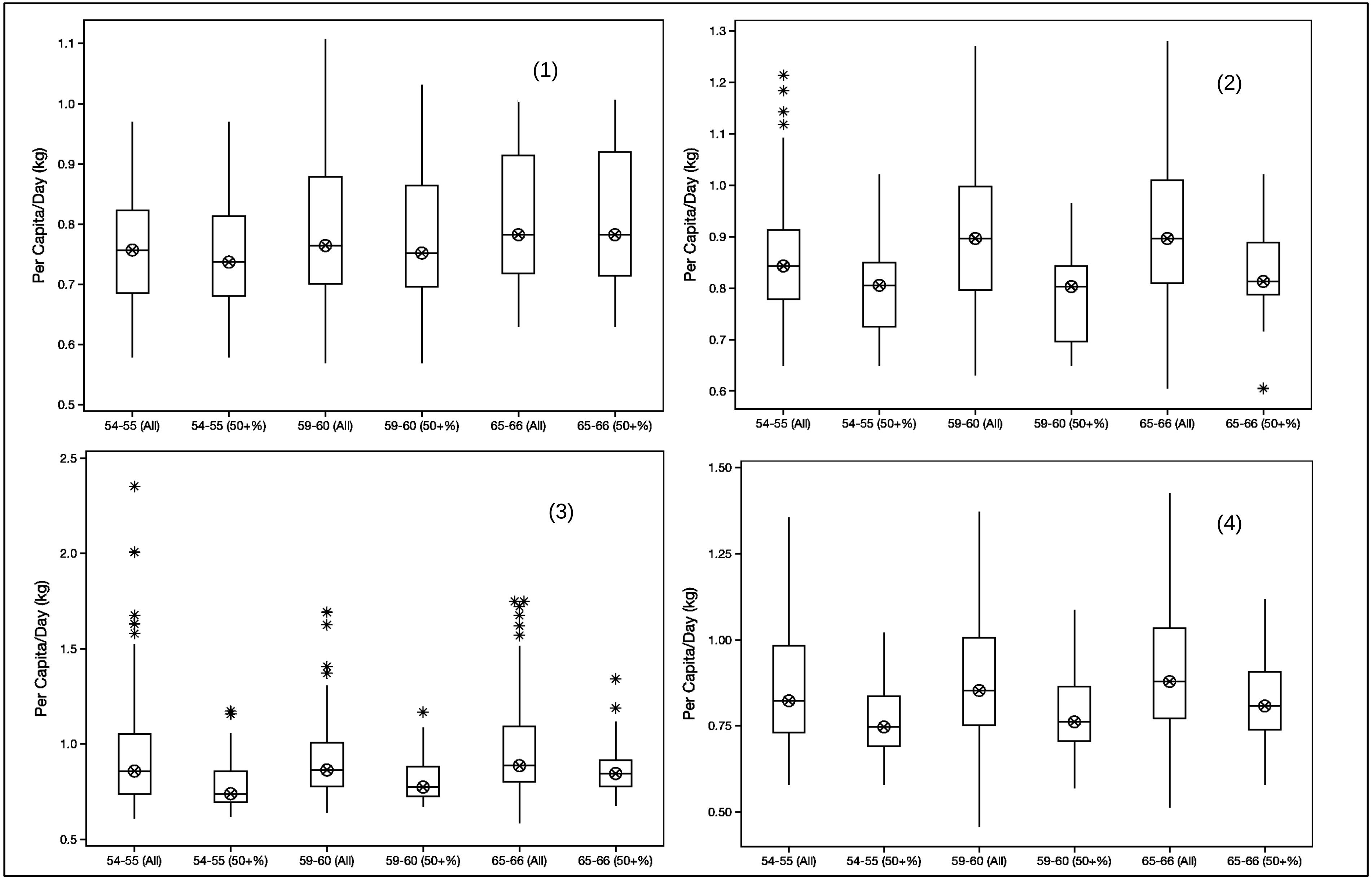
Appendix 3.3 – Formulae for descriptive statistics used by Minitab where *N* is the sample size.

Descriptive statistic	Formula
Mean (average) $(\bar{X})$	$\bar{X} = \frac{\sum_{i=1}^{N} x_i}{N}$
Median	*The mid-term of the data set. For an even number of data elements, the median is the average of the 2 mid-terms.*
Coefficient of variation (CV)	$CV = \frac{\sigma}{\mu}$
Range (R)	$R = maximum - minimum$
Standard deviation (s)	$s = \sqrt{\frac{\sum(x-\bar{x})^2}{N-1}}$
Variance (s^2)	$s^2 = \frac{\sum(x-\bar{x})^2}{N-1}$
Standard error of the mean *(SE Mean)*	$SE\ Mean = \frac{s}{\sqrt{N}}$
Coefficient of variation *(CV) or Relative Standard Deviation (RSD)*	$CV = \frac{s}{\bar{\bar{X}}}$
Interquartile range *(IQR)*	*IQR = 2nd quartile + 3rd quartile* *The IQR relates to the spread of data.*
Kurtosis (b_2)	$b_2 = \frac{N(N+1)}{(N-1)(N-2)(N-3)} \sum \left[\frac{x_i - \bar{x}}{s}\right]^4$
Skewness (b_1)	$b_1 = \frac{N}{(N-1)(N-2)} \sum \left[\frac{x_i - \bar{x}}{s}\right]^3$

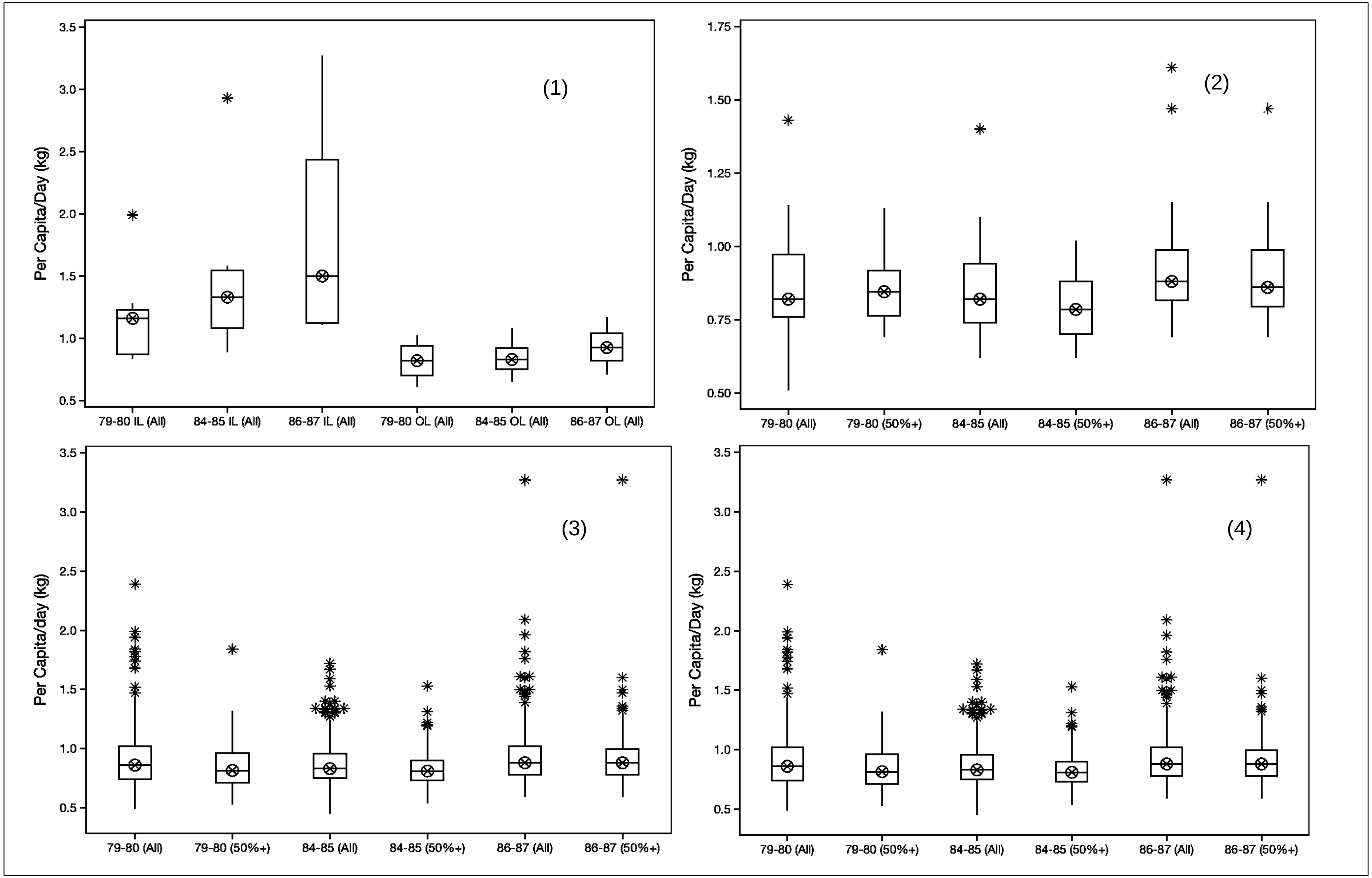
Appendix 3.4 – Supporting data for Figure 3.3

	Mean	Median	Std Deviation	Coefficient of Variation	Mean	Median	Std Deviation	Coefficient of Variation	Mean	Median	Std Deviation	Coefficient of Variation	Mean	Median	Std Deviation	Coefficient of Variation	Mean	Median	Std Deviation	Coefficient of Variation	Mean	Median	Std Deviation	Coefficient of Variation	Mean	Median	Std Deviation	Coefficient of Variation
County borough	0.80	0.81	0.08	10%	0.79	0.8	0.09	11%	0.83	0.81	0.09	11%	0.80	Not given with the published data	Not given with the published data	Not given with the published data	-	-	-	-	-	-	-	-	-	-	-	-
London councils	0.83	0.76	0.28	34%	0.92	0.85	0.30	33%	0.84	0.78	0.19	23%	0.89				-	-	-	-	-	-	-	-	-	-	-	-
Borough councils	0.80	0.74	0.14	18%	0.82	0.78	0.12	15%	0.87	0.85	0.15	17%	0.84				-	-	-	-	-	-	-	-	-	-	-	-
Urban district	0.80	0.68	0.26	32%	0.81	0.91	0.24	30%	0.74	0.69	0.14	19%	0.71				-	-	-	-	-	-	-	-	-	-	-	-
Rural district	-	-	-	-	-	-	-	-	0.60	0.60	0.03	5%	0.74				-	-	-	-	-	-	-	-	-	-	-	-
Inner London	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.05	Not given with the published data	Not given with the published data	Not given with the published data	1.16	1.16	0.32	28%	1.44	1.33	0.60	42%
Outer London	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-					0.82	0.82	0.12	14%	0.84	0.83	0.12	15%
Metropolitan Districts	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.02				0.85	0.85	0.12	14%	0.79	0.79	0.11	14%
Non-met Districts	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.03				0.84	0.82	0.20	24%	0.84	0.81	0.17	20%
	1986/87				1991/92				1995/96				1998/99				1999/00				2002/03				2005/06			
Inner London	1.72	1.50	0.89	52%	1.42	1.43	0.55	39%	1.45	1.47	0.21	15%	1.62	1.69	0.13	8%	1.69	1.67	0.05	3%	1.78	1.78	-	-	1.39	1.43	0.15	11%
Outer London	0.93	0.93	0.14	15%	1.27	1.23	0.18	14%	1.09	1.02	0.22	20%	1.10	1.06	0.13	11%	1.09	1.09	0.07	6%	1.36	1.43	0.14	10%	1.26	1.20	0.14	11%
Metropolitan Districts	0.90	0.86	0.13	15%	1.21	1.19	0.30	25%	1.19	1.13	0.18	15%	1.34	1.13	0.44	33%	1.16	1.12	0.14	12%	1.18	1.20	0.05	4%	1.11	1.10	0.13	12%
Non-met Districts	0.89	0.86	0.17	19%	1.00	0.97	0.25	25%	0.97	0.97	0.13	14%	1.10	1.09	0.2	18%	1.12	1.11	0.17	15%	1.17	1.17	0.16	14%	0.88	0.87	0.18	20%

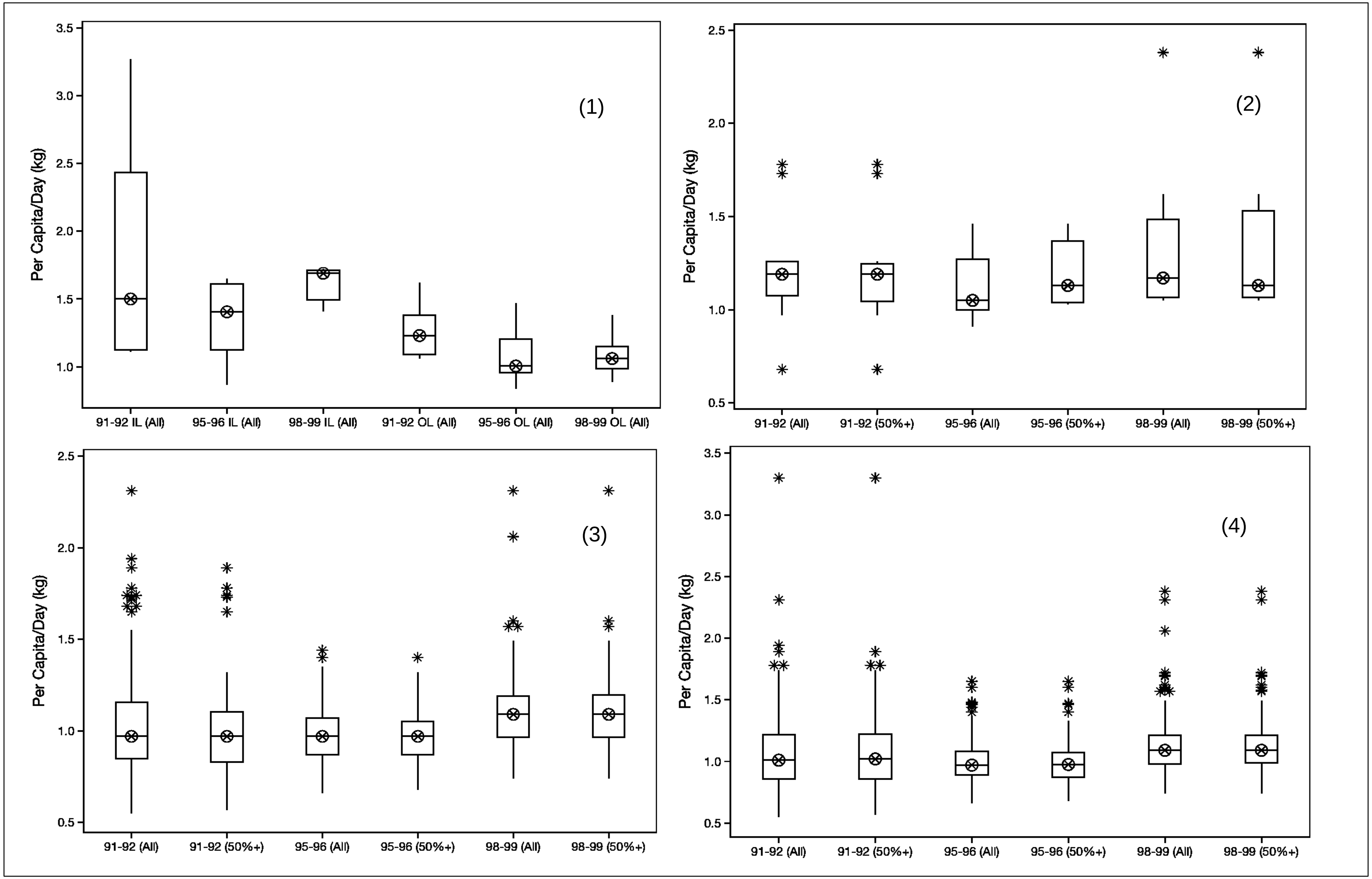
Appendix 3.5(a) - Per capita MSW generation (1) London 1954–66. (2) County Boroughs 1954–66. (2) Boroughs 1954–66. (4) All data1954–66.



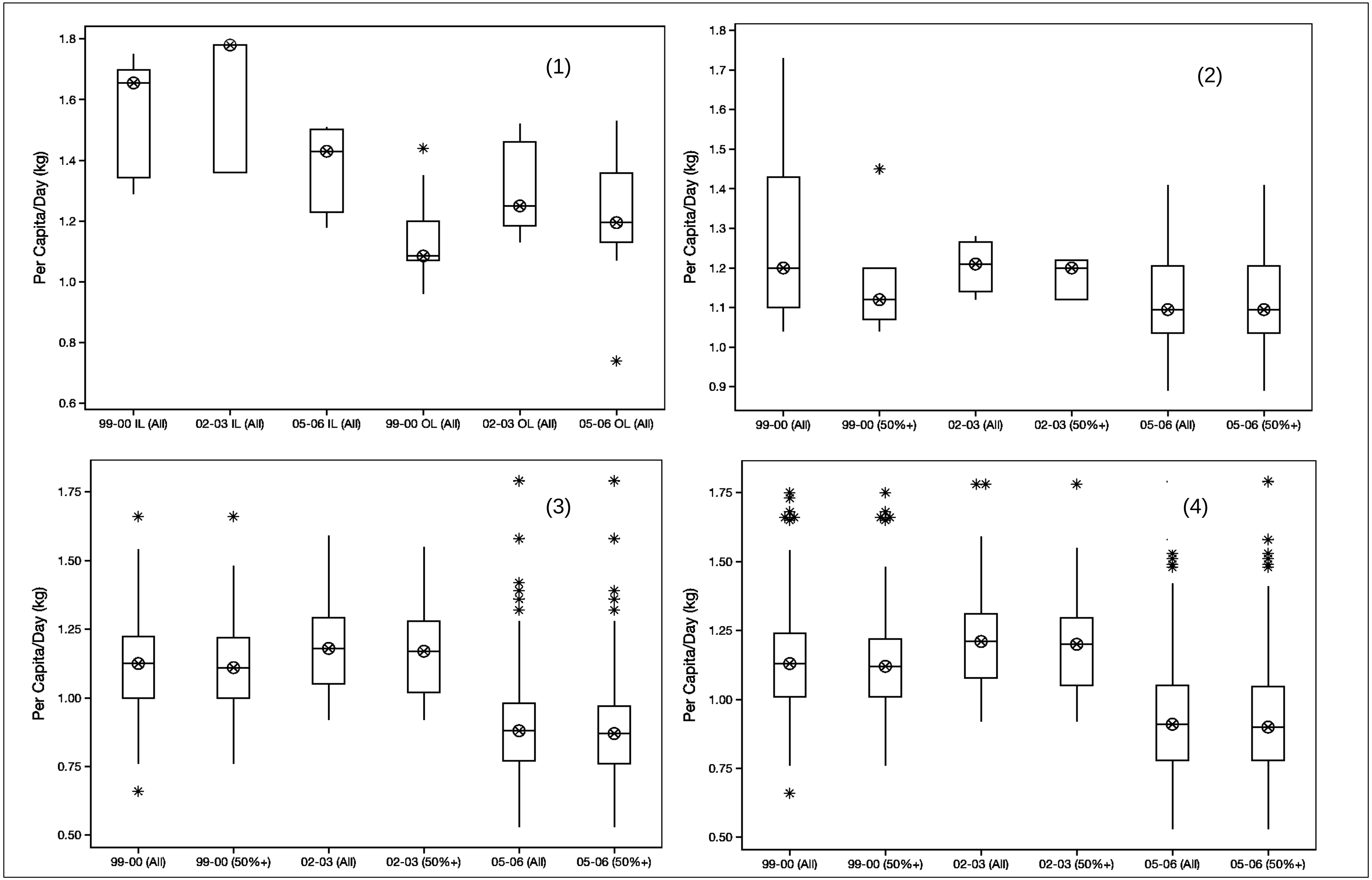
Appendix 3.5(b) - Per capita MSW generation (1) London 1979-87. (2) Non-met Counties1979-87. (3) Districts1979-87. (4) All data 1979-87.



Appendix 3.5(c) - Per capita MSW generation (1) London 1991-99. (2) Non-met Counties1991-99. (3) Districts1991-99. (4) All data 1991-99.



Appendix 3.5(d) - Per capita MSW generation. (1) London 1999-07. (2) Non-met Counties 1999-07. (3) Districts 1999-07. (4) All data 1999-07.



Appendix 4.1 – Bin and civic amenity waste compositions 2000/02

	'BIN WASTE' 'dustbin' residuals + kerbside recycling & non CA bring recycling			CIVIC AMENITY SITE WASTE Total CA residuals + Recycling Excluded: building rubble		
Category	**000's tonnes**	**Kg/Hhld/yr**	**% wt**	**000's tonnes**	**Kg/Hhld/yr**	**% wt**
Newspapers & Magazines	1,501	71	8.1%	71	3	1.3%
Other recyclable paper	1,073	51	5.8%	52	2	0.9%
Liquid cartons	77	4	0.4%	1	0	0.0%
Board packaging	228	11	1.2%	90	4	1.6%
Card and paper packaging	646	31	3.5%	2	0	0.0%
Other card	29	1	0.2%	5	0	0.1%
Non-recyclable paper	638	30	3.5%	14	1	0.3%
Plastic Bottles	388	18	2.1%	7	0	0.1%
Other dense plastic packaging	395	19	2.1%	10	0	0.2%
Other dense plastic	114	5	0.6%	33	2	0.6%
Plastic film	733	35	4.0%	18	1	0.3%
Textiles	589	28	3.2%	111	5	2.0%
Glass bottles and jars	1,463	69	7.9%	69	3	1.2%
Other glass	95	4	0.5%	13	1	0.2%
Wood	507	24	2.7%	488	23	8.8%
Furniture	49	2	0.3%	255	12	4.6%
Disposable nappies	444	21	2.4%	0	0	0.0%
Other Miscellaneous combustibles	111	5	0.6%	127	6	2.3%
Miscellaneous non-combustibles	382	18	2.1%	827	39	15.0%
Metal cans & foil	622	29	3.4%	1	0	0.0%
Other non-ferrous metals	0	0	0.0%	5	0	0.1%
Scrap metal/white goods	544	26	2.9%	535	25	9.7%
Batteries	0	0	0.0%	12	1	0.2%
Engine oil	0	0	0.0%	7	0	0.1%
Garden waste	2,824	134	15.3%	2,078	98	37.6%
Soil & other organic waste	211	10	1.1%	624	30	11.3%
Kitchen waste	2,234	106	12.1%	17	1	0.3%
Non-home compostable kitchen waste	1,865	88	10.1%	0	0	0.0%
Fines	682	32	3.7%	50	2	0.9%
TOTAL	**18,441**	**872**	**100.0%**	**5,521**	**261**	**100.0%**

Appendix 5.1 – Individual MSW generation rates (kg/day) 1945 – 2008. Green background Epoch 1, blue background Epoch 2, yellow background Epoch 3 and grey background Epoch 4. Years estimated 26/63 years are identified in red.

Year	Generation Rate - LA collected H/H (kg/day)	Generation Rate – LA Adjusted (kg/day)	Generation Rate - All MSW (kg/day)	Population	Source/Method
1945	0.6704	0.6704	0.6704	40175830	Extrapolation based on JC Wylie and post war rationing
1946	0.6704	0.6704	0.6704	40175830	Extrapolation based on JC Wylie and post war rationing
1947	0.6704	0.6704	0.6704	40509811	Extrapolation based on JC Wylie and post war rationing
1948	0.6704	0.6704	0.6704	40943939	Extrapolation based on JC Wylie and post war rationing
1949	0.6704	0.6704	0.6704	41118000	Extrapolation based on JC Wylie and post war rationing
1950	0.6704	0.6704	0.6704	41432631	JC Wylie quoted 10,000,000 tonnes annually
1951	0.7357	0.7357	0.7357	41218150	Estimated average based on Interpolation
1952	0.7357	0.7357	0.7357	41347767	Estimated average based on Interpolation
1953	0.8010	0.8010	0.8010	41502014	Taken from MoH[1] 1954/55 Average for Column B[2] 50+ (tonnes/day) (including London)
1954	0.8010	0.8010	0.8010	41669446	Taken from MoH 1954/55 Average for Column B 50+ (tonnes/day) (including London)
1955	0.8010	0.8670	0.8010	41833690	Taken from MoH 1954/55 Average for Column B 50+ (tonnes/day) (including London)
1956	0.7960	0.7420	0.7960	42055950	Taken from MoH 1955/56 Average for Column B 50+ (tonnes/day) Column C[2] all data
1957	0.8060	1.2360	0.8060	42285170	Taken from MoH 1956/57 Average for Column B 50+ (tonnes/day) Column C all data
1958	0.8030	0.8930	0.8030	42483500	Taken from MoH 1957/58 Average for Column B 50+ (tonnes/day) Column C all data
1959	0.8134	0.9210	0.8134	42757500	Columns B/D[2] taken from Report of Working Party on Refuse Collection 1967
1960	0.8134	0.9210	0.8134	43143900	Columns B/D taken from Report of Working Party on Refuse Collection 1968
1961	0.8134	0.9210	0.8134	43552177	Estimated average of known based on MoH
1962	0.8210	0.9210	0.8210	44001676	Estimated average of known based on MoH
1963	0.8210	0.9490	0.8210	44305201	Taken from MoH 1962/63 Average for Column B 50+ (tonnes/day) Column C from all data

1964	0.8500	0.9640	0.8500	44641645	Taken from MoH 1963/64 Average for Column B 50+ (tonnes/day) Column C from all data
1965	0.8179	0.9400	0.8179	44975515	Taken from DoE[3] Report 1971 Columns B/D Column C estimated
1966	0.8330	0.9100	0.8330	45260392	Taken from MoH 1965/66 Average for Column B 50+ (tonnes/day) Column C all data
1967	0.7900	0.9100	0.7900	45555246	Taken from DoE Report 1971 Columns B/D Column C estimated
1968	0.7900	0.9428	0.7900	45791812	Taken from DoE Report 1971 Columns B/D Column C estimated
1969	0.8000	0.9428	0.8000	46017078	Taken from DoE Report 1971 Columns B/D Column C estimated
1970	0.8453	0.9755	0.8453	46167171	Estimated average based on Interpolation
1971	0.8905	0.9755	0.8905	46420796	Estimated average based on Interpolation
1972	0.8905	0.9932	0.8905	46591941	Estimated average based on Interpolation
1973	0.9192	0.9932	0.9192	46719886	Estimated average based on Interpolation
1974	0.9479	1.0110	0.9479	46727015	DoE 1974-78 Waste disposal stats Column B no outliers Column C including outliers
1975	0.9479	1.0110	0.9479	46728343	DoE 1974-78 Waste disposal stats Column B no outliers Column C including outliers
1976	0.8932	0.9397	0.8932	46719381	DoE 1974-78 Waste disposal stats Column B no outliers Column C including outliers
1977	0.8767	1.3100	0.8767	46703188	DoE 1974-78 Waste disposal stats Column B no outliers Column C including outliers
1978	0.9151	0.9699	0.9151	46691966	DoE 1974-78 Waste disposal stats Column B no outliers Column C including outliers
1979	0.9168	1.4806	0.9168	46744344	Estimated Column B Column C CIPFA WDA[4]
1980	0.9185	1.5673	0.9185	46825822	CIPFA[5] WCA[6] Col B Col C CIPFA WDA
1981	0.9478	1.5223	0.9478	46843800	Estimated Column B Column C CIPFA WDA
1982	0.9478	1.4943	0.9478	46777300	Estimated Column B Column C CIPFA WDA
1983	0.9478	1.5200	0.9478	46813700	Estimated Column B Column C CIPFA WDA
1984	0.9770	1.5721	0.9770	46912400	Estimated Column B Estimated Column C
1985	0.9770	1.6242	0.9770	47057400	CIPFA WCA Column B Column C CIPFA WDA
1986	1.0445	1.5400	1.0445	47187006	Estimated Column B Estimated Column C
1987	1.1120	1.4557	1.1120	47300400	CIPFA WCA Col B Col C CIPFA WDA
1988	1.1566	1.4143	1.1566	47412300	Est Col B Est Col C
1989	1.2011	1.3729	1.2011	47552700	Est Col B Est Col C

1990	1.2456	1.3315	1.2456	47699100	Est Col B Est Col C
1991	1.2901	1.2901	1.2901	47875000	CIPFA WCA Col B Col C CIPFA WDC
1992	1.2691	1.3226	1.2691	47998000	Estimated average based on Interpolation
1993	1.2481	1.3551	1.2481	48102300	Estimated average based on Interpolation
1994	1.2271	1.3876	1.2271	48228800	Estimated average based on Interpolation
1995	1.2061	1.4200	1.2061	48383500	CIPFA WCA Column B Column C/D DETR[7]
1996	1.1752	1.5500	1.5500	48519100	CIPFA WCA Column B Column C/D DETR
1997	1.2695	1.2695	1.3843	48664800	CIPFA WCA Column B Column C/D DEFRA[8] 2003
1998	1.2654	1.3094	1.4429	48820600	CIPFA WCA Column B Column C/D DEFRA 2003
1999	1.2840	1.3148	1.4716	49032900	CIPFA WCA Column B Column C/D DEFRA 2003
2000	1.3094	1.3778	1.5292	49233300	CIPFA WCA Column B Column C/D DEFRA 2003
2001	1.3718	1.3895	1.5545	49449778	CIPFA WCA Column B Column C/D DEFRA 2003
2002	1.3831	1.4114	1.5895	49679267	CIPFA WCA Column B Column C/D DEFRA 2003
2003	1.3253	1.4169	1.6084	49925517	CIPFA WCA Column B Column C/D DEFRA 2004
2004	1.3890	1.3890	2.1587	50194600	https://www.gov.uk/government/publications/uk-waste-data
2005	1.3891	1.3891	2.1005	50606034	Estimated average based on Interpolation
2006	1.3683	1.3683	2.0424	50965186	WasteDataFlow, Department for Environment, Food and Rural Affairs (Defra)
2007	1.3742	1.3744	2.0256	51381100	WasteDataFlow, Department for Environment, Food and Rural Affairs (Defra)
2008	1.3371	1.3371	2.0089	51815900	WasteDataFlow, Department for Environment, Food and Rural Affairs (Defra)

Notes:
LA collected includes MoH, 1974 - 78 excluding outliers CIPFA collected. LA Adjusted includes: 1974 - 78 Outliers, WDA figures and DETR/DEFRA HH. LA All includes MoH, 1974 - 78 excl. outliers CIPFA collected and all MSW like materials from 1995/96

1. MoH: Ministry of Housing and Local Government
2. Column B is Generation Rate - LA collected H/H, Column C is Generation Rate – LA Adjusted and Column D is Generation Rate - All MSW
3. Department of Environment
4. Waste Disposal Authority
5. Chartered Institute of Public Finance and Accountancy
6. Waste Collection Authority
7. Department for Environment, Transport and Regions
8. Department for Environment, Food and Rural Affairs

Appendix 5.2 – MSW disposal rates 1945 – 2008 (% expressed as decimal). Green background Epoch 1, blue background Epoch 2, yellow background Epoch 3 and grey background Epoch 4. Years estimated 38/63 years are identified in red.

Year	Percentage waste to Landfill	Percentage waste to Incineration	Percentage waste to Salvage & Recycle	Percentage waste to Other Disposal	Source/Method
1945	0.85	0.13	0.02	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1946	0.85	0.13	0.02	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1947	0.85	0.13	0.02	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1948	0.84	0.13	0.03	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1949	0.84	0.13	0.03	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1950	0.84	0.13	0.03	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1951	0.84	0.13	0.03	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1952	0.84	0.13	0.03	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1953	0.84	0.13	0.03	0.00	Estimated average based on 1954 stats & recycle based on Higginson's figs for paper
1954	0.84	0.13	0.03	0.00	Taken from MoH 1954/55
1955	0.83	0.13	0.04	0.00	Estimated average based on Interpolation also determined by incineration and recycle rates
1956	0.84	0.13	0.03	0.00	Estimated average based on Interpolation also determined by incineration and recycle rates
1957	0.84	0.13	0.03	0.00	Estimated average based on Interpolation also determined by incineration and recycle rates
1958	0.84	0.13	0.03	0.00	Estimated average based on Interpolation also determined by incineration and recycle rates
1959	0.84	0.13	0.03	0.00	Estimated average based on Interpolation also determined by incineration and recycle rates
1960	0.87	0.13	0.00	0.00	Taken from MoH 1967
1961	0.88	0.12	0.00	0.00	Estimated average based on Interpolation also determined by incineration
1962	0.88	0.12	0.00	0.00	Estimated average based on Interpolation also determined by incineration
1963	0.89	0.11	0.00	0.00	Estimated average based on Interpolation also determined by incineration
1964	0.89	0.11	0.00	0.00	Estimated average based on Interpolation also determined by incineration

1965	0.90	0.10	0.00	0.00	Estimated average based on Interpolation also determined by incineration
1966	0.90	0.10	0.00	0.00	Estimated average based on Interpolation also determined by incineration
1967	0.90	0.09	0.00	0.01	Taken from DoE Report 1967
1968	0.90	0.09	0.00	0.01	Taken from DoE Report 1968
1969	0.90	0.09	0.00	0.01	Taken from DoE Report 1969
1970	0.90	0.09	0.00	0.01	Taken from DoE Report 1970
1971	0.90	0.09	0.00	0.01	Taken from DoE Report 1971
1972	0.90	0.09	0.00	0.01	Estimated average based on Interpolation
1973	0.91	0.09	0.00	0.01	Estimated average based on Interpolation
1974	0.91	0.09	0.00	0.01	DofE 1974-78 Waste disposal stats Col B no outliers Col C incl outliers
1975	0.91	0.09	0.00	0.01	DofE 1974-78 Waste disposal stats Col B no outliers Col C incl outliers
1976	0.90	0.09	0.00	0.01	DofE 1974-78 Waste disposal stats Col B no outliers Col C incl outliers
1977	0.90	0.10	0.00	0.00	DofE 1974-78 Waste disposal stats Col B no outliers Col C incl outliers
1978	0.89	0.10	0.00	0.01	DofE 1974-78 Waste disposal stats Col B no outliers Col C incl outliers
1979	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1980	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1981	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1982	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1983	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1984	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1985	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1986	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1987	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1988	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates

1989	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1990	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1991	0.89	0.10	0.00	0.01	Estimated average based on Interpolation also determined by incineration and recycle rates
1992	0.86	0.10	0.03	0.01	1991-10 UK Gov web archive Household recycling
1993	0.86	0.10	0.03	0.01	Estimated average based on Interpolation
1994	0.86	0.10	0.03	0.01	1991-10 UK Gov web archive Household recycling
1995	0.83	0.10	0.07	0.00	DETR
1996	0.83	0.10	0.07	0.00	DETR
1997	0.84	0.09	0.07	0.00	DEFRA 2003
1998	0.85	0.07	0.08	0.00	DEFRA 2003
1999	0.82	0.08	0.10	0.00	DEFRA 2003
2000	0.80	0.08	0.11	0.01	DEFRA 2003
2001	0.79	0.09	0.12	0.01	DEFRA Regional Waste by type 2000 - 2017
2002	0.78	0.08	0.14	0.00	DEFRA Regional Waste by type 2000 - 2018
2003	0.75	0.09	0.16	0.01	DEFRA Regional Waste by type 2000 - 2019
2004	0.72	0.09	0.19	0.00	DEFRA Regional Waste by type 2000 - 2020
2005	0.67	0.10	0.24	0.00	DEFRA Regional Waste by type 2000 - 2021
2006	0.62	0.10	0.27	0.01	DEFRA Regional Waste by type 2000 - 2022
2007	0.58	0.11	0.31	0.00	DEFRA Regional Waste by type 2000 - 2023
2008	0.50	0.12	0.37	0.01	DEFRA Regional Waste by type 2008 - 2009

Appendix 5.3 – MSW composition 1945 – 2008 (% expressed as decimal). Green background Epoch 1, blue background Epoch 2, yellow background Epoch 3 and grey background Epoch 4. Years estimated 23/63 years are identified in red.

Year	Fines (L)	Fines (Av)	Fines (H)	Organic (L)	Organic (Av)	Organic[1] (H)	P&C[2] (L)	P&C (Av)	P&C (H)	Glass (L)	Glass (Av)	Glass (H)
1945	0.570	0.638	0.706	0.071	0.104	0.137	0.097	0.120	0.143	0.028	0.031	0.034
1946	0.538	0.628	0.718	0.049	0.073	0.097	0.081	0.106	0.131	0.030	0.044	0.057
1947	0.505	0.617	0.729	0.028	0.042	0.056	0.065	0.092	0.119	0.032	0.056	0.081
1948	0.505	0.617	0.729	0.028	0.042	0.056	0.065	0.092	0.119	0.032	0.056	0.081
1949	0.672	0.680	0.689	0.027	0.039	0.050	0.073	0.083	0.092	0.042	0.048	0.054
1950	0.672	0.680	0.689	0.027	0.039	0.050	0.073	0.083	0.092	0.042	0.048	0.054
1951	0.672	0.680	0.689	0.027	0.039	0.050	0.073	0.083	0.092	0.042	0.048	0.054
1952	0.632	0.642	0.651	0.034	0.049	0.064	0.120	0.122	0.124	0.050	0.051	0.053
1953	0.632	0.642	0.651	0.034	0.049	0.064	0.120	0.122	0.124	0.050	0.051	0.053
1954	0.632	0.642	0.651	0.034	0.049	0.064	0.120	0.122	0.124	0.050	0.051	0.053
1955	0.632	0.642	0.651	0.034	0.049	0.064	0.120	0.122	0.124	0.050	0.051	0.053
1956	0.592	0.601	0.610	0.043	0.051	0.058	0.126	0.128	0.131	0.055	0.057	0.059
1957	0.592	0.601	0.610	0.043	0.051	0.058	0.126	0.128	0.131	0.055	0.057	0.059
1958	0.552	0.560	0.568	0.051	0.052	0.052	0.131	0.134	0.137	0.061	0.063	0.066
1959	0.552	0.560	0.568	0.051	0.052	0.052	0.131	0.134	0.137	0.061	0.063	0.066
1960	0.464	0.483	0.502	0.080	0.085	0.089	0.172	0.194	0.216	0.066	0.070	0.073
1961	0.464	0.483	0.502	0.080	0.085	0.089	0.172	0.194	0.216	0.066	0.070	0.073
1962	0.375	0.406	0.436	0.110	0.117	0.125	0.213	0.254	0.295	0.071	0.076	0.081
1963	0.403	0.444	0.485	0.091	0.139	0.186	0.222	0.259	0.297	0.074	0.085	0.097
1964	0.363	0.411	0.458	0.090	0.123	0.156	0.270	0.291	0.312	0.064	0.068	0.072
1965	0.324	0.348	0.372	0.132	0.150	0.174	0.300	0.311	0.322	0.066	0.071	0.076
1966	0.284	0.284	0.285	0.175	0.176	0.192	0.329	0.331	0.333	0.067	0.073	0.080
1967	0.193	0.251	0.310	0.152	0.172	0.192	0.295	0.311	0.327	0.071	0.090	0.109
1968	0.168	0.205	0.241	0.173	0.186	0.199	0.338	0.366	0.394	0.050	0.079	0.107
1969	0.143	0.158	0.172	0.195	0.200	0.206	0.380	0.421	0.461	0.030	0.067	0.105

1970	0.140	0.141	0.142	0.187	0.243	0.298	0.305	0.380	0.455	0.075	0.086	0.097
1971	0.140	0.141	0.142	0.187	0.243	0.298	0.290	0.355	0.419	0.075	0.086	0.097
1972	0.143	0.159	0.176	0.176	0.240	0.304	0.274	0.328	0.382	0.091	0.102	0.113
1973	0.109	0.149	0.190	0.180	0.229	0.278	0.330	0.353	0.376	0.050	0.070	0.089
1974	0.174	0.172	0.198	0.156	0.191	0.221	0.293	0.333	0.353	0.060	0.075	0.106
1975	0.183	0.194	0.205	0.139	0.152	0.165	0.294	0.312	0.330	0.070	0.079	0.087
1976	0.168	0.178	0.187	0.155	0.167	0.179	0.301	0.324	0.348	0.075	0.083	0.090
1977	0.152	0.161	0.169	0.170	0.182	0.193	0.307	0.336	0.365	0.080	0.087	0.092
1978	0.136	0.144	0.151	0.185	0.197	0.207	0.313	0.348	0.383	0.085	0.091	0.095
1979	0.120	0.127	0.133	0.200	0.211	0.221	0.319	0.360	0.400	0.090	0.094	0.097
1980	0.100	0.110	0.120	0.170	0.170	0.170	0.421	0.426	0.430	0.090	0.104	0.117
1981	0.071	0.076	0.081	0.274	0.274	0.274	0.339	0.341	0.343	0.092	0.099	0.106
1982	0.042	0.042	0.042	0.377	0.377	0.377	0.257	0.257	0.257	0.094	0.094	0.094
1983	0.045	0.046	0.048	0.355	0.357	0.360	0.258	0.263	0.267	0.092	0.093	0.094
1984	0.048	0.051	0.054	0.333	0.337	0.342	0.260	0.269	0.278	0.090	0.092	0.093
1985	0.048	0.051	0.054	0.333	0.337	0.342	0.260	0.269	0.278	0.090	0.092	0.093
1986	0.051	0.056	0.061	0.311	0.317	0.324	0.262	0.276	0.289	0.088	0.091	0.092
1987	0.054	0.061	0.067	0.289	0.297	0.306	0.263	0.282	0.300	0.086	0.089	0.091
1988	0.058	0.066	0.074	0.267	0.277	0.288	0.265	0.289	0.311	0.084	0.088	0.090
1989	0.061	0.071	0.080	0.245	0.257	0.270	0.266	0.295	0.322	0.082	0.086	0.089
1990	0.061	0.071	0.080	0.245	0.257	0.270	0.266	0.295	0.322	0.082	0.086	0.089
1991	0.064	0.076	0.086	0.267	0.237	0.252	0.268	0.301	0.333	0.080	0.085	0.088
1992	0.067	0.080	0.092	0.200	0.217	0.234	0.269	0.307	0.344	0.078	0.083	0.087
1993	0.031	0.031	0.031	0.303	0.319	0.335	0.264	0.312	0.359	0.057	0.061	0.064
1994	0.031	0.031	0.031	0.303	0.319	0.335	0.264	0.312	0.359	0.057	0.061	0.064
1995	0.003	0.003	0.003	0.440	0.440	0.440	0.231	0.235	0.239	0.061	0.061	0.061
1996	0.003	0.003	0.003	0.440	0.440	0.440	0.231	0.235	0.239	0.061	0.061	0.061
1997	0.007	0.012	0.017	0.338	0.372	0.405	0.206	0.252	0.295	0.066	0.066	0.067
1998	0.007	0.012	0.017	0.338	0.372	0.405	0.206	0.251	0.295	0.066	0.066	0.067
1999	0.010	0.020	0.030	0.236	0.303	0.370	0.180	0.266	0.351	0.070	0.071	0.072

2000	0.010	0.020	0.030	0.236	0.303	0.370	0.180	0.266	0.351	0.070	0.071	0.072
2001	0.010	0.020	0.030	0.236	0.303	0.370	0.180	0.266	0.351	0.070	0.071	0.072
2002	0.010	0.020	0.030	0.236	0.303	0.370	0.180	0.266	0.351	0.070	0.071	0.072
2003	0.006	0.012	0.017	0.286	0.335	0.384	0.223	0.238	0.252	0.062	0.064	0.066
2004	0.006	0.012	0.017	0.286	0.335	0.384	0.223	0.238	0.252	0.062	0.064	0.066
2005	0.046	0.046	0.046	0.351	0.351	0.351	0.236	0.236	0.236	0.072	0.072	0.072
2006	0.000	0.009	0.018	0.337	0.371	0.406	0.181	0.204	0.227	0.049	0.057	0.064
2007	0.000	0.009	0.018	0.337	0.371	0.406	0.181	0.204	0.227	0.049	0.057	0.064
2008	0.000	0.009	0.018	0.337	0.371	0.406	0.181	0.204	0.227	0.049	0.057	0.064

1945	Textiles (L)	Textiles (Av)	Textiles (H)	Metals (L)	Metals (Av)	Metals (H)	Comb[3] (L)	Comb (Av)	Comb (H)	Contain[4] (L)	Contain (Av)	Contain (H)
1946	0.006	0.012	0.019	0.040	0.047	0.053	0.014	0.019	0.024	0.028	0.029	0.031
1947	0.012	0.019	0.026	0.039	0.054	0.069	0.031	0.033	0.036	0.028	0.029	0.031
1948	0.019	0.026	0.034	0.037	0.061	0.084	0.048	0.048	0.048	0.028	0.029	0.031
1949	0.019	0.026	0.034	0.037	0.061	0.084	0.048	0.048	0.048	0.028	0.029	0.031
1950	0.008	0.014	0.020	0.059	0.060	0.061	0.031	0.031	0.031	0.028	0.029	0.031
1951	0.008	0.014	0.020	0.059	0.060	0.061	0.031	0.031	0.031	0.028	0.029	0.031
1952	0.008	0.014	0.020	0.059	0.060	0.061	0.031	0.031	0.031	0.028	0.029	0.031
1953	0.013	0.013	0.013	0.009	0.010	0.011	0.021	0.022	0.024	0.032	0.035	0.038
1954	0.013	0.013	0.013	0.009	0.010	0.011	0.021	0.022	0.024	0.032	0.035	0.038
1955	0.013	0.013	0.013	0.009	0.010	0.011	0.021	0.022	0.024	0.032	0.035	0.038
1956	0.013	0.013	0.013	0.009	0.010	0.011	0.021	0.022	0.024	0.032	0.035	0.038
1957	0.017	0.018	0.019	0.005	0.020	0.034	0.022	0.035	0.049	0.052	0.052	0.052
1958	0.017	0.018	0.019	0.005	0.020	0.034	0.022	0.035	0.049	0.052	0.052	0.052
1959	0.022	0.023	0.024	0.002	0.029	0.057	0.022	0.048	0.073	0.052	0.052	0.052
1960	0.022	0.023	0.024	0.002	0.029	0.057	0.022	0.048	0.073	0.052	0.052	0.052
1961	0.020	0.023	0.025	0.034	0.049	0.064	0.022	0.048	0.073	0.000	0.000	0.000
1962	0.020	0.023	0.025	0.034	0.049	0.064	0.022	0.048	0.073	0.000	0.000	0.000
1963	0.018	0.022	0.026	0.066	0.069	0.071	0.022	0.048	0.073	0.000	0.000	0.000
1964	0.023	0.025	0.028	0.066	0.078	0.091	0.022	0.048	0.073	0.000	0.000	0.000
1965	0.011	0.016	0.021	0.048	0.049	0.050	0.022	0.048	0.073	0.000	0.000	0.000

1966	0.017	0.020	0.023	0.060	0.069	0.078	0.022	0.048	0.073	0.000	0.000	0.000
1967	0.023	0.024	0.024	0.071	0.089	0.106	0.022	0.048	0.073	0.000	0.000	0.000
1968	0.021	0.023	0.024	0.080	0.093	0.106	0.022	0.048	0.073	0.000	0.000	0.000
1969	0.013	0.014	0.015	0.071	0.087	0.102	0.022	0.037	0.052	0.000	0.000	0.000
1970	0.023	0.025	0.027	0.062	0.080	0.097	0.022	0.026	0.030	0.000	0.000	0.000
1971	0.020	0.025	0.030	0.075	0.081	0.086	0.022	0.026	0.030	0.000	0.000	0.000
1972	0.020	0.025	0.030	0.075	0.081	0.086	0.022	0.026	0.030	0.000	0.000	0.000
1973	0.028	0.033	0.038	0.071	0.073	0.076	0.022	0.026	0.030	0.000	0.000	0.000
1974	0.021	0.028	0.035	0.091	0.096	0.100	0.022	0.026	0.030	0.000	0.000	0.000
1975	0.032	0.043	0.054	0.077	0.084	0.096	0.050	0.050	0.050	0.000	0.000	0.000
1976	0.043	0.059	0.074	0.063	0.071	0.078	0.050	0.050	0.050	0.000	0.000	0.000
1977	0.040	0.053	0.065	0.068	0.074	0.079	0.050	0.050	0.050	0.000	0.000	0.000
1978	0.037	0.046	0.055	0.072	0.076	0.080	0.050	0.050	0.050	0.000	0.000	0.000
1979	0.034	0.039	0.045	0.076	0.079	0.081	0.050	0.050	0.050	0.000	0.000	0.000
1980	0.030	0.033	0.035	0.080	0.081	0.082	0.050	0.050	0.050	0.000	0.000	0.000
1981	0.019	0.025	0.030	0.090	0.100	0.110	0.050	0.050	0.050	0.000	0.000	0.000
1982	0.023	0.026	0.028	0.078	0.083	0.088	0.051	0.051	0.051	0.000	0.000	0.000
1983	0.026	0.026	0.026	0.065	0.065	0.065	0.051	0.051	0.051	0.000	0.000	0.000
1984	0.026	0.026	0.026	0.066	0.066	0.067	0.049	0.049	0.050	0.000	0.000	0.000
1985	0.026	0.026	0.026	0.067	0.067	0.068	0.046	0.047	0.048	0.000	0.000	0.000
1986	0.026	0.026	0.026	0.067	0.067	0.068	0.046	0.047	0.048	0.000	0.000	0.000
1987	0.026	0.026	0.026	0.068	0.068	0.069	0.044	0.045	0.046	0.000	0.000	0.000
1988	0.025	0.025	0.025	0.068	0.069	0.070	0.041	0.043	0.044	0.000	0.000	0.000
1989	0.025	0.025	0.025	0.069	0.070	0.071	0.039	0.041	0.043	0.000	0.000	0.000
1990	0.025	0.025	0.025	0.070	0.071	0.072	0.036	0.039	0.041	0.000	0.000	0.000
1991	0.025	0.025	0.025	0.070	0.071	0.072	0.036	0.039	0.041	0.000	0.000	0.000
1992	0.025	0.025	0.025	0.071	0.072	0.073	0.034	0.037	0.039	0.000	0.000	0.000
1993	0.024	0.024	0.024	0.071	0.073	0.074	0.031	0.034	0.037	0.000	0.000	0.000
1994	0.044	0.046	0.048	0.039	0.046	0.053	0.021	0.027	0.032	0.000	0.000	0.000
1995	0.044	0.046	0.048	0.039	0.046	0.053	0.021	0.027	0.032	0.000	0.000	0.000
1996	0.030	0.030	0.030	0.047	0.047	0.047	0.055	0.055	0.055	0.000	0.000	0.000
1997	0.030	0.030	0.030	0.047	0.047	0.047	0.055	0.055	0.055	0.000	0.000	0.000
1998	0.027	0.029	0.030	0.059	0.067	0.075	0.028	0.040	0.051	0.000	0.000	0.000

1999	0.027	0.029	0.030	0.059	0.067	0.075	0.028	0.040	0.051	0.000	0.000	0.000
2000	0.024	0.027	0.030	0.070	0.086	0.102	0.001	0.024	0.046	0.000	0.000	0.000
2001	0.024	0.027	0.030	0.070	0.086	0.102	0.001	0.024	0.046	0.000	0.000	0.000
2002	0.024	0.027	0.030	0.070	0.086	0.102	0.001	0.024	0.046	0.000	0.000	0.000
2003	0.024	0.027	0.030	0.070	0.086	0.102	0.001	0.024	0.046	0.000	0.000	0.000
2004	0.015	0.021	0.027	0.036	0.041	0.046	0.058	0.069	0.080	0.000	0.000	0.000
2005	0.015	0.021	0.027	0.036	0.041	0.046	0.058	0.069	0.080	0.000	0.000	0.000
2006	0.024	0.024	0.024	0.046	0.046	0.046	0.046	0.046	0.046	0.000	0.000	0.000
2007	0.028	0.032	0.036	0.034	0.039	0.043	0.014	0.026	0.037	0.000	0.000	0.000
2008	0.028	0.032	0.036	0.034	0.039	0.043	0.014	0.026	0.037	0.000	0.000	0.000

1945	Sanitary (L)	Sanitary (Av)	Sanitary (H)	Plastic (L)	Plastic (Av)	Plastic (H)	WEEE (L)	WEEE (Av)	WEEE (H)
1946	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1947	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1948	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1949	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1950	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1951	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1952	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1953	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1954	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1955	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1956	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1957	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1958	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1959	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1960	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1961	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1962	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1963	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1964	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1965	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1966	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000

1967	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1968	0.000	0.000	0.000	0.012	0.013	0.015	0.000	0.000	0.000
1969	0.000	0.000	0.000	0.010	0.012	0.014	0.000	0.000	0.000
1970	0.000	0.000	0.000	0.008	0.011	0.014	0.000	0.000	0.000
1971	0.000	0.000	0.000	0.015	0.021	0.027	0.000	0.000	0.000
1972	0.000	0.000	0.000	0.015	0.021	0.027	0.000	0.000	0.000
1973	0.000	0.000	0.000	0.024	0.024	0.025	0.000	0.000	0.000
1974	0.000	0.000	0.000	0.015	0.019	0.023	0.000	0.000	0.000
1975	0.000	0.000	0.000	0.016	0.025	0.033	0.000	0.000	0.000
1976	0.000	0.000	0.000	0.017	0.030	0.043	0.000	0.000	0.000
1977	0.000	0.000	0.000	0.021	0.036	0.050	0.000	0.000	0.000
1978	0.000	0.000	0.000	0.024	0.041	0.057	0.000	0.000	0.000
1979	0.000	0.000	0.000	0.027	0.046	0.064	0.000	0.000	0.000
1980	0.000	0.000	0.000	0.030	0.051	0.071	0.000	0.000	0.000
1981	0.000	0.000	0.000	0.040	0.045	0.050	0.000	0.000	0.000
1982	0.000	0.000	0.000	0.047	0.050	0.052	0.000	0.000	0.000
1983	0.000	0.000	0.000	0.054	0.054	0.054	0.000	0.000	0.000
1984	0.042	0.053	0.064	0.059	0.060	0.061	0.000	0.000	0.000
1985	0.042	0.053	0.064	0.064	0.066	0.068	0.000	0.000	0.000
1986	0.042	0.053	0.064	0.064	0.066	0.068	0.000	0.000	0.000
1987	0.042	0.053	0.064	0.069	0.072	0.075	0.000	0.000	0.000
1988	0.042	0.053	0.064	0.073	0.077	0.082	0.000	0.000	0.000
1989	0.042	0.053	0.064	0.078	0.083	0.089	0.000	0.000	0.000
1990	0.042	0.053	0.064	0.082	0.089	0.096	0.000	0.000	0.000
1991	0.042	0.053	0.064	0.082	0.089	0.096	0.000	0.000	0.000
1992	0.042	0.053	0.064	0.087	0.095	0.103	0.000	0.000	0.000
1993	0.042	0.053	0.064	0.091	0.100	0.109	0.000	0.000	0.000
1994	0.031	0.041	0.050	0.089	0.089	0.089	0.000	0.000	0.000
1995	0.031	0.041	0.050	0.092	0.092	0.092	0.000	0.000	0.000
1996	0.020	0.041	0.036	0.095	0.095	0.095	0.000	0.000	0.000
1997	0.020	0.041	0.036	0.095	0.095	0.095	0.000	0.000	0.000
1998	0.020	0.035	0.036	0.083	0.091	0.099	0.000	0.000	0.000
1999	0.020	0.035	0.036	0.083	0.091	0.099	0.000	0.000	0.000

2000	0.020	0.028	0.036	0.070	0.086	0.102	0.007	0.007	0.007
2001	0.020	0.028	0.036	0.070	0.086	0.102	0.007	0.007	0.007
2002	0.020	0.028	0.036	0.070	0.086	0.102	0.007	0.007	0.007
2003	0.020	0.028	0.036	0.070	0.086	0.102	0.007	0.007	0.007
2004	0.050	0.060	0.069	0.087	0.096	0.104	0.008	0.010	0.011
2005	0.050	0.060	0.069	0.087	0.096	0.104	0.008	0.010	0.011
2006	0.036	0.036	0.036	0.102	0.102	0.102	0.007	0.007	0.007
2007	0.047	0.047	0.047	0.100	0.118	0.135	0.012	0.017	0.022
2008	0.047	0.047	0.047	0.100	0.118	0.135	0.012	0.017	0.022

Notes:
1. Organic: Food and garden waste
2. P&C: Paper and card
3. Comb: Miscellaneous combustibles
4. Contain: Metal containers (tins and the like)

Appendix 5.4 – Estimated Non-combustibles and unclassified materials flowing to landfill.

Year	MSW Generated	Non-combustibles and unclassified MSW (%)	Total to landfill	Year	MSW Generated	Non-combustibles and unclassified MSW (%)	Total to landfill (tonnes)
1945	9,831,354	6.80	668,532	1977	14,945,020	7.02	1,049,140
1946	9,831,354	6.80	668,532	1978	15,595,117	7.02	1,094,777
1947	9,913,082	6.80	674,090	1979	15,642,153	7.02	1,098,079
1948	10,019,317	6.80	681,314	1980	15,698,474	2.68	420,719
1949	10,061,911	5.00	503,096	1981	16,204,617	2.68	434,284
1950	10,138,904	5.80	588,056	1982	16,181,613	2.68	433,667
1951	11,068,330	5.95	658,566	1983	16,194,205	2.68	434,005
1952	11,103,137	6.09	676,181	1984	16,729,196	2.68	448,342
1953	12,133,736	6.09	738,945	1985	16,780,904	2.68	449,728
1954	12,182,688	6.24	760,200	1986	17,989,692	2.68	482,124
1955	12,230,707	6.38	780,319	1987	19,198,286	2.68	514,514
1956	12,218,936	5.85	714,808	1988	20,015,529	2.68	536,416
1957	12,439,874	5.48	681,705	1989	20,847,175	4.90	1,021,512
1958	12,451,701	5.48	682,353	1990	21,686,110	4.90	1,062,619
1959	12,694,317	5.11	648,680	1991	22,543,691	4.90	1,104,641
1960	12,809,036	6.02	771,104	1992	22,233,706	5.45	1,211,737
1961	12,930,249	6.02	778,401	1993	21,913,315	5.45	1,194,276
1962	13,185,762	6.02	793,783	1994	21,601,270	7.70	1,663,298
1963	13,276,718	6.02	799,258	1995	21,300,000	7.70	1,640,100
1964	13,850,070	5.90	817,154	1996	20,812,171	7.70	1,602,537
1965	13,425,877	5.98	802,867	1997	22,550,000	7.70	1,736,350
1966	13,761,196	5.98	822,920	1998	22,549,000	7.70	1,736,273
1967	13,135,855	5.98	785,524	1999	22,979,759	7.70	1,769,441
1968	13,204,069	5.98	789,603	2000	23,531,000	7.70	1,811,887
1969	13,436,987	5.98	803,532	2001	24,760,000	2.56	633,856
1970	14,244,165	7.02	999,940	2002	25,079,000	2.56	642,022
1971	15,088,267	7.02	1,059,196	2003	24,150,695	2.56	618,258
1972	15,143,895	7.02	1,063,101	2004	25,448,000	2.56	651,469
1973	15,674,896	7.02	1,100,378	2005	25,658,000	2.56	656,845
1974	16,167,547	7.02	1,134,962	2006	25,453,616	2.56	651,613
1975	16,168,007	7.02	1,134,994	2007	25,771,081	2.56	659,740
1976	15,230,518	7.02	1,069,182	**Total material to landfill (tonnes**			55,115,545

Appendix 5.5 – Comparison of model outputs for soils/fines and value materials presented in Tables 5.6, 5.8 and 5.9 to the results published by van Vossen and Prent (2011) from their study of 60 landfill mining projects. Model output refers solely to that produced by proposed system dynamics model. The data for Austria and the REMO site were published after 2011.

% Composition by weight[1]	PPC (%)	Soils / fines (%)	Glass (%)	Metals (%)	Textiles (%)	Plastic (%)
Van Vossen and Prent (2011) mean values	5.3	57.3	1.1	2.0	1.6	4.6
Van Vossen and Prent (2011) range	4.84 -5.76	48.8 - 60.8	1.07 -1.13	1.93 - 2.07	1.56 -1.64	4.2 - 5.0
1960s model output for UK MSW (from boxplots 40% cover)	10.1 - 13.4	70.0 - 80.0	4.8 - 5.4	2.1 - 4.4	1.4 - 1.8	-
Table 5.6 1980s model output for UK MSW	19.2 - 22.2	58.1 - 64.3	5.2 – 6.5	4.6 – 5.1	1.8 - 2.5	2.2 - 3.1
Table 5.6 1990s model output for UK MSW	14.5 – 18.2	63.4 - 69.0	3.73 – 4.0	3.38 – 3.76	1.82 – 1.89	5.47 – 6.03
Table 5.8 model output: using Austrian MSW composition data	7.84	51.1	1.81	1.81	7.24	5.43
Table 5.8 model output: using calculated Austrian MSW data	5.32	49.8	1.18	1.77	5.32	4.14
Table 5.9 model output: using Belgium MSW & 40% cover	8.24	53.1	1.41	1.77	1.15	9.78
Table 5.9 model output: using Belgium MSW & 30% cover	8.66	50.7	1.48	1.86	1.21	10.28

Note: 1 this table does not include wood, leather, construction and demolition waste, or other wastes included by van Vossen and Prent.