APPENDIX FOUR a

ECONOMIC MODELLING FOR THE MAYOR'S MUNICIPAL WASTE MANAGEMENT STRATEGY

Economic Modelling for the Mayor's Municipal Waste Management Strategy

The Greater London Authority: PN497

APPENDICES

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Contents

A.1.0 Structure of Economic and Environmental Model

An Excel® based model was developed to provide analysis of the following elements of waste management within London:

- Performance and costs of *dry* recycling collection systems;
- Performance and costs of *biowaste* collections systems;
- Carbon dioxide (CO_2) emissions or benefits associated with recycling;
- CO_2 emissions associated with biowaste and residual treatment processes; and
- Financial costs of biowaste and residual treatments.

A 'Do Nothing New' baseline was constructed, followed by modelling of 11 scenarios. The net change in outputs was then measured against this baseline to show which scenarios provided the greatest benefits.

Section A.3.1 below describes the development of the 'Do Nothing New' baseline. Section A.3.2 then describes the approach to modelling scenarios, and calculating the outputs required for the study.

A.2.0 Assumed Waste Compositions

A.2.1 Current Waste Arisings

Current waste arisings were taken from the latest relevant Environment Agency WasteDataFlow (WDF) reports (2008/09) for each London Borough. We believe these to be the best representation of arisings currently available.

A.2.2 Changes in Arisings over Time

Modelling forward to 2031, changes in waste arisings are based on a recent study undertaken on behalf of the GLA.[1] The study seeks to understand what impacts various factors, such as waste prevention and population growth, will have on the total waste generated in London.

Sections A.2.2.1 and A.2.2.2 describe the approach taken for some of the key streams, household and commercial wastes, included in the model. These growth rates are included in both our 'Do Nothing New' baseline, and in each 'Do Something' scenario.

A.2.2.1 Household Waste

It should first be acknowledged that waste arisings are difficult to predict 20 years into the future, as is required for this study. The main determinants of waste growth have been historically attributed to household growth and consumption. Communication campaigns to raise awareness and increased service provision are expected to reduce the growth from households. Household growth has most recently been predicted by Defra at a level of 0.5% per annum.[2]

We use the approach set out in the SLR study in the modelling of household waste arisings. The key features of this study can be summarized as follows:

- The number of households in London will continue to increase, with projections indicating a 17% increase by 2031;
- Household waste arisings, per household, will decrease over time, as a result of waste prevention effects considered to arise through communication campaigns, and additional service provisions;
- The combination of <u>increased</u> housing and <u>decreased</u> waste, per household, means that the absolute growth in household waste will be <u>zero</u>; and
- All elements of 'household waste', for example, RRCs, street sweepings etc, will also be given a zero growth rate in the model.

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[1] LRS / SLR (2010) Future Waste Arisings in London, 2009 - 2031: Project summary and methodological memo., 2010

[2] DEFRA (2007) *Waste Factsheet: National Waste Targets for England,* available at: http://www.defra.gov.uk/Environment/waste/strategy/factsheets/targets.htm

A.2.2.2 Commercial Wastes

There are many different factors to consider when forecasting changes to future arisings of commercial wastes. Such consideration is complex and falls outside the scope of this study. We have therefore assumed that arisings from commercial waste will match those forecast in the aforementioned SLR study undertaken on behalf of the GL. Overall this study shows that non-household municipal waste will grow by 30% by 2031.

A.2.3 Waste Compositions

A.2.3.1 Kerbside-collected Household Wastes

The kerbside composition used for this study was taken from the recent review of municipal waste component analysis for England, published in early 2010.[3] This was then benchmarked against London studies such as the North London Waste Authority (NLWA) waste compositional analysis.[4] The composition used for our model was altered to reflect the generally lower than average garden waste arisings in London. This was thought to represent the average London composition better than by using the UK wide study 'as is'. Furthermore, specific regional studies in London were not thought to represent the average for the whole area.

It should be noted that this dataset is not considered to be a critical element of the study. It was used only to benchmark future collection systems yields against likely capture rates.

A.2.3.2 Commercial Wastes

Data on the composition of commercial wastes is relatively scarce. Our model uses the most recent survey of commercial wastes published by the Environment Agency (Wales) in 2009 as the basis for this composition.[5] The data produced by the survey suggests that around 50% of the stream is considered to be 'mixed wastes'. We have used data from the previous analysis of commercial wastes provided by AEA Technology for the Welsh Assembly Government (WAG) as the basis for modelling the composition of this non-differentiated "mixed wastes" fraction.[6]

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[3] Resource Futures (2010) *Municipal Waste Composition: Review of Municipal Waste Component Analyses - WR0119*, Final Report for Defra

[4] Entec (2009) Waste Composition Analysis Project for NLWA, Interim Report for North London Waste Authority. See *https://ukr.hybis.info/Projects/WX/Awarded/WX64532/Issued1/Shared%20Documents/Composition%20studies/NLWA%20waste%20composition%20draft%20report.pdf*

[5] Environment Agency Wales (2009) Survey of Industrial & Commercial Waste Arisings, May 2009

[6] AEA (2003) The Composition of Municipal Solid Waste in Wales: A Report Produced for the Welsh Assembly Government, December 2003

A.2.3.3 Residual Composition

We have assumed that residual waste composition varies depending on the extent of recycling occurring. As a result, we have therefore developed separate compositions for a 'low' recycling scenario, two 'medium' recycling scenarios (one with collection focused capture of dry materials, the other with collection focused on capture of food waste) and a 'high' recycling scenario.

A.3.0 Scenario Modelling Assumptions

This section describes the underlying assumptions and data used to develop and model all 'scenarios'. First a 'Do Nothing New' baseline scenario is described in Section A.3.1. This was, in essence, the baseline used to measure the relative effects of all the 'Do Something' scenarios described in Section A.3.2, whereby some change in waste management practice occurs.

A.3.1 'Do Nothing New' Baseline

The 'Do Nothing' baseline contains the following elements, for each Borough:

- Kerbside waste arisings and recycling tonnages for 08/09, taken from WasteDataFlow (WDF);
- Kerbside dry recycling, residual and organic waste system descriptions for both doorstep and communal properties; and
- Waste arisings and recycling tonnages from RRCs, bring sites, commercial enterprises and 'other' tonnages, where 'other' refers to systems such as 'on-the-go' recycling and voluntary or third sector organisation (TSO) schemes.

A.3.1.1 Reuse

Foremost, we have obtained reuse tonnages from WDF. In addition to this, voluntary and TSO reuse schemes are in operation, which do not report to the London Boroughs and therefore the associated reuse tonnage is not included in WDF. Additional reuse of 7,000 tonnes was therefore added to the baseline tonnage to reflect the estimated 10,000 tonnes reused in 2008/09.[7] The cost associated with reuse is detailed in Appendix A.6.4.

A.3.1.2 Household Kerbside Modelling

Alongside the use of in-house data sources, information relating to the types of collection systems in place, in each Borough, originated primarily from WasteDataFlow (WDF), a recent report on behalf of the GLA on best performing recycling schemes in London.[8] Each collection system is assigned a specific code. It is important to note that 'communal' and 'doorstep' properties have been treated separately, as the costs and performance will be very different. <u>Doorstep</u> properties are defined as those receiving a collection service where the collection crew comes to the householder's door, to collect recyclables or biowaste. <u>Communal</u> properties are therefore defined as properties where the householder has to bring the material to a

[7] Mayor of London (2010) *The Mayor's Draft Municipal Waste Management Strategy,* available at: http://www.london.gov.uk/mayor/environment/waste/docs/draft-mun-waste-strategy-jan2010.pdf

[8] Hyder Consulting (2010) The Performance of London's Municipal Recycling Collection Services, Final Report for GLA, March 2010

central 'communal' point. The collection crew goes to this location to tip waste in the vehicle.

The number of communal properties was extracted from the household figures, as supplied by the aforementioned study undertaken by Hyder. Where these were unavailable, we have used the Office of National Statistics (ONS) data to ascertain the number of 'hard to reach' properties on communal collection systems.[9]

The tonnages of recycling and refuse collected from household (at the doorstep) are taken directly from WDF. As WDF does not split the recycling, however, doorstep and communal recycling is therefore aggregated in our baseline.

As noted above, all households in London were assigned a specific code relating to the types of recycling schemes with which they are serviced. These codes are based upon the type and number of materials collected, the frequency of collection, and the frequency of refuse collection. For each of the codes, collection costs per household, were taken from publicly available data (see Section A.1.0). Total collection costs are thus derived from these 'per household' costs and the number of households on each scheme.

Cost data is readily available for doorstep properties, and considered to provide reasonable estimates of service costs. Data on communal collection systems is far less available. As such, assumptions are required to estimate the average costs of communal based services across London. It should be acknowledged, therefore, that there are significant uncertainties associated with these estimations, as discussed in Appendix A.1.1.1.

A.3.1.3 Other Household Wastes

Tonnages of waste managed by the following routes, were also extracted from WDF:

- Recycling:
  - Reuse and Recycling Centres (RRCs);
  - Bring Banks;
  - 'On-the-go' recycling; and
  - Non-contracted / voluntary services.
- Refuse:
  - Regular household collection;
  - Street sweepings;
  - Bulky waste;

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[9] The most recent ONS data refers to 2001. We have estimated relative percentages of household type to the total household numbers and adjusted to reflect increase in intermediate years. Using information from WRAP regarding household collection systems we have estimated the number of flats receiving doorstep and communal collection. We have assumed all detached and semi-detached households receive a doorstep collection.

- RRCs;
- Healthcare wastes;
- Fly-tipped wastes; and
- Other wastes collected for disposal.

The baseline costs of managing these wastes are set out in Section A.1.0

A.3.1.4 Commercial Waste

Tonnage data for commercial waste recycling, and refuse collection was extracted from WDF. Costs, on a per tonne basis, were calculated using a model designed to estimate likely collection charges seen by commercial enterprises today. More information on this collection cost model is given in Section A.6.7 below.

A.3.2 'Do Something' Scenarios

The scenario modelling comprised of two main elements. These were a) collection system scenarios and b) residual waste treatment scenarios. The approach to this modelling is given in the Main Report (Sections 4.0 and 5.0). The outcomes, in terms of changes in waste management, are also presented here. This Section also gives further detail on the performance of the different collection systems, over time, for each scenario.

Table 1 shows the assumed 'switches' from the current baseline to these best performing systems. The order of switches has been made according to cost, i.e. with the lower cost switches being made first. These switches have been modelled across London as a whole, and it should be emphasised that there is no implied preference as to which Boroughs should change their collection systems first.

Table 1: Switches to 'Best Practice' Collection Schemes

Waste Stream	System / Material	'Doorstep' Households	'Communal' Households
Dry Recycling	Twin Stream	Fortnightly, all materials	Weekly co-mingled
	Co-mingled	Weekly collection, all materials	All fortnightly switch to weekly
	Source Separated	Weekly collection, all materials	All fortnightly switch to weekly
Organic Wastes	Garden	As per baseline system	As per baseline system
	Food	As per baseline system	Weekly source separate
Refuse	Residual	Fortnightly collection, 240 litre wheeled bin or Weekly sack based where appropriate	As per baseline

A.3.2.1 Household Dry Recycling Rationale

Dry recycling performance for each collection system published by WRAP has been used as a basis for our modelling.[10] The performance of the recycling schemes is also a key factor in achieving future recycling targets. Current performance of the recycling services in London (both inner and outer) is lower than the national average.[11] London will not meet its targets if the average performance of current systems stays constant. The approach to modelling increasing performance is given in Sections 3.2.1.1 and 3.2.1.2.

[10] WRAP (2009) *Analysis of kerbside dry recycling performance in England 2007/08,* available at: http://www.wrap.org.uk/local_authorities/research_guidance/collections_recycling/benchmarking.html

[11] ibid

3.2.1.1 Doorstep Properties

The approach to modelling 'doorstep' dry collection systems is as follows:

- The best practice 'doorstep' type collection systems for dry materials are given in Table 2. In essence, all co-mingled, twin stream or kerbside sort systems remain as such, but changes are made in terms of what materials are collected, and the frequency of collection of both the recyclables and refuse. Both co-mingled and twin stream systems, whereby the materials are collected in a 120 or 240L bin, are collected fortnightly, whilst where the materials are collected in a sack, collection frequency becomes weekly (if not already). For 'kerbside sort' schemes, which utilise kerbside boxes, the frequency of collection becomes weekly; and
- Yields from recycling systems are modelled to increase to *current* maximum levels by 2015 (see Table 2). This approach might be considered optimistic and will depend upon behavioural change, but alternatively, to model meeting the 45% target, significant additional tonnage would need to be assumed as captured from non-kerbside schemes, which would appear even more challenging.

Table 2: Dry Recycling Yields modelled for Best Practice 'Doorstep' Properties in 2015 (kg/hhld/annum)

System Type	Refuse frequency	Recycling Frequency	Materials Collected	Paper & card	Cans	Glass	Plastic	Total Yield
Twin Stream	Fortnightly	Fortnightly	Paper/Card, Glass, Tins, All Plastics	184	15	70	15	284
Single Stream Comingled (Bin)	Fortnightly	Fortnightly	Paper/Card, Glass, Tins, All Plastics	207	11	70	15	303
Single Stream Comingled (Sack)	Fortnightly	Weekly	Paper/Card, Glass, Tins, All Plastics	200	23	67	23	313
Kerbside Sort (Box)	Fortnightly	Weekly	Paper/Card, Glass, Tins, Plastic bottles	179	13	65	18	275

Note: Additional capture of Textiles and WEEE was modelled at 1.5 and 0.15 kg/hhld/annum respectively in 2015

It should also be noted that only the most significant materials, in terms of arisings, were modelled in this study. Materials such as aerosols and tin foil were excluded to simplify the modelling. It is recognised that this is a limitation. However, the yields of such materials are insignificant compared to the total quantity of key materials captured, and thus do not have a key impact on the achievement of recycling targets, within the error bounds of our approach to modelling.

- In future years, yields from the different collection systems are increased on a material-specific basis. This includes additional capture of materials such as Textiles and WEEE;
- The average capture rates from the best performing systems (described above) are as follows:
    - 2015 – 253 kg/hhld/annum;
    - 2020 – 270 kg/hhld/annum; and
    - 2031 – 313 kg/hhld/annum.
- For the 'max GHG abatement' scenario (see Section 4.0 of Main Report) the capture rates are increased to higher levels.

3.2.1.2 Communal Properties

The current recycling performance per household varies from Borough to Borough. In future years, estimates of the performance of recycling schemes from 'hard to reach' properties, such as flats, is required. However, the data on communal properties is currently very limited. There have been very few studies carried out on the performance, and even less so on the costs, of different systems. Waste Watch, for example, note that barriers to the collection of performance data from communal properties include lack of funding and lack of consistent approach to data collection.[12] Capture rates per household have, however, been published by WRAP.[13] This data has been used in our model and cross checked with internal data on such capture rates.

The approach to modelling capture rates can be summarised as follows:

- The 'average' and 'high' capture rates are given in Table 3 and are based on the aforementioned WRAP data;
- All dry recyclable collection systems from communal based properties were only assumed to achieve current 'average' yields by 2015;
- By 2020 all systems are assumed to increase yields to 80% of current 'high' levels;

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[12] Waste Watch (2006) *Recycling for Flats,* Available at: http://www.wastewatch.org.uk/Homepage

[13] WRAP (2009) *Recycling Collections for Flats,* Available at: http://www.wrap.org.uk/local_authorities/research_guidance/collections_recycling/recycling_collections_for_flats/operation_of_different_collection_schemes/door_to_door.html

- By 2031 all systems are assumed to have increased yields to levels equal to current 'high' levels; and
- Collection frequency remains the same. This is because additional collections provide no direct incentive to the householder to recycle more, unless capture rates increase significantly in flats where there is only a small area to accommodate materials.

This approach was considered appropriate given the uncertainties and likely barriers to recycling from 'hard to reach' communal properties in London.

Table 3: Dry Recycling Yields from Communal Properties (kg/hhld/annum)

System Type		Collection Frequency	'High' Yield	'Average' Yield
Bring	Co-mingled	Weekly	117	85
	Co-mingled	Fortnightly	93	68
	Twin Stream	Weekly	178	129
	Source Separated	Weekly	156	113
Floor	Co-mingled	Weekly	0	131

A.3.2.2 Household Organic Wastes Rationale

For organic wastes, modelling of the captures of materials was based upon work undertaken by Eunomia and published by WRAP in 2008 and 2009. The yields of material from the UK's best performing garden waste and kitchen waste collection systems are given in Table 4 and Table 5. It should be noted that these relate to 'doorstep' type properties only, as similar data is not available for 'communal' properties due to a lack of schemes currently in operation in the UK. For this study, therefore, we have used a figure of around 50% of that potentially achievable for doorstep properties, i.e. 50 kg/hhld/annum.

Table 4: Best Practice Yields from Garden Waste Collection Systems

Charging Basis	Frequency	Container	Yield (kg/hhld/annum)
Charged	Monthly	Renewable Sack	22.5
	Monthly	Non-Renewable Sack	22.5
	Fortnightly	Renewable Sack	45
	Fortnightly	Non-Renewable Sack	45
	Fortnightly	240L Bin	87.5
	Weekly	Renewable Sack	81
	Weekly	Non-Renewable Sack	81
Free	Fortnightly	Renewable Sack	130
	Fortnightly	Non-Renewable Sack	130
	Fortnightly	180L Bin	200
	Weekly	Renewable Sack	150
	Weekly	Non-Renewable Sack	150

Table 5: Best Practice Yields from Kitchen / Garden Waste Collection Systems

System Type	Collection Frequency	Garden (kgs/hhld/annum)	Kitchen (kg/hhld/annum)
Source Separated Kitchen Waste	Weekly	n/a	100
Co-mingled Kitchen and Garden Waste	Fortnightly	200	20
	Weekly	220	70

When the yields in Table 4 and Table 5 were applied to the current collection systems in place in London, the estimated overall tonnage was significantly higher than that reported by WDF. This is a factor of both a) the demographics within London, b) the lower garden waste proportion in household kerbside waste compared to the national average and c) the high proportion of flats in the housing stock.

Garden waste yields in London reported in WDF are equal to around 40% of those reported in the Table 4 and Table 5. This is again due to the lower levels of waste generated by a large proportion of flats in London compared to houses, and the lower associated performance of the schemes in place.

The increase in future performance modelled for the existing and additional systems put in place, can be summarised as follows:

- 'Doorstep' yields of garden waste were increased from the current 40%, to 60% of the figures given in Table 4, by 2015;
- These were further increased to 65% and 70% in 2020 and 2031 respectively;
- The proportion of garden waste from 'communal' properties remains constant until 2031 when it was increased from 60% to 70%;
- Yields of food waste from doorstep properties remains at 100 kg/hhld/annum in 2015, and increases to 110 and 120 kg/hhld/annum in 2020 and 2031 respectively; and
- Yields of food waste from communal properties remain at 50 kg/hhld/annum in 2015, and increase to 55 and 60 kg/hhld/annum in 2020 and 2031 respectively.

It should be noted that when food waste collection systems are rolled-out, a waste prevention effect can be observed.[14] In essence, this is because when a food waste collection is implemented the householder considers the quantity of waste being discarded and will often seek to reduce the quantity they are producing. This is unknown beforehand as all the material is placed in the refuse. In this study we model a 30 kg/hhld/yr reduction in waste arising as a result of the introduction of a new source separated food waste collection system.

A.3.2.3 Household Residual Waste Rationale

In terms of the collection of residual waste the key factors influencing the modelling are a) the frequency of collection and b) the type of container. The greatest reduction in arisings (and thus increase in recycling yields) takes place when new recycling schemes are implemented alongside a change in refuse collection frequency from weekly to fortnightly. This consideration is only relevant to 'doorstep' properties, as the same incentive to reduce residual waste does not exist with communal bins.

The only limiting factor to this approach is the location of dwellings and the type of container system in place. Figure 1 below shows that nearly 1.5 million households in London receive a weekly, sack based collection. Many of these properties may be in locations whereby the use of 120 or 240L bins is not feasible. Therefore switching to a fortnightly collection may not be a desired option. Furthermore, there will be an additional capital cost in switching from sack to bin type collections.

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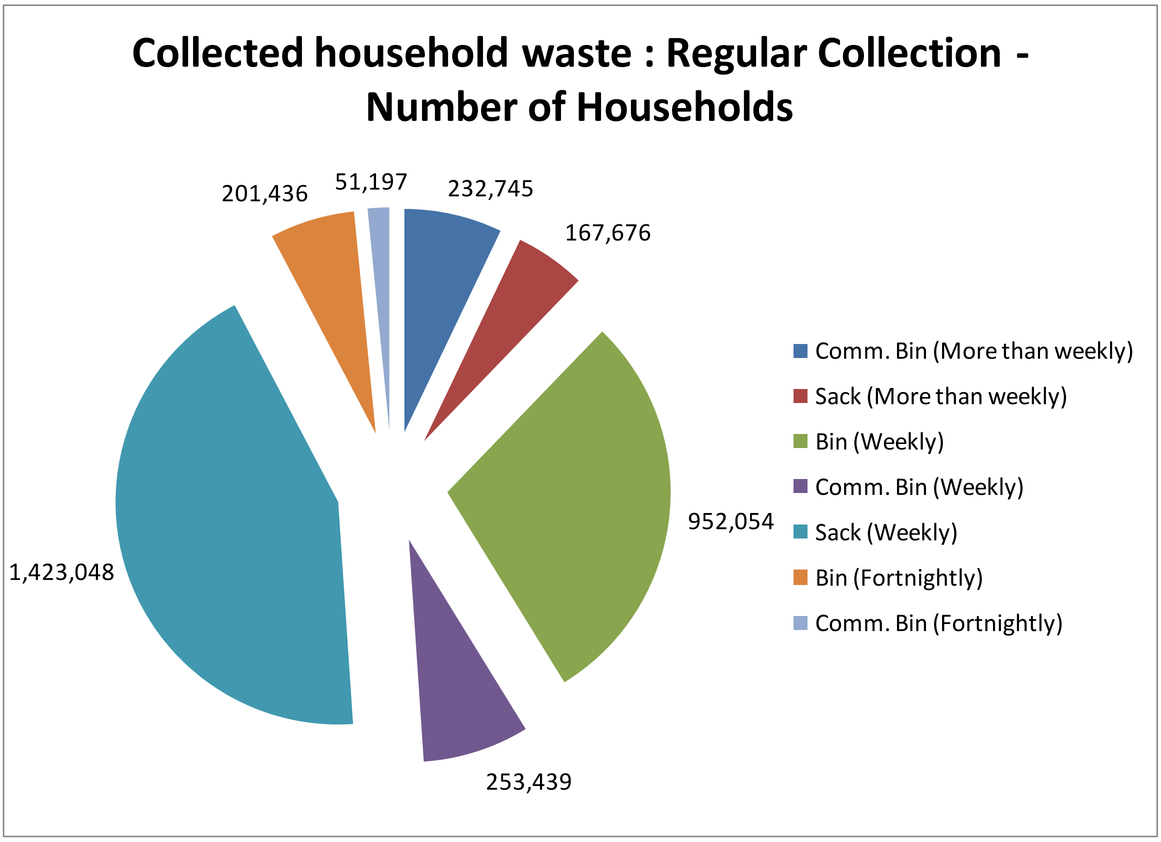
[14] Eunomia et al (2010) Assessment of the Options to Improve the Management of Bio-waste in the European Union, Final Report to DG Environment, February 2010

Understanding the proportion of households in London limited by this factor is not possible within the scope of this study. We therefore use a simplistic approach of assuming only 50% of the 1.5 million households in London can be switched from weekly sack to fortnightly bin collection systems.

Figure 1: Refuse Collection Frequency and Container Type for London Households



Source: Adapted from information provided by WasteDataFlow (2008/09)

A.3.2.4 Bring Banks

Currently, there is around 55ktpa of waste recovered for reuse and recycling from bring banks in London. The most commonly collected materials are:

- Glass;
- Paper / Card (including books);
- Metals Cans;
- Plastic bottles; and
- Textiles.

We assume that, over time, there will be an increase in the number of bring banks in London. In the modelling, therefore, we increase the capture of materials, on an annual basis, by an additional percentage point above 2008/09 levels. *Additional* captures can therefore be summarised as follows:

- 5% by 2015;

- 10% by 2020; and
- 20% by 2031.

It should be noted that as a result of these increased captures from bring banks, we reduce in the model the tonnage of refuse collected from households.

A.3.2.5 On-the-Go Recycling

On-the-Go recycling has been targeted as a potential method to increase recycling rates and enhance awareness of recycling in London. Levels of performance of current schemes, however, are currently difficult to assess. Few direct comparisons can be made between systems due to variation in duration of the scheme, numbers and size of bins and materials collected. In addition, bins are often collected on existing household or commercial waste collection rounds and thus tonnage data is usually recorded as such.

With such schemes, there are three main approaches in terms of materials collected:

- Paper only (mainly in stations and areas where free papers are distributed);
- Co-mingled collection of paper, plastic bottles, cans and glass; and
- Separate collection of paper, plastic bottles, cans and glass.

Schemes that are in place across the UK have frequently been designed alongside the current household recycling service and therefore vary across authorities. For example, in Poole Borough Council, a twin-bin household recycling service is in place, with blue bins for co-mingled recycling and black bins for residual waste. The on-street recycling bins have followed this design. Personal communications with officers from Poole Borough Council suggest that this matching of street and household recycling bins has led to increased performance. Similarly DEFRA guidance and a London Assembly report suggest that a standard approach to branding of street recycling bins is necessary to increase performance.[15] [16]

Table 6 summarises a number of schemes in place across the UK. In order to examine the performance of on-the-go recycling, data was extracted from WasteDataFlow (WDF) where available, whilst several UK authorities with established schemes were contacted directly by Eunomia. Few conclusions can be drawn, however, from this small sample set as the design of the schemes differs and little accurate tonnage data exists.

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[15] DEFRA (2008) *Recycle Bins in Public Places 'Recycle on the Go': A Good Practice Guide,* available at:
http://www.defra.gov.uk/environment/waste/localauth/recycleonthego/documents/recycleonthego-guide.pdf

[16] London Assembly (2009) *'On the Go' Recycling,* Report to Environment Committee May 2009

Table 6: Summary of Known On-the-Go Recycling Schemes

Authority	Bin Description	Reported capture (tonnes)	No. of Bins	Duration of scheme
Blackpool	Co-mingled collection of cans, glass and plastic bottles	144	159	Apr-06
City of London	Paper	79	36	Jan-08
Hillingdon	Co-mingled; glass, paper, cans, and plastic	90	45	Jun-08
Hinckley and Bosworth BC	Separate collection of glass, cans, paper and plastic bottles	1.4	20	Jul-08
North Tyneside Council	Separate collection of glass, cans, paper and plastic bottles	144	50	Apr-05
Poole	Co-mingled collection of glass, cans, paper and plastic bottles	93	300	Jun-08
Southwark	Paper	103	31	Jun-05

Based upon the data provided by authorities across the UK in Table 6, we have taken average yields for the three main types of on street recycling bins. These are given in Table 7.

Table 7: On-the-go Recycling Performance Assumptions for London

Material	tpa/per bin
Co-mingled collection of glass, cans, paper and plastic bottles	1
Separate collection of glass, cans, paper and plastic bottles	0.5
Paper	3

For modelling purposes we have assumed that co-mingled bins, using the ‘Recycle Now’ iconography, will be put in place across London Boroughs. We have assumed that the number of on-street recycling bins will increase in number over time, which will result in additional capture of materials. The number of bins and related tonnages modelled for each year are given in Table 8.

Table 8: Number of On-the-Go Recycling Bins and related Materials Captures

Year	Number of Co-mingled Bins per Authority	Tonnes of Recyclable Material Collected
2015	30	990
2020	100	3,330
2031	200	6,660

A.3.2.6 Reuse and Recycling Centres (RRCs)

Recycling from RRCs is important towards ensuring London meets the targets set out in the Mayor's Draft MWMS. The current average recycling rate for all RRCs in London is around 46%. This is low compared to the national average because of the lower proportion of garden waste in London.

The overall recycling rate for RRCs in London is assumed to increase as shown in Table 9. The recycling rate modelled increases only marginally from 2015 to 2020. This is because very little further increase is required to meet the overall 50% recycling target (in 2020) as improved performance modelled for kerbside collection systems in the 5 year period is adequate to meet this higher target. Higher performance rates have been modelled for the 'max GHG abatement' scenario to reflect the need to increase recycling rates above those within the Mayor's Draft MWMS under this scenario.

Table 9: Recycling Rates Modelled for RRCs in London

Year	Central Assumptions	'Max GHG Abatement' scenario
2015	55%	60%
2020	56%	70%
2031	70%	80%

A.3.2.7 Commercial Wastes

In our model, assumed increases in commercial waste recycling play a significant role in meeting the recycling targets in the Draft MWMS. The achievement of these targets, therefore, is very sensitive to variation in rates of commercial waste recycling. This sensitivity is tested in Section 9.0 of the Main Report, whilst only the central assumptions are presented here.

Current recycling data for commercial waste recycling services provided by Boroughs (or their contractors) was gathered from WDF. To disaggregate the 'co-mingled' fraction reported in WDF the following assumptions were made, in relation to the composition of this fraction:

- Paper / card – 70%;
- Glass – 20%;
- Metals – 5%; and
- Plastics – 5%.

The generic composition modeled for commercial wastes is discussed above in Section A.2.3. Using the 2008/09 yields from WDF and this composition the captures of each material were modelled as shown in Table 10, along with the estimated captures by material for each of the future target years modelled.

Table 10: Current and Future Commercial Waste Recycling Performance, by Material,

Material	2008	2015	2020	2031
Paper / Card	60%	65%	75%	60%
Plastics	15%	25%	45%	15%
Textiles etc	5%	35%	60%	5%
Wood / Furniture	30%	40%	70%	30%
Glass	45%	50%	75%	45%
Metals	30%	50%	75%	30%
WEEE	20%	50%	65%	20%
Garden	50%	60%	75%	50%
Food	20%	35%	50%	20%
Hazardous	0%	0%	0%	0%
Misc Comb.	40%	60%	70%	40%
Misc Non-Comb.	50%	60%	80%	50%
Total	**4%**	**40%**	**50%**	**65%**

Given the large proportion of paper and card in the commercial waste stream, the recycling of this material will have a clear effect on the overall recycling rate attained in London. Using the composition modelled for this study it appears that the current capture of paper and card is very low (5%). National averages for paper and card recycling from the commercial waste stream, however, are estimated to be around

75%.[17] These low capture rates could be attributed to (a) the composition, (b) the number of businesses receiving a collection service or (c) the performance of the services currently in place. Given this far higher national average, however, it does not seem unfeasible that the recycling of paper and card could increase significantly in London. Furthermore, there appears no significant limitations to this change occurring within 5 years. Therefore, we have modelled a sharp increase to 60% paper / card recycling by 2015.

All other materials follow likely increases in capture rates based on internal information held by Eunomia. When the 'max GHG abatement' scenario is considered, these capture rates are increased in 2031 to best practice levels currently achieved in other EU Member States, and with a faster rate of increase than under the central case.

A.3.2.8 Voluntary / Non-Contract

There are a number of voluntary or non-contract kerbside and bring sites operating in London. As the current tonnage of waste collected by such means is small, this is not changed in future years within our model.

[17] Eunomia (2010) *Feasibility of Landfill Bans Research*, Report on behalf of WRAP, March 2010

A.4.0 Modelled Municipal Waste Flows in London

The waste flows in the model are complex and due to the number of scenarios and modelled years (11 x 4) there is a large amount of data to present. A flow diagram may, perhaps, be a more logical choice of presentation, but given that 44 diagrams would be required, and that it would be harder to compare year-by-year results. In this Appendix, we therefore present waste flows in table format. To aid understanding of the tables, however, in Figure 4 we have provided an example waste flow for Scenario 11 (Max GHG Abatement) and the year 2031.

As discussed in the Main Report, under the 'focus on dry 'and 'focus on food' scenarios, broadly the same amount of food and dry materials require collection to meet the 2015, 2020 and 2031 targets. The main difference between these two scenarios, therefore, is in the order of roll-out of services to 2015, which are shown in Figure 2 and Figure 3. These show that under all 'focus on dry' scenarios, dry recycling services are rolled out first, followed by food and green waste collections to reach the 45% recycling / composting target in 2015. The reverse is the case for the the 'focus on food' scenarios, whereby food waste services are rolled out first.

The abbreviations used along the x-axis in both Figure 2 and Figure 3 are set out in Table 11.

Table 11: Abbreviations for Collection Systems

Abbreviation	Collection System
DS Dry	Dry recyclables from the doorstep
Com Dry	Dry recyclables from 'communal' systems (i.e. for flats)
Garden	Garden waste from the doorstep
DS Food	Food waste from the doorstep
DS Food (Co.)	Food waste, comingled with green waste from the doorstep
Com Food	Food waste from 'communal' systems (i.e. for flats)

Figure 2: Roll out Scenarios 3 and 4 with 'Focus on Dry' (2008 – 2015)'

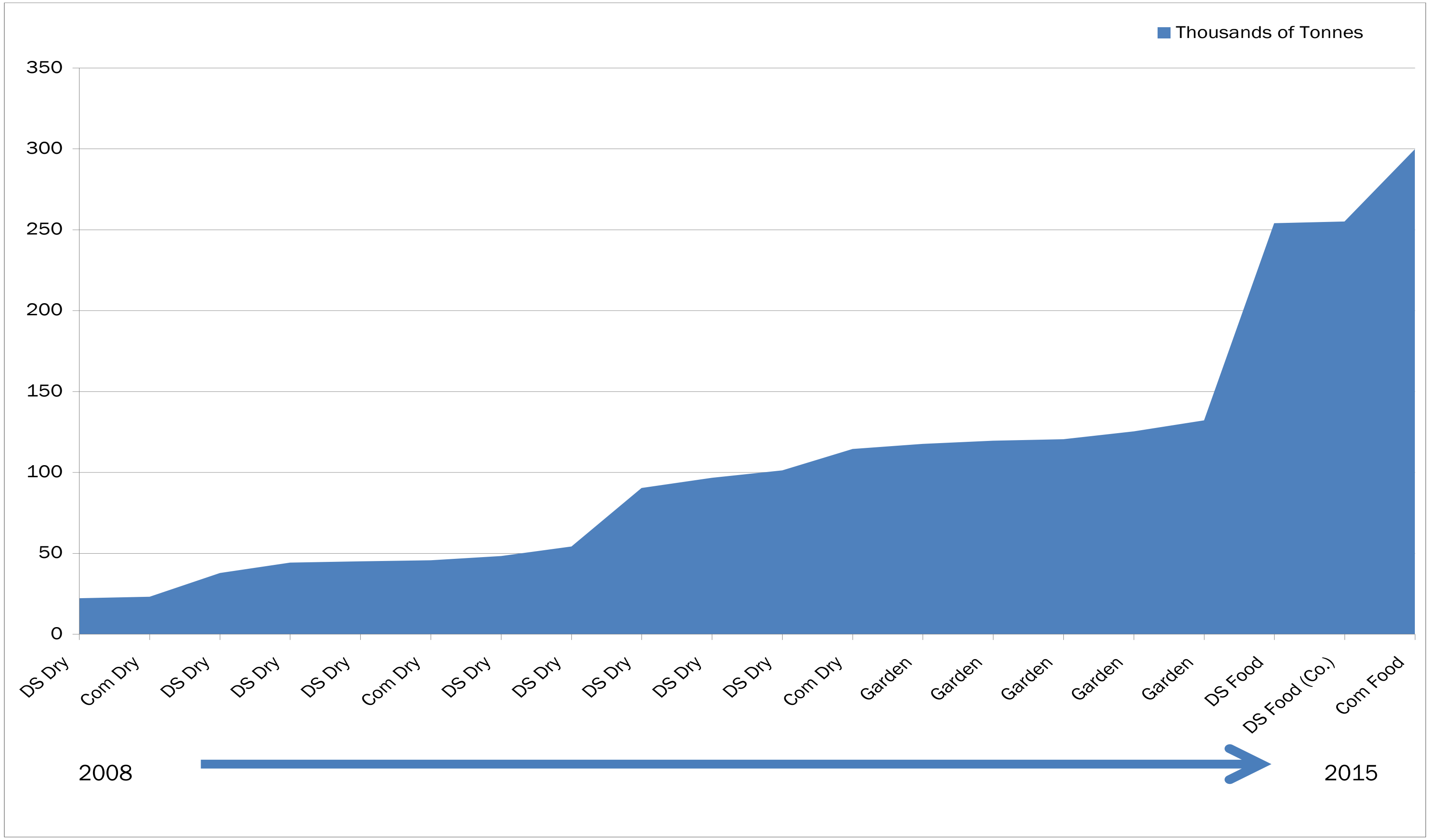

Figure 3: Roll out Scenarios 3 and 4 with 'Focus on Food' (2008 – 2015)'

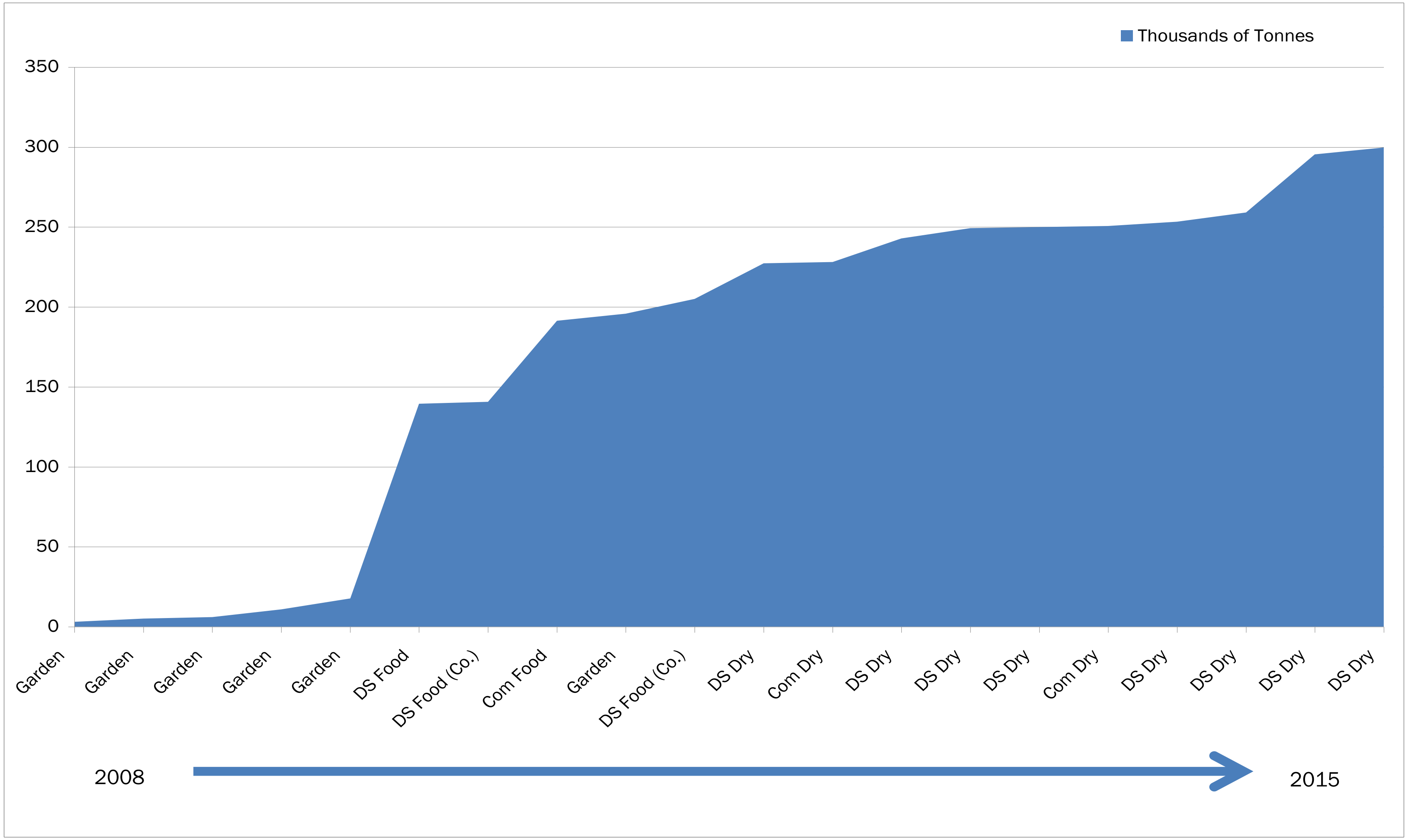

Figure 4: Example Municipal Waste Flows (Scenario 11 – 'Max GHG Abatement' - 2031)

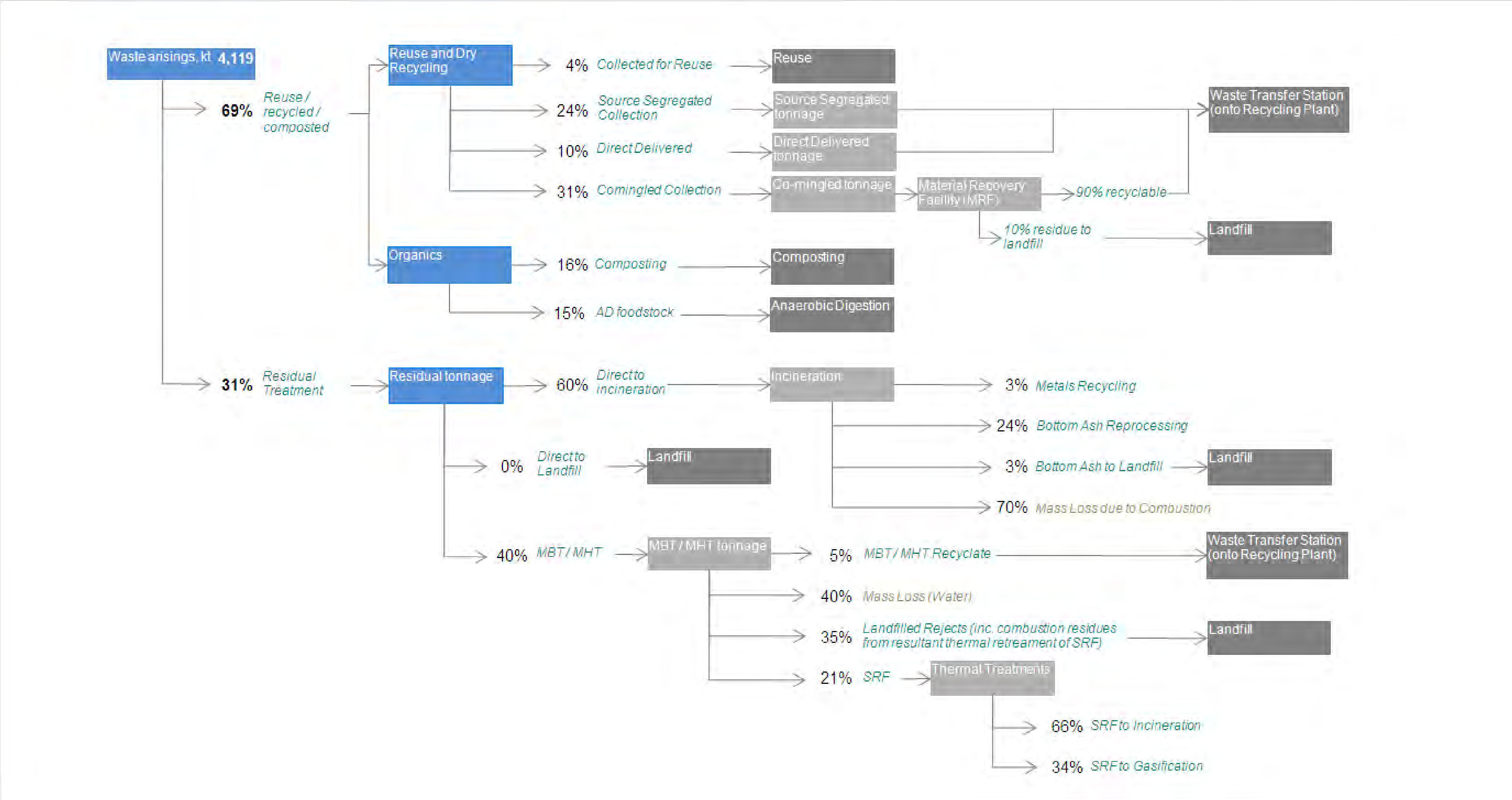

Table 12: Waste Flows for Scenario 1: 'Do Nothing'

Waste Collected for:			2008	2015	2020	2031
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	25%	26%	26%
	Dry Recyclables	Reuse	1%	1%	1%	1%
		Source Segregated Collections	22%	17%	18%	18%
		Comingled Collections (i.e. MRF Input[1])	31%	36%	37%	38%
		Direct Delivered	19%	19%	18%	18%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	27%	26%	25%
		Anaerobic Digestion (AD)	0%	0%	0%	0%
Residual Treatment	Total Residual Treatment		75%	75%	74%	74%
	Direct to Incineration	Input	28%	44%	43%	25%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	9%	9%	9%
		Recycling	0%	0%	0%	0%
		SRF	22%	22%	22%	22%
		% SRF to Incineration	100%	100%	100%	100%
		% SRF to Gasification	0%	0%	0%	0%
		Landfilled Rejects	36%	36%	36%	36%
		Mass Loss (Water)	41%	41%	41%	41%
	Landfill	Input - Direct to Landfill	62%	47%	48%	66%
		Input - Rejects from Sorting / Treatment[2]	4%	5%	5%	4%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 13: Waste Flows for Scenario 2: Residual to low biomass new techs only

Waste Collected for:			**2008**	**2015**	**2020**	**2031**
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	25%	26%	26%
	Dry Recyclables	Reuse	1%	1%	1%	1%
		Source Segregated Collections	22%	17%	18%	18%
		Comingled Collections (i.e. MRF Input[1])	31%	36%	37%	38%
		Direct Delivered	19%	19%	18%	18%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	27%	26%	25%
		Anaerobic Digestion (AD)	0%	0%	0%	0%
Residual Treatment	Total Residual Treatment		75%	75%	74%	74%
	Direct to Incineration	Input	28%	44%	43%	25%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	46%	52%	75%
		Recycling	0%	3%	3%	4%
		SRF	22%	28%	29%	29%
		% SRF to Incineration	100%	66%	48%	27%
		% SRF to Gasification	0%	34%	52%	73%
		Landfilled Rejects	36%	33%	33%	33%
		Mass Loss (Water)	41%	35%	35%	35%
	Landfill	Input - Direct to Landfill	62%	10%	5%	0%
		Input - Rejects from Sorting / Treatment[2]	4%	17%	19%	25%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 14: Waste Flows for Scenario 3: Residual to high biomass new techs only

Waste Collected for:				**2008**	**2015**	**2020**	**2031**
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting			25%	25%	26%	26%
	Dry Recyclables	Reuse		1%	1%	1%	1%
		Source Segregated Collections		22%	17%	18%	18%
		Comingled Collections (i.e. MRF Input[1])		31%	36%	37%	38%
		Direct Delivered		19%	19%	18%	18%
	Source Segregated Organics	Composting (Windrow and IVC)		26%	27%	26%	25%
		Anaerobic Digestion (AD)		0%	0%	0%	0%
Residual Treatment	Total Residual Treatment			75%	75%	74%	74%
	Direct to Incineration	Input		28%	44%	43%	25%
		Metals Recycling		3%	3%	3%	3%
		Recycled Bottom Ash		24%	24%	24%	24%
		Landfilled Bottom Ash		3%	3%	3%	3%
		Mass Loss (Combustion)		70%	70%	70%	70%
	MBT / MHT	Input		9%	46%	52%	75%
		Recycling		0%	8%	8%	9%
		SRF		22%	20%	20%	20%
			% SRF to Incineration	100%	69%	52%	31%
			% SRF to Gasification	0%	31%	48%	69%
		Landfilled Rejects		36%	33%	33%	33%
		Mass Loss (Water)		41%	38%	38%	38%
	Landfill	Input - Direct to Landfill		62%	10%	5%	0%
		Input - Rejects from Sorting / Treatment[2]		4%	17%	19%	26%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 15: Waste Flows for Scenario 4: Focus on Dry + Landfilling

Waste Collected for:			**2008**	**2015**	**2020**	**2031**
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	46%	51%	61%
	Dry Recyclables	Reuse	1%	3%	4%	5%
		Source Segregated Collections	22%	23%	24%	23%
		Comingled Collections (i.e. MRF Input[1])	31%	31%	31%	29%
		Direct Delivered	19%	12%	11%	11%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	24%	24%	24%
		Anaerobic Digestion (AD)	0%	7%	8%	8%
Residual Treatment	Total Residual Treatment		75%	54%	49%	39%
	Direct to Incineration	Input	28%	61%	68%	49%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	13%	14%	17%
		Recycling	0%	0%	0%	0%
		SRF	22%	22%	22%	22%
		% SRF to Incineration	100%	100%	100%	100%
		% SRF to Gasification	0%	0%	0%	0%
		Landfilled Rejects	36%	36%	36%	36%
		Mass Loss (Water)	41%	41%	41%	41%
	Landfill	Input - Direct to Landfill	62%	27%	18%	34%
		Input - Rejects from Sorting / Treatment[2]	4%	7%	8%	8%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 16: Waste Flows for Scenario 5: Focus on Dry + Low Biomass New Techs

Waste Collected for:			2008	2015	2020	2031
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	46%	51%	61%
	Dry Recyclables	Reuse	1%	3%	4%	5%
		Source Segregated Collections	22%	23%	24%	23%
		Comingled Collections (i.e. MRF Input[1])	31%	31%	31%	29%
		Direct Delivered	19%	12%	11%	11%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	24%	24%	24%
		Anaerobic Digestion (AD)	0%	7%	8%	8%
Residual Treatment	Total Residual Treatment		75%	54%	49%	39%
	Direct to Incineration	Input	28%	61%	68%	49%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	26%	25%	51%
		Recycling	0%	2%	2%	3%
		SRF	22%	26%	26%	27%
		% SRF to Incineration	100%	77%	70%	42%
		% SRF to Gasification	0%	23%	30%	58%
		Landfilled Rejects	36%	34%	35%	34%
		Mass Loss (Water)	41%	37%	38%	36%
	Landfill	Input - Direct to Landfill	62%	14%	8%	0%
		Input - Rejects from Sorting / Treatment[2]	4%	11%	11%	19%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 17: Waste Flows for Scenario 6: Focus on Dry + High Biomass New Techs

<table>
<tr><th colspan="3">Waste Collected for:</th><th>2008</th><th>2015</th><th>2020</th><th>2031</th></tr>
<tr><td rowspan="7">Reuse / Recycling / Composting</td><td colspan="2">Total Reuse, Recycling and Composting</td><td>25%</td><td>46%</td><td>51%</td><td>61%</td></tr>
<tr><td rowspan="4">Dry Recyclables</td><td>Reuse</td><td>1%</td><td>3%</td><td>4%</td><td>5%</td></tr>
<tr><td>Source Segregated Collections</td><td>22%</td><td>23%</td><td>24%</td><td>23%</td></tr>
<tr><td>Comingled Collections (i.e. MRF Input[1])</td><td>31%</td><td>31%</td><td>31%</td><td>29%</td></tr>
<tr><td>Direct Delivered</td><td>19%</td><td>12%</td><td>11%</td><td>11%</td></tr>
<tr><td rowspan="2">Source Segregated Organics</td><td>Composting (Windrow and IVC)</td><td>26%</td><td>24%</td><td>24%</td><td>24%</td></tr>
<tr><td>Anaerobic Digestion (AD)</td><td>0%</td><td>7%</td><td>8%</td><td>8%</td></tr>
<tr><td rowspan="16">Residual Treatment</td><td colspan="2">Total Residual Treatment</td><td>75%</td><td>54%</td><td>49%</td><td>39%</td></tr>
<tr><td rowspan="5">Direct to Incineration</td><td>Input</td><td>28%</td><td>60%</td><td>67%</td><td>48%</td></tr>
<tr><td>Metals Recycling</td><td>3%</td><td>3%</td><td>3%</td><td>3%</td></tr>
<tr><td>Recycled Bottom Ash</td><td>24%</td><td>24%</td><td>24%</td><td>24%</td></tr>
<tr><td>Landfilled Bottom Ash</td><td>3%</td><td>3%</td><td>3%</td><td>3%</td></tr>
<tr><td>Mass Loss (Combustion)</td><td>70%</td><td>70%</td><td>70%</td><td>70%</td></tr>
<tr><td rowspan="7">MBT / MHT</td><td>Input</td><td>9%</td><td>26%</td><td>26%</td><td>52%</td></tr>
<tr><td>Recycling</td><td>0%</td><td>5%</td><td>4%</td><td>7%</td></tr>
<tr><td>SRF</td><td>22%</td><td>21%</td><td>21%</td><td>21%</td></tr>
<tr><td>% SRF to Incineration</td><td>100%</td><td>80%</td><td>75%</td><td>49%</td></tr>
<tr><td>% SRF to Gasification</td><td>0%</td><td>20%</td><td>25%</td><td>51%</td></tr>
<tr><td>Landfilled Rejects</td><td>36%</td><td>34%</td><td>35%</td><td>34%</td></tr>
<tr><td>Mass Loss (Water)</td><td>41%</td><td>39%</td><td>40%</td><td>39%</td></tr>
<tr><td rowspan="2">Landfill</td><td>Input - Direct to Landfill</td><td>62%</td><td>14%</td><td>8%</td><td>0%</td></tr>
<tr><td>Input - Rejects from Sorting / Treatment[2]</td><td>4%</td><td>11%</td><td>11%</td><td>19%</td></tr>
<tr><td colspan="7">Notes:
1) 10% of this input is assumed to be rejected and landfilled.
2) % of ‘Total Residual Treatment’.</td></tr>
</table>

Table 18: Waste Flows for Scenario 7: Focus on Food + Landfilling

Waste Collected for:			2008	2015	2020	2031
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	46%	51%	61%
	Dry Recyclables	Reuse	1%	3%	4%	5%
		Source Segregated Collections	22%	23%	24%	23%
		Comingled Collections (i.e. MRF Input[1])	31%	31%	31%	29%
		Direct Delivered	19%	12%	11%	11%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	24%	24%	24%
		Anaerobic Digestion (AD)	0%	7%	8%	8%
Residual Treatment	Total Residual Treatment		75%	54%	49%	39%
	Direct to Incineration	Input	28%	61%	68%	49%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	13%	14%	17%
		Recycling	0%	0%	0%	0%
		SRF	22%	22%	22%	22%
		% SRF to Incineration	100%	100%	100%	100%
		% SRF to Gasification	0%	0%	0%	0%
		Landfilled Rejects	36%	36%	36%	36%
		Mass Loss (Water)	41%	41%	41%	41%
	Landfill	Input - Direct to Landfill	62%	27%	18%	34%
		Input - Rejects from Sorting / Treatment[2]	4%	7%	8%	8%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 19: Waste Flows for Scenario 8: Focus on Food + Low Biomass New Techs

Waste Collected for:			2008	2015	2020	2031
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	46%	51%	61%
	Dry Recyclables	Reuse	1%	3%	4%	5%
		Source Segregated Collections	22%	23%	24%	23%
		Comingled Collections (i.e. MRF Input[1])	31%	31%	31%	29%
		Direct Delivered	19%	12%	11%	11%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	24%	24%	24%
		Anaerobic Digestion (AD)	0%	7%	8%	8%
Residual Treatment	Total Residual Treatment		75%	54%	49%	39%
	Direct to Incineration	Input	28%	61%	68%	49%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	26%	25%	51%
		Recycling	0%	2%	2%	3%
		SRF	22%	26%	26%	27%
		% SRF to Incineration	100%	77%	70%	42%
		% SRF to Gasification	0%	23%	30%	58%
		Landfilled Rejects	36%	34%	35%	34%
		Mass Loss (Water)	41%	37%	38%	36%
	Landfill	Input - Direct to Landfill	62%	14%	8%	0%
		Input - Rejects from Sorting / Treatment[2]	4%	11%	11%	19%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 20: Waste Flows for Scenario 9: Focus on Food + High Biomass New Techs

Waste Collected for:			2008	2015	2020	2031
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	46%	51%	61%
	Dry Recyclables	Reuse	1%	3%	4%	5%
		Source Segregated Collections	22%	23%	24%	23%
		Comingled Collections (i.e. MRF Input[1])	31%	31%	31%	29%
		Direct Delivered	19%	12%	11%	11%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	24%	24%	24%
		Anaerobic Digestion (AD)	0%	7%	8%	8%
Residual Treatment	Total Residual Treatment		75%	54%	49%	39%
	Direct to Incineration	Input	28%	60%	67%	48%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	26%	25%	52%
		Recycling	0%	5%	4%	7%
		SRF	22%	21%	21%	21%
		% SRF to Incineration	100%	80%	75%	49%
		% SRF to Gasification	0%	20%	25%	51%
		Landfilled Rejects	36%	34%	35%	34%
		Mass Loss (Water)	41%	39%	40%	39%
	Landfill	Input - Direct to Landfill	62%	14%	8%	0%
		Input - Rejects from Sorting / Treatment[2]	4%	11%	11%	19%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 21: Waste Flows for Scenario 10: Recycling Collections from Doorstep Only + High Biomass New Tech

Waste Collected for:			2008	2015	2020	2031
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	44%	49%	59%
	Dry Recyclables	Reuse	1%	3%	4%	5%
		Source Segregated Collections	22%	23%	24%	23%
		Comingled Collections (i.e. MRF Input[1])	31%	32%	32%	30%
		Direct Delivered	19%	12%	11%	11%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	23%	23%	24%
		Anaerobic Digestion (AD)	0%	6%	6%	7%
Residual Treatment	Total Residual Treatment		75%	56%	51%	41%
	Direct to Incineration	Input	28%	59%	64%	46%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	28%	29%	54%
		Recycling	0%	6%	5%	7%
		SRF	22%	21%	21%	21%
		% SRF to Incineration	100%	79%	70%	46%
		% SRF to Gasification	0%	21%	30%	54%
		Landfilled Rejects	36%	34%	34%	34%
		Mass Loss (Water)	41%	39%	39%	39%
	Landfill	Input - Direct to Landfill	62%	13%	7%	0%
		Input - Rejects from Sorting / Treatment[2]	4%	12%	12%	20%

Notes:

1) 10% of this input is assumed to be rejected and landfilled.
2) % of 'Total Residual Treatment'.

Table 22: Waste Flows for Scenario 11: 'Max Greenhouse Gas Abatement'

Waste Collected for:			2008	2015	2020	2031
Reuse / Recycling / Composting	Total Reuse, Recycling and Composting		25%	46%	59%	69%
	Dry Recyclables	Reuse	1%	3%	3%	4%
		Source Segregated Collections	22%	24%	24%	24%
		Comingled Collections (i.e. MRF Input)	31%	32%	31%	31%
		Direct Delivered	19%	13%	11%	10%
	Source Segregated Organics	Composting (Windrow and IVC)	26%	19%	16%	16%
		Anaerobic Digestion (AD)	0%	10%	15%	15%
Residual Treatment	Total Residual Treatment		75%	54%	41%	31%
	Direct to Incineration	Input	28%	60%	79%	60%
		Metals Recycling	3%	3%	3%	3%
		Recycled Bottom Ash	24%	24%	24%	24%
		Landfilled Bottom Ash	3%	3%	3%	3%
		Mass Loss (Combustion)	70%	70%	70%	70%
	MBT / MHT	Input	9%	26%	17%	40%
		Recycling	0%	5%	0%	5%
		SRF	22%	21%	22%	21%
		% SRF to Incineration	100%	80%	100%	66%
		% SRF to Gasification	0%	20%	0%[1]	34%
		Landfilled Rejects	36%	34%	36%	35%
		Mass Loss (Water)	41%	39%	41%	40%
	Landfill	Input - Direct to Landfill	62%	14%	4%	0%
		Input - Rejects from Sorting / Treatment	4%	11%	9%	16%

1) The proportion of SRF to gasification drops back down to 0% in 2020 as no new residual treatment, above the baseline, is required in that year. This is because the overall recycling rate increases beyond the 50% set out in the draft MWMS. The 100% SRF to incineration figure relates to the continued use of the output of the ELWA MBT as a fuel for combustion in a thermal facility, the assumption being that the proportion and destination of SRF produced won't change over time.

A.5.0 Economic Modelling Assumptions

It should be noted that the cost modelling has been undertaken with a view to:

- Seeking to preserve some 'reality' in the modelling of the costs of switches in management practice; and
- Seeking to ensure that flexibility to changes in the parameters which the GLA may seek to vary (following hand-over of the model) is preserved.

Sections A.5.1 to A.5.4 provide information on all cost-related assumptions used in our model, which are also summarised in Section 7.0 of the Main Report.

A.5.1 Chosen Cost Metric

We have carried out modelling using the 'private' cost metric, reflecting the costs, including taxes and subsidies, faced by operators in the UK waste market. Landfill Tax charges, and support mechanisms such as the Renewables Obligation (RO) are included. The private metric applies a private Weighted Average Cost of Capital (WACC) valuing the opportunity cost of capital investments – either the cost of capital charges, or the opportunity cost of not reinvesting capital in an alternative project. For our analysis, the WACC varies from 10% - 15% depending on the infrastructure to which it is applied.

The costs are presented in real 2009 sterling values. Where estimates are based on figures from earlier years, these are inflated by the relevant GDP deflator.

A.5.2 Costs and Gate Fees

Where matters of cost are concerned, the waste sector is typically used to dealing with the issue in terms of 'gate fees'. Gate fees are not 'costs', and there are various reasons why the gate fee at a facility may differ from average costs, or marginal costs, as they might be conventionally understood. Gate fees may, depending upon the nature of the treatment, be affected by, *inter alia*:

- Local competition (affected by, for example, haulage costs);
- Amount of unutilised capacity;
- The desire to draw in, or limit the intake of, specific materials in the context of seeking a specific feedstock mix;
- Strategic objectives of the facility operator; and
- Many other factors besides.

Any one of these can influence the market price, or gate fee, for a service offered by a waste management company.

Another feature of the waste treatment market at present is the use of long-term contracts in the municipal waste market to procure services. The nature and length of these contracts, and the nature and extent of the risks which the public sector may wish to transfer to the private sector, influences the unitary payment, or gate fee, offered under any given contract. The nature of risk transfer may relate, for example, to technology and its reliability, or to specific outputs which a contract seeks to deliver, and these may, in turn, relate to existing policy mechanisms such as the Landfill Allowances Trading Scheme (LATS). In the merchant sector, the key risk relates to the supply of waste into the plant, and financiers will be keen to satisfy themselves that the amount of waste available is sufficient for their plant. However, once a merchant plant is up and running, in principle, it might be capable of charging lower gate fees to customers.

The key point is that the nature of the risk transfer associated with a given contract, as well as the source of the finance, affects gate fees. In the municipal waste sector, contract prices are typically wrapped up in the form of a Unitary Payment, which may be composed of a number of different elements associated with the delivery of the contract against the specified outputs. This 'unitary payment' is typically determined on a contractual basis, and so is somewhat different to gate fees which might be realised at facilities operating in a more openly competitive market. In the merchant sector, gate fees are not constrained by the same contracts, and the aim is to run the facility profitably, in the context of ensuring financing costs are met. In the approach used in this study, issues of risk transfer are not considered.

It should be noted that whilst much of the major infrastructure for municipal waste has, in the past, been financed using project finance, it remains possible that corporate finance (in balance sheet) could be used to support projects in future. This would have the effect of changing the cost of capital used to support any given project. Finally, local authorities themselves may increasingly make use of Prudential Borrowing, particularly for items for which the quantum of capital required is relatively small (such as some biowaste treatment facilities).

Generally, therefore, the costs we have used may be different from 'gate fees' or 'unitary payments' which may be experienced in a given contractual agreement, or spot market transaction, though they will approximate to them under the private metric that we apply in this study. In general, the calculated costs might be lower than gate fees / unitary payments agreed under local authority contracts, except in those cases where, locally, either markets are very competitive, or strategic actions of operators have the effect of depressing gate fees in the area.

If operating in a truly competitive market with shorter term contracts, the gate fees charged by merchant plant should be close to the costs of the treatment process as estimated under the private metric approach.

It should also be recognised that different treatments are more and less sensitive to variables which underpin the analysis of costs. For example, changes in the cost of capital (see Section A.5.4.1) affect the unit (per tonne) cost of more capital intense treatments in a more significant way than they do for those with lower unit capital costs. Similarly, assumptions concerning Landfill Tax, and ROC values will affect different treatments in different ways.

In summary, this is not a straightforward analysis to carry out. However, the treatment cost derived from analysis under the private metric should bear a close resemblance to costs as they are experienced by actors in the market place.

A.5.3 The Nature of 'Switches'

The nature of 'switches' varies in the profundity of the change in waste management system that they imply. For example, some merely imply the direction of waste away from one management route (e.g. landfill) into another (e.g. incineration). However, others imply a switch from one management route (e.g. landfill) to another (e.g. recycling) which may imply a change in collection system *as well as* the management of the material. These might be referred to as 'treatment switches', and 'system switches', respectively. The latter are far more difficult to model.

Where additional waste is being collected for recycling, for example, the costs of doing this depend on a whole host of factors, not least of which is how that additional material is being obtained (i.e. what combination of change in system, change in participation, change in capture rate, change in relative collection frequency of recycling and refuse, etc.), and the costs of this change relative to a given baseline. In the general case, these costs could be positive or negative, depending upon the assumptions one was to use concerning how the additional material is collected, and the nature of any counterpart changes in the collection system.

A.5.4 Key Economic Modelling Variables

A.5.4.1 Weighted Average Cost of Capital

There is no readily available figure for the WACC in the waste sector. The Committee on Climate Change (CCC), in commissioning a report requiring the development of marginal abatement cost curves for the waste sector, originally proposed the use of a default figure of 10%.[18] Subsequently, a report citing a figure estimated by Oxera of 4.7-5.3% emerged. [19] [20] Both of these seem rather low in our experience, especially insofar as the municipal waste sector is concerned.

A possible explanation follows:

- The Oxera work used a two stage approach to assessing the cost of capital to firms.
    - The first was a high-level sectoral examination, which used data from different sources to estimate sectoral averages. These themselves vary, with regulators' estimates being at the higher end. The 4.7-5.3% range was derived from this first stage alone; and

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[18] CCC (2008) The Committee On Climate Change's Methodology And Approach To Using Marginal Abatement Cost Curves To Derive Domestic Carbon Budgets, Internal Draft.

[19] CCC Shadow Secretariat (2008) *Capital Costs, Discount Rates, and MAC Curves*, Internal paper

[20] Oxera Consulting (2007) Economic Analysis for the Water Framework Directive: Estimating the Cost of Capital for the Cost-Effectiveness Analysis, Financial Viability Assessment and Disproportionate Costs Assessment—Phase II, Report for Defra, DfT and the Collaborative Research Programme, June 20th 2007. It should be re-emphasised that these are intended to represent the WACC in real terms. As such, the implied nominal rates would be higher owing to the effects of inflation.

    - The second was based on examining firm-specific differences to assess the actual cost of capital to specific types of firm. The size of firm, and potential constraints experienced by investors, were considered at this stage.

- The second stage essentially led to significant uplifts in the cost of capital.

The waste sector's WACC is affected by the risk associated with the investment being made. As the waste sector shifts away from 'traditional ways' of doing things, and to the extent that contract structures seek to ensure risk is borne, where appropriate, by the private sector, so the cost of capital appears to have increased. Many investments in the municipal sector are financed using project finance, with Special Purpose Vehicles (SPVs) set up for the purpose of delivering a specific service, or range of services. SPVs are financed using debt and equity, with the equity investors expecting greater returns on their investment. The ratio of debt:equity will have an influence on the effective cost of capital to the company concerned. It may well be that in future, more investments are financed corporately, with associated impacts on the weighted cost of capital. Interestingly, Ernst & Young, advisers on many PFI projects in the waste sector, assumed a 15% real pre tax cost of capital for gasification, pyrolysis and anaerobic digestion, and, reflecting Ilex's analysis of CHP, 12% cost of capital for incineration with CHP (it is not clear, from the Ilex analysis that the 12% figure is a real, as opposed to nominal, cost of capital).[21] These figures seem to reflect experience in the context of municipal contracts.

It seems possible that the average cost of capital may be lower in 'merchant' transactions where the transfer of risk is not explicitly priced in to the cost of capital. However, obtaining financial support for a given project may be more difficult owing to the issues associated with securing supply of waste into the project. Moreover, for fixed-throughput infrastructure, such as incinerators or gasification plant, investors would expect a higher rate of return than for a municipal contract where the supply is secure.[22]

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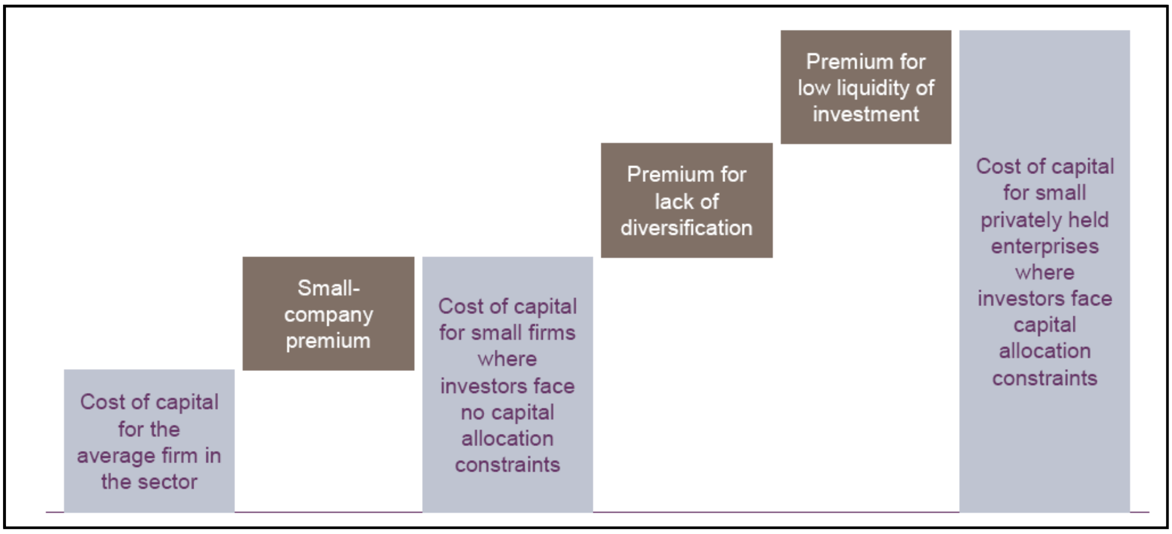
[21] Ernst & Young (2007) Impact of Banding the Renewables Obligation – Costs of Electricity Production, Report to DTI, April 2007.

[22] Audit Commission (2008) *Well disposed: Responding to the Waste Challenge.* Local Government National Report, September 2008.

Figure 5: Adjustments Made to Average Cost of Capital Estimates



Source: Oxera

In our modelling, we have therefore taken the following approach:

- We have used Ernst & Young's figure of 15% for large capital items of infrastructure such as incinerators, for MBT plant, and for the less well established technologies of MHT (autoclave) and gasification;
- We have used a figure of 12% for items of infrastructure where the quantum of capital required is lower (IVC and AD plants). This reflects the fact that treatment facilities are likely to be constructed outside of contracts on a more commercial basis; and
- We have used a lower figure of 10% for collection and sorting systems, as well as for landfill and open air windrow composting facilities.

This reflects, we believe, a reasonable assessment of the opportunity cost of capital going forward. It seems reasonable to suggest, however, that there might be variations in the cost of capital across technology types, and between contract (and risk-sharing) structures. For example, local authorities might well be more inclined to have recourse to Prudential Borrowing where the quantum of capital associated with a given treatment project is relatively small.

It is worth stating that the current environment is one in which the availability of credit is constrained, leading to a worsening in the terms upon which credit is made available. This would be expected to increase the cost of capital. However, the analysis here is forward looking, and extends beyond the short-term so we consider the above figures to be reasonable looking forward.

A.5.4.2 Revenue from Electricity Sales

The wholesale price for electricity, 7.2p/kWh, is the central value contained within the most recent updated energy projection (UEP) published by DECC.[23]

The nature of Power Purchase Agreements (PPAs) and the quality of the deal they deliver for generators, varies considerably. In our modelling, we have assumed that the generator benefits from a proportion of the wholesale price, with the default figure set at 80%. The generator thus receives 5.8p/kWh.

A.5.4.3 Revenues from Heat Sales

A value for heat sales of £30/MWh is given by Ernst & Young, based on the company's proprietary data, in a review for BERR/Defra of the initial business case for renewable heat.[24] These figures seem rather high. More recent work by Jacobs for SEPA used a lower figure of 1.5p/kWh (£15/MWh).[25]

In this study, whereby facilities would typically export heat rather than displace alternative fuel costs, a heat offtake price of £15/MWh has been assumed.

For AD where biogas is injected to the grid, we assume revenues equal to the wholesale price of gas, minus a supplier margin. We assume the income received is 80% of the wholesale price of gas. The wholesale price is taken to be 1.5p/kWh, based on data from DECC.[26] Therefore the revenue received is 1.2p/kWh.

A.5.4.4 Renewable Heat Incentive

The UK Government intends to introduce a Renewable Heat Incentive (RHI), with a planned implementation date of April 2011. RHI payments will be funded by a levy on suppliers of fossil fuels for heat, including gas suppliers, and suppliers of coal, heating oil and liquefied petroleum gas (LPG). The RHI will apply at all scales, covering a wide range of technologies including biogas produced from anaerobic digestion (for localised heat use) and injection of biomethane into the gas grid.

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[23]DECC(2009) Energy and emissions projections webpage, Table E: price assumptions, available at http://www.decc.gov.uk/en/content/cms/statistics/projections/projections.aspx (accessed 3rd November 2009)

[24] Ernst & Young (2007) *Renewable Heat Initial Business Case*, Report to Defra/BERR, 20 September 2007

[25] Jacobs (2008) Development of a Policy Framework for the Tertiary Treatment of Commercial and Industrial Wastes: Technical Appendices, Report for SNIFFER / SEPA, March 2008.

[26] DECC (2009) Average Prices of Fuels Purchased by the Major UK Power Producers and of Gas at UK Delivery Points. Table 3.2.1, Energy Statistics: Prices on DECC website. Available at http://www.decc.gov.uk/en/content/cms/statistics/source/prices/prices.aspx (accessed 14th September 2009). The most recent annual figure, of 1.481 pence per kWh for 2008 is rounded up to 1.5pence per kWh.

The DECC consultation suggests that the tariff for biogas on-site combustion will be 5.5p/kWh for installations up to 200kW. [27] The consultation proposes that biomethane injection into the grid, at all scales, should receive 4p/kWh.

The consultation also suggests that in the case of energy from waste (CHP), RHI support will be paid on the biomass content, taken to be 50% unless proven otherwise. The level of support, for installations of 500kWth and above is set at between 1.6 and 2.5p/kWh. For modelling purposes, we assume a level of support of 2p/kWhth, attributed to the non-fossil proportion of the feedstock for all heat generating residual treatment routes.

A.5.4.5 Revenues from Sales of Biomethane for Transport Use

There is very little use of biomethane for transport in the UK at present, and accordingly, cost data is not widely available. Use of biomethane is focused on local authority transport fleets, such as buses and refuse collection vehicles, where refuelling takes place at a depot. Information from a supplier has indicated that they are able to provide biomethane at a cost of between £0.65 and £0.75 per kg, before duty and VAT. [28] Using the lower end of the quoted price range, we model on the basis of revenues of £0.65 per kg, which equates to £0.46 per cubic metre, based on the density of CH_4 of 0.71kg/Nm^3.

A.5.4.6 ROC Values and Feed-in Tariff

We use the weighted average of ROC values for 2009, which is £51.16/MWh.[29] As with electricity revenues, we have assumed that 80% of the ROC value (£40.93/MWh) is realised by the generator in the default situation. ROCs only apply to Landfill Gas (0.25 ROCs/MWh), Good Quality CHP (1 ROC/MWh for the biomass fraction), gasification (2 ROCs/MWh for the biomass fraction) and AD (2 ROCs/MWh)

From April 2010, there will be a Feed-In Tariff (FIT) available for smaller (<5MW) generators of renewable electricity although landfill gas will not be eligible.[30] Installations of capacity 50kW and below will only be eligible for FITs, while operators of facilities of between 50kW – 5MW will be able to make a one-off choice between the FIT and the RO.

The FIT is made up of the following components:

- A fixed payment from the electricity supplier for every kilowatt hour (kWh) generated (the "generation tariff"). For electricity generated from AD, the

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[27] DECC (2010) Renewable Heat Incentive: Consultation on the proposed RHI financial support scheme, February 2010.

[28] Personal communication with Stephen McCulloch, Chesterfield Biogas. 10th September 2009

[29] Non-Fossil Purchasing Agency website, Average ROC prices webpage. Available at http://www.e-roc.co.uk/trackrecord.htm (accessed January 2010).

[30] DECC (2010) Feed-in Tariffs: Government's Response to the Summer 2009 Consultation, February 2010

generation tariff is proposed to be 9p/kWh for facilities greater than 500kW, and 11.5p/kWh for facilities of 500kW and below.

- Another payment additional to the generation tariff for every kWh exported to the wider energy market (the "export tariff"). This guaranteed price is to be set at 3p/kWh;

For the purposes of the modelling we assume that AD operators opt for the FIT, receiving both the generation tariff and the export tariff, as these would be considered more 'bankable' by financiers than ROCs.

A.5.4.7 Levy Exemption Certificates for Good Quality CHP

Fuel used by energy from waste projects qualifying as Good Quality CHP (certified via the CHP Quality Assurance Programme (CHPQA)) is exempt from the Climate Change Levy (CCL). Electricity from new renewable energy such as anaerobic digestion is also exempt from the levy. Energy from Waste projects that do not meet the CHPQA standards are not eligible.[31]

Under the CCL, electricity is currently (with effect from 1 April 2009) subject to a rate of £4.70/MWh. [32] We assume for modelling purposes that 80% of the value (£3.76/MWh) is realised by the generator.

A.5.4.8 RTFC Values and Road Fuel Duty Derogations

Suppliers of biomethane from anaerobic digestion of MSW are eligible to receive Renewable Transport Fuel Certificates under the Renewable Transport Fuel Obligation (RTFO). One certificate is issued per kg of biomethane supplied.

For the purposes of our modelling, we assume that suppliers operate at a relatively small scale, with a focus on biogas rather than hydrocarbons supply, and are therefore unlikely to be 'obligated suppliers' under the RTFO. The RTFO Order 2007 states:

> *"A renewable transport fuel obligation is imposed on every transport fuel supplier who in a specified period-*
>
> *(a) owns relevant hydrocarbon oil at the time when the requirement to pay duty of excise with which the oil is chargeable takes effect, and*
> *(b) supplies that oil at or for delivery to places in the United Kingdom.*

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[31] Ofgem (2009) CCL:CHP Exemption, Ofgem website, available at http://www.ofgem.gov.uk/Sustainability/Environment/CCLCHPEx/Pages/CCLCHPEx.aspx

(accessed June 2009).

[32] HMRC (2008) Budget 2008, Climate Change Levy: Rates. Available at http://www.hmrc.gov.uk/budget2008/bn84.pdf (accessed 3rd November 2009)

This obligation does not apply to a transport fuel supplier who, in a specified period, supplies less than 450,000 litres in total of the oil (a 'non-obligated' supplier)".

The RTFC buy-out price is set at 15 pence per litre (ppl) for the first two years of operation (from April 2008). This is the price that would be paid by an obligated supplier that had failed to acquire sufficient RTFCs to meet the target (currently 3.25% by volume for 2009-10).[33]

However, unless the target is going to be missed by a considerable margin, it is unlikely that non-obligated suppliers could sell their RTFCs for anything approaching the buy-out price. The first 'e-TOC' auction was held by the Non-Fossil Purchasing Agency on 17th July 2009, with over 5 million 2008/09 RTFCs offered for sale. However, none were purchased by obligated suppliers, due to the surplus of certificates on the market for the first obligation year. [34]

There is also a duty derogation available for road fuel natural gas (i.e. CNG, LNG and biogas, which is effectively a consumption subsidy. The current differential of approximately 41ppl equivalent with petrol and diesel will continue through to 2013.[35] While this may assist in making biomethane more attractive for transport end users, it does not mean that the suppliers can be assumed to benefit from a portion of this subsidy beyond the revenue received (see A.5.4.5). Arguably, the effects of this stimulus to consumption are already reflected in the price received by the supplier.

A.5.4.9 Landfill Tax, Standard Rate

The standard rate of Landfill Tax is currently at a level of £40 per tonne, and will increase at the rate of £8 per tonne per year until it reaches £72 per tonne in 2013.[36] What levels it may be set at beyond this date are not entirely clear.

For the purpose of this analysis, we assume that the tax increases to £72 per tonne, in nominal terms, in 2013. In real terms, the value will be lower than this because of the effects of inflation. In the 2009 Budget Report, there was no announcement of intent to increase rates beyond this point, but we have taken the view for this study that the tax rate remains constant in real terms (i.e. that its nominal rate increases in line with inflation once the £72 per tonne level is reached). [36] We therefore adjust

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[33] Renewable Fuels Agency website, RTFO Targets page. Available at http://www.renewablefuelsagency.org/aboutthertfo/rtfotargets.cfm

[34] Renewable Fuels Digest Issue no.13, August 2009. Available at http://www.renewablefuelsagency.org/_db/_documents/Renewable_Fuels_Digest_issue_13_August_2009.pdf

[35] HM Revenue & Custome (2009) Budget 2009, BN66, Hydrocarbon Oils: Duty Rates, 22 April 2009; Personal Communication with HMRC Transport Tax Team

[36] HM Treasury (2009) Budget 2009: Building Britain's Future, Economic and Fiscal Strategy Report and Financial Statement and Budget Report, April 2009.

the nominal rates of landfill tax to real 2009 prices by the Bank of England's long term inflation target (2.5%) as a deflator. [37]

A.5.4.10 Landfill Tax, Lower Rate

The lower rate of Landfill Tax stood at £2.00 per tonne for many years before it was increased, in 2008, to £2.50 per tonne. The 2009 Budget Report stated that this lower rate applying to inactive wastes will be frozen at £2.50 per tonne for 2010-2011.[36] Therefore, the lower rate tax is assumed to remain constant in nominal terms (from 2009) over time.

A.5.4.11 Material Values

Recycling collection systems recover materials which have a value in the market place. These material values will, of course, be susceptible to considerable fluctuations, as with any commodity. Evidently, the scope for modelling forward commodity prices in the sector is beyond the scope of this study (and in any case, fraught with difficulty).

A set of material values were used in a recent analysis by WRAP.[38] However, although these were not at the higher end of the market at the time, recent events have suggested that caution should be used in using high prices as a basis for extrapolating forward over the longer-term. Equally, it would be wrong to assume the lower values being experienced towards the start of 2009 should be used as a basis for all modelling going forward.

There is no 'right answer' in this matter. Whatever values are used is bound to be considered controversial, but likewise, any assumption around commodity prices is likely to be contentious (and almost certainly wrong).

The values we have used are shown below in Table 23 for municipal waste. The municipal waste values were used in a recent study by Eunomia for WRAP (post price crash) and were viewed as representing what might be a 'medium-term average'.

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[37] HM Treasury Green Book. Available at http://www.hm-treasury.gov.uk/d/green_book_complete.pdf (accessed September 2009).

[38] WRAP (2008) Kerbside Recycling: Indicative Costs and Performance, June 2008.

Table 23: Values for Secondary Materials, Municipal Waste

Material	Price per tonne
Mixed paper	£55
Card	£50
Mixed plastic bottles	£110
Mixed glass bottles/jars	£20
Cans (based on market share between steel and aluminium)	£142
Textiles	£200

A.5.4.12 Landfill Gate Fee, Hazardous

Some facilities generate a residue which is classified as hazardous. Although these are not likely to contribute significantly to emissions, they do exert some influence on the cost of some treatments, for example, incineration.[39] For the purpose of this study, we have not included a model, as such, of a hazardous waste landfill site. We have assumed, however, a cost per tonne of landfilling hazardous waste of £180 before landfill tax, but including transport costs.

A.5.4.13 Landfill Gate Fee, Non-hazardous

We assume a capital cost of £115 per tonne of installed capacity, and operating costs of £7 per tonne, whilst restoration, post-closure and aftercare are estimated to cost a further £7 per tonne. Based on these assumptions, the estimated gate fees modelled for the landfilling of London's waste are summarised in Table 23.

[39] It is not clear that either APC residues, or bottom ash should be considered entirely inert in this regard. Some studies indicate that both residues may include organic carbon which has not been completely combusted, and which is likely, therefore, to contribute to landfill emissions.

Table 24: Non-hazardous Landfill Gate Fees (including Landfill Tax)

Year	2008	2009	2010	2011	2012	2013-2031[1]
Gate Fee	£64	£72	£79	£87	£93	£100

Note:

1. This information was correct at the time of submission of the final version of this report. This note has been added to acknowledge that the 2010 Budget clarified that the Landfill Tax will rise an extra £8 to £80 per tonne in 2014. This further increase has not been included in the modelling undertaken for this study

A.5.4.14 Bottom and Fly Ash from Incineration

For bottom ash, we assume that on average, around two-thirds of material is put to some form of use in the construction industry at a cost of £5/tonne. The remaining third is assumed to be landfilled at non-hazardous waste sites, attracting the lower rate of Landfill Tax.[40] While bottom ash is currently classified as inert, therefore qualifying for lower rate of landfill tax, a Treasury consultation proposes that it could be reclassified, and subject to the Standard Rate of Tax.[41] We have modelled on the basis of the lower Tax rate until any change has been confirmed.

For fly ash, the waste is assumed to be landfilled at a hazardous waste landfill. We have not modelled this explicitly but have used a fixed pre-tax figure for the costs of landfilling, inclusive of haulage, of £180/tonne.

[40] We note that recent sampling by the Environment Agency suggests that bottom ash may, in future, have to be treated as hazardous waste dependent upon the outcomes of tests regarding ecotoxicity.

[41] HM Treasury, HMRC (2009) Modernising Landfill Tax Legislation, April 2009. Available at http://www.hm-treasury.gov.uk/d/Budget2009/bud09_landfill_tax_964.pdf (accessed September 2009).

A.6.0 Collection Cost Assumptions

Changes in treatment options will, in certain cases, require changes in the collection systems being provided. For example, the switch from landfilling to increased treatment of source-segregated biowaste, such as composting or anaerobic digestion implies a change in the way materials are collected, with less being collected as refuse and more being separately collected. This has cost implications quite distinct from those associated with treatment systems.

There are significant problems which one faces in seeking to cost how, at the margin, moving tonnages into one collection system and away from another affects the collection logistics. Much depends upon how that material is being acquired. For example, where households are concerned, this could be through RRCs, through bring sites, or through kerbside collection systems, and where kerbside collection services are concerned, whether it is being acquired through increases in participating households, improved recognition of materials by households, and so on.

It is not true to say, as many economists tend to assume, that increasing recycling rates necessarily increases collection costs. If the way the material is being delivered into the collection system improves the efficiency of collection logistics, marginal costs can be lower than average costs, and average costs fall. We have based our assumptions on average costs of reasonably well performing systems. These may decline in future if policies act to increase the capture of materials per participating household.

Another important feature of collection systems is that collecting different materials separately for recycling has different cost implications depending upon the nature of that material. When expressed per unit weight of material, bulk density plays an important role in determining costs. Materials with lower bulk density in collection occupy more space in vehicles and lead to, other things being equal, vehicles reaching volume capacity more quickly than in cases where materials are of a higher density. This increases the requirement for vehicles and staff, and increases collection costs.

In an ideal world, we would model the marginal costs of adding each material of a given type to a given recycling system, and equivalently, seeing it *not* collected as refuse. In practice, the range of collection systems is large, and such a modelling process is extraordinarily difficult in the absence of some simplifying assumptions. We have differentiated between refuse and other materials as will become clear below. We have done this with a view to giving reasonable average estimates of costs for the collection of different materials as far as possible.

A.6.1 Collection of Dry Recyclables from the Kerbside

The cost to a Borough for recovering recyclables, from the kerbside, will comprise of the ‘collection only’ cost and either a) a revenue stream if the materials are collected separated at source or b) a cost for sorting in a MRF if the materials are collected as co-mingled.

Our assumptions for revenues from the sales of recyclables and be found in Section A.5.4.11. The 'sort' costs, or gate fees for MRFs can be found in Section A.7.8. Only the *total costs* of collection are presented in Sections A.6.1.1 to A.1.1.1.

A.6.1.1 Doorstep Properties

Doorstep properties are those which receive a collection service whereby the householder has to place their own 'box' or 'bag / sack' of recyclables on the doorstep to their property.

Each type of collection system currently existing in London, along with best practice schemes, was given a unique identification code. This related to the number of materials collected and the frequency of collection. For each type 'collection only' costs were taken from a WRAP study into indicative kerbside collection costs.[42] Estimated yields of each of the main materials were taken from the more recent WRAP study into kerbside performance.[43] For the kerbside sort systems these yields were then used, along with material revenues, to calculate the total revenue, per household, from each system.

For co-mingled systems, the type and number of materials collected provide an indication as to the type of MRF required to split the various waste fractions. These different types of MRF attract varying gate fees. In this context, it should be noted that MRF gate fees vary significantly across the country, and that there is very little associated data available, mainly due the commercial sensitivity around contracts.[44] Using point estimates for sort costs is not ideal, but in the absence of knowledge of the details of each collection or sorting contract in London, no better methodology can be adopted within the scope of this study.

The kerbside collection costs for dry recyclables, used in this study, are summarised below in Table 25.

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[42] WRAP (2008) *Kerbside Recycling: Indicative Costs and Performance. Technical Annex,* Available at: http://www.wrap.org.uk/downloads/KerbsideReportAnnexFinal_1.bac022de.5634.pdf

[43] Icaro Consulting (2009) Analysis of kerbside dry recycling performance in England 2007/08 (WRAP Project EVA034-087),Summary Report

[44] WRAP (2009) Gate Fees Report, Eunomia on behalf of WRAP, 2009

Table 25: Kerbside Dry Recycling Costs

Collection Method	Materials Collected	Recycling Frequency	Refuse frequency	Collection Only Costs (£/hhld /yr)	Total Cost Net of Revenues / Sort Costs (£/hhld /yr)
Twin Stream Co-mingled	Paper/Card, Glass, Tins, Plastic	Fortnightly	Fortnightly	£10.95	£9.69
	Paper/Card, Glass, Tins, Plastic	Fortnightly	Weekly	£10.87	£9.90
Single Stream Co-mingled	Paper/Card, Glass, Tins, Plastic	Fortnightly	Fortnightly	£11.35	£17.06
	Paper/Card, Glass, Tins, Plastic	Weekly	Weekly	£9.91	£14.86
	Paper/Card, Glass, Tins, Plastic	Weekly	Fortnightly	£16.85	£22.90
	Paper/Card, Glass, Tins, Plastic	Fortnightly	Weekly	£8.35	£12.38
Kerbside Sort	Paper/Card, Glass, Tins, Plastic bottles	Weekly	Weekly	£20.06	£11.40
	Paper/Card, Glass, Tins, Plastic bottles	Fortnightly	Weekly	£16.01	£8.42
	Paper/Card, Glass, Tins, Plastic bottles	Weekly	Fortnightly	£24.71	£15.49
	Paper/Card, Glass, Tins, Plastic bottles	Fortnightly	Fortnightly	£21.00	£12.65
	Paper, Glass, Tins, Plastic bottles	Weekly	Fortnightly	£22.21	£12.85
	Paper, Glass, Tins, Plastic bottles	Weekly	Weekly	£17.56	£10.12
	Paper, Glass, Tins, Plastic bottles	Fortnightly	Fortnightly	£18.50	£10.13
	Paper, Glass, Tins, Plastic bottles	Fortnightly	Weekly	£13.51	£7.03
Single Stream Co-mingled	Paper/Card, Tins, Plastic	Fortnightly	Fortnightly	£11.29	£14.52
	Paper/Card, Tins, Plastic	Weekly	Weekly	£9.91	£12.75
	Paper/Card, Tins, Plastic	Fortnightly	Weekly	£8.29	£10.71
Kerbside Sort	Paper, Glass, Tins	Weekly	Fortnightly	£18.82	£10.89
	Paper, Glass, Tins	Fortnightly	Fortnightly	£12.27	£5.13
	Paper, Glass, Tins	Weekly	Weekly	£13.86	£7.55
	Paper, Glass, Tins	Fortnightly	Weekly	£9.29	£3.77
Single Material	Paper/Card	Fortnightly	Weekly	£8.00	£2.09
	Paper/Card	monthly	Weekly	£8.00	£3.43
	Paper	Fortnightly	Fortnightly	£5.00	£0.88
	Paper	Fortnightly	Weekly	£5.00	£0.60
	Paper	Weekly	Weekly	£5.00	£1.15
	Paper	monthly	Weekly	£5.00	£1.43

A.6.1.2 Communal Properties

Communal properties have been defined as households of multiple occupation (HMO), tower blocks, mansion blocks, and estates receiving a bring style collection. Households of this nature receiving a door-to-door collection i.e. a box or sack collection from the door of the property have been treated as doorstep properties and are discussed separately.

There are a number of communal collection schemes in place and we have modelled each of these in the 'Do Nothing New' baseline. We have assumed in all of the scenarios that all communal households will switch to the best performing collection scheme, but have maintained the existing type of collection (co-mingled or source separated). We have not differentiated these properties in terms of type of container (see Table 26 for type of containers used).

There are several difficulties in estimating the cost of communal schemes. Historically, this has not been a well researched area. We have therefore used a number of sources in order to estimate such costs and related performance. The baseline costs for a weekly co-mingled and source separated system were calculated by Eunomia for a report developed for the London Borough of Hackney in 2008.[45] This report captured actual information from collection rounds and calculated the cost per household for a number of systems. The data has been adjusted to reflect captures, material revenues and gate fees for this report.

In order to calculate a per household cost for fortnightly systems capturing different materials we have assumed that the proportional reduction in cost that occurs for doorstep properties can be applied to communal properties. Table 27 describes the cost per household for each of the modelled systems.

We have assumed a weekly co-mingled system sack based collection has broadly similar costs to that of 240 litre bins for doorstep collection.[46] In a communal bin the added cost of a 1100 litre bin will result in a marginal cost differential between sack and 1100 bin.

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[45] Eunomia (2008) Estates Recycling in the London Borough of Hackney

[46] WRAP (2008) *Kerbside Recycling: Indicative Costs and Performance. Technical Annex,* Available at: http://www.wrap.org.uk/downloads/KerbsideReportAnnexFinal_1.bac022de.5634.pdf

Table 26: Container Types

Container	Collection System	Cost for Kerbside Collection
240 litre plus bin	Co-mingled (Weekly)	£22.7
Sack	Co-mingled (Weekly)	£23..25
Sack	Source Separated (Weekly)	N/A
Frame	Source Separated (Weekly)	N/A

Table 27: Modelled Cost

Collection System	Frequency	Materials	Cost per Household	Source
Co-mingled	Weekly	All materials	£31.98	Hackney Study
Co-mingled	More than weekly	All materials	£39.98	WRAP data
Co-mingled	Fortnightly	All materials	£25.56	WRAP data
Co-mingled	Weekly	No Glass	£32.93	WRAP data
Twin Stream	Fortnightly	No Glass	£25.56	WRAP data
Twin Stream	Fortnightly	All materials	£25.56	WRAP data
Source Separated	Weekly	All materials	£27.45	Hackney Study
Source Separated	More than weekly	All materials	£34.31	WRAP data
Source Separated	Weekly	No Card	£30.94	WRAP data
Source Separated	Weekly	No Card/Plastic	£26.22	WRAP data
Source Separated	Weekly	No Plastic	£26.22	WRAP data
Co-mingled	Weekly	All materials	£31.98	Hackney Study

A.6.2 On-the-Go Recycling

We have assumed in the modelling that the cost of on-street recycling bins will be confined to a capital cost, as the additional cost of collection are integrated into the current collection service. Capital costs have been supplied by London Boroughs and are broadly similar. These are detailed in Table 28. The revenue from material sales is difficult to determine due to lack of data recording.

Table 28: On-the-go Recycling Assumptions

Material	Capital Cost per Bin
Co-mingled collection of glass, cans, paper and plastic bottles	£500
Separate collection of glass, cans, paper and plastic bottles	£600
Paper	£400

A.6.3 Reuse and Recycling Centres

The approach taken to modelling the change in cost of reaching the required level of performance on RRC sites has been to treat all 35 RRC sites in the GLA as though they constitute one site. This approach reflects the likelihood that performance (and the steps already planned or taken to improve performance) vary considerably between individual sites, meaning that it would be impossible in a study of this nature to model costs at every individual site.

In reality, different sites will target different materials and performance improvement initiatives in different orders in future years. As all of the material that needs to be recycled in each scenario will have to be captured at some point, we have made the somewhat crude (but necessary) methodological decision to calculate an overall average cost per tonne of additional recycling at RRC sites for all materials. This is a less problematic methodology for RRC sites than for household collected waste because a larger part of the cost of dealing with any tonne of waste is fixed (i.e. provision of infrastructure, staff etc.) rather than variable (i.e. the cost or revenue associated with a particular material).

It should be emphasised that for RRCs, we have not modelled *marginal* costs per tonne of improvement, but *average* costs per tonne.

A bespoke model was developed to calculate the average per tonne cost using assumptions derived from the National Assessment of Civic Amenity Sites (NACAS) study, which sought, primarily by means of multiple regression analysis of date from hundreds of RRC or CA sites across the UK, to understand the factors that lead to increased recycling and re-use performance at RRC sites.[47] These measures have been supported by a 2008 report examining best practice for RRCs in London.[48] The NACAS study was able not only to identify the factors that had led to improved performance, but also to report the probable contribution to recycling performance that introducing different initiatives would have on site recycling rates. Key factors reported by NACAS as having the most significant impact on performance and which were taken into account in our modelling include:

- The number of bulk recyclables collected (e.g. garden waste, timber etc);
- The number of small recyclables collected (e.g. paper, glass etc);
- The presence of a system for reuse which collects a substantial range of reusable items;
- The quality, clarity and completeness of signage;
- The number and effectiveness of staff on site; and
- The presence of economic incentives to recycle for the service provider.

Within the model, we have made assumptions as to the number of opportunities across the whole network (based on current performance) to introduce each of the performance initiatives outlined above.

We subsequently attributed costs to each of those initiatives in terms of both capital investment (e.g. where new roll-on roll-off containers would be required) and increased operating costs (e.g. increased staffing levels). Capital investments were assumed to be financed over the estimated useful life of each type of asset, allowing a 'revenue equivalent' cost to be calculated for each additional tonne of recycling. The cost of capital for such initiatives is a variable in the model, set at 3.5% for the purposes of this study, approximately reflecting rates currently available from the Public Works Loan Board over a 10 year period.

The model also considered the extent of contribution that could be made by:

- Introducing facilities to separate a wider range of recyclables;
- Improving re-use facilities;
- Introducing contract incentives;
- Increasing CA dedicated management time;
- Introducing a 'meet and greet' staff role;
- Introducing dedicated training programmes for site staff;
- Increasing the daily compliment of staff on site;
- Increasing completeness of signage;

[47] Future West and Network Recycling (2004) *National Assessment of Civic Amenity Sites*, Final Report for Biffaward, March 2004

[48] Resource Futures (2008) London Reuse and Recycling Best Practice Guidance, RF Project no.: 376

- Increasing clarity of signage;
- Increasing quality of signage;
- Improving communication with the public on RRC sites;
- Improving recycling facility/bin order;
- Improving site layout and traffic management; and
- Introducing trade waste control improvements.

The impact of each 'material specific' (e.g. targeting a new material) initiative was then calculated, generating an amount of additional tonnage for each of the materials affected.

The impact of the 'non-material specific' initiatives (e.g. increasing general staffing levels) was then distributed across all materials targeted for recycling (and re-use, which we have essentially included as a component of recycling), whilst ensuring that the ultimate capture rate for each material was consistent with the mass flow model.

The income received or payment made (net of transport cost) by reprocessors for each material was then applied to the number of additional tonnes of each material recycled, to calculate the impact (positive or negative) of material values.

Table 29 outlines the headline figures used in order to arrive at the per tonne figure used in the modelling. Current tonnage information has been extracted from WDF. We have modelled improvements on current systems based on the reported operation of the RRCs across all Boroughs, available in the Resource Futures report. Revenues from materials include haulage. The cost associated with the implementation of initiatives, such as contract incentives and staff training, originate from the NACAS report and have been adjusted for inflation.

Table 29: Headline RRC Figures

Parameter	**Annual Costs**
Total site infrastructure	£19,755
Initiatives	£32,172
Revenue from additional materials	£5,249
Total cost	£57,177
Total new materials managed (tonnes)	115,631
Cost per tonne (£s)	**£49.45**

A.6.4 Collection for Reuse

For the costs of collection for reuse, we have taken information from a number of industry sources.[49] An average cost of collection of £400/tonne, for the most common materials and products reused, such as wood, furniture, white goods and rubble, is used in our model. We acknowledge that this might be considered to be a higher figure than quoted for some individual projects, but we believe it represents a reasonable mean value based on the data made available for this study.

A.6.5 Refuse Collection

The costs of refuse collection vary depending on the frequency of collection and the type of container. The costs in the Table below are based on internal data held by Eunomia. When (bin) collection systems are switched from weekly to fortnightly a saving of £10 per household per annum is achieved.

Table 30: Refuse Collection Costs

System Type	Collection Cost (£/hhld/annum)
Communal Bin - More than weekly	£35
Sack - More than weekly	£35
Bin - Weekly	£30
Communal Bin - Weekly	£20
Sack - Weekly	£20
Bin - Fortnightly	£20
Communal Bin - Fortnightly	£18

A.6.6 Organic Waste Collection

The costs of collecting food and garden waste have mostly been taken from a study undertaken by Eunomia on behalf of WRAP, which assessed the management of organic wastes from households in the UK.[50] Costs of garden waste collection for communal properties have been assumed to be the same as for doorstep properties,

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[49] These include LCRN and Caroline Lee-Smith, formerly of FRN, now an independent consultant

[50] Eunomia (2007) *Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-Benefit Analysis*, Appendices to the Final Report, WRAP. See http://www.wrap.org.uk/downloads/Biowaste_CBA_Report_Appendices_May_2007.e79686b2.3823.pdf

as no suitable data exists by which to make another assumption, albeit such costs are likely to be different. Costs of collecting source separated food waste from communal properties, however, has been estimated higher than for doorstep properties, at £15/hhld/annum, although it should be acknowledged that in the absence of any robust data, this is a 'best guess' estimate. Further research into the collection costs of food waste from 'hard to reach' properties in London would help tighten such assumptions.

Where a charged garden service is in place, the cost to Boroughs has been assumed as zero. This is because the collection costs will be paid for by 'pay-per-use' or annual charges levied on households.

The costs modelled for garden and food waste collection services in this study are summarised in Table 31 and Table 32.

Table 31: Modelled Cost of Garden Waste Collection Systems

Charging Basis	Frequency	Container	Cost (£/hhld/annum)
Charged	Monthly	Renewable Sack	£0.0
	Monthly	Non-Renewable Sack	£0.0
	Fortnightly	Renewable Sack	£0.0
	Fortnightly	Non-Renewable Sack	£0.0
	Fortnightly	240L Bin	£0.0
	Weekly	Renewable Sack	£0.0
	Weekly	Non-Renewable Sack	£0.0
Free	Fortnightly	Renewable Sack	£6.3
	Fortnightly	Non-Renewable Sack	£7.0
	Fortnightly	180L Bin	£8.3
	Weekly	Renewable Sack	£14.3
	Weekly	Non-Renewable Sack	£15.9

Table 32: Modelled Cost of Food Waste Collection Systems

System Type	Collection Frequency	Cost (£/hhld/annum)
Source Separated Kitchen Waste – Doorstep	Weekly	£8.6
Source Separated Kitchen Waste – Communal	Weekly	£15.0
Comingled Kitchen and Garden Waste	Fortnightly	£10.9
	Weekly	£20.8

A.6.7 Commercial Waste Collection

Within each subsector of commercial waste the size of each business varies significantly. As a result we have not modelled the differences between sub-sectors (offices, hospitality and retail) separately. Costs for the collection of commercial waste have been derived from internal data and, where possible, supplemented by data from the aforementioned study undertaken by Hyder Consulting. The internal data used is based on that used by Eunomia in previous studies on behalf of the Committee on Climate Change and Defra / WRAP.[51] [52]

[51] Eunomia (2008) Development of Marginal Abatement Cost Curves for the Waste Sector. Report for Committee on Climate Change, December 2008

[52] Eunomia (2010) The Environmental, Economic and Practical Impacts of Landfill Bans: Feasibility Research. Report for WRAP, March 2010

A.7.0 Treatment Process Cost Assumptions

A.7.1 Open-air Windrow Composting

Open-air windrow composting schemes are relatively low-cost processes. In 2002, Eunomia modeled costs for open-air windrow facilities of £14.47 - £20 per tonne (net of compost sales) for low- and high-specification windrow facilities.[53] These figures are only marginally higher today.

AEA Technology has also examined the effects of scale on gate fees for open air windrow composting.[54] These figures seem high, with gate fees supposedly never dropping below around £23 per tonne, even at a scale of 200,000 tonnes (which is more or less unprecedented for such facilities). Recent work for WRAP confirms this with gate fees ranging from £17-£33 per tonne with a median figure of £22.50 per tonne.[55] The AEA study gave no information on unit capital costs, even though the study sought to demonstrate economies of scale at different plant sizes.

We have modelled on the basis of a facility of the order 20,000 tonnes with a 15 year lifetime, and have taken figures from previous studies undertaken by Eunomia, and inflated these to give a unit capital cost, including land, of £85 per tonne of throughput.[56] We have tested this with industry representatives who have confirmed this as a sensible figure.

Operating costs have been estimated at £5 per tonne of throughput before disposal of rejects. Annual maintenance costs are modelled as 3% of unit capital cost per tonne, which equates to £2.55 per tonne throughput.

For rejects, we have assumed 5% of input material is sent to landfill.[57] This is assumed to attract Landfill Tax at the standard rate.

The revenues from sales of compost are frequently ignored in studies assessing treatment costs. However, revenues from compost sales have the potential to increase in significance as energy prices increase. In most countries with more mature compost markets, as more material becomes available, so there tends to be

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[53] Eunomia (2002) The Legislative Driven Economic Framework Promoting MSW Recycling in the UK, Final Report to the National Resources and Waste Forum.

[54] AEA Technology (2007) *Economies of Scale – Waste Treatment Optimisation Study* by AEA Technology, Final Report, April 2007

[55] WRAP (2008) *Comparing the Cost of Alternative Waste Treatment Options*, http://www.wrap.org.uk/downloads/W504GateFeesReport_FINAL.c9481350d.5755.pdf

[56] Eunomia (2002) *The Legislative Driven Economic Framework Promoting MSW Recycling in the UK*, Final Report to the National Resources and Waste Forum;

[57] In theory, one might suggest that this type of material could be used for other purposes. In practice, rejects from garden waste facilities tend to consist more of grit and stones, and to a lesser degree, materials associated with garden implements which find their way into the facility. The potential for, for example, energy recovery is less obvious with such reject streams.

more effort spent in marketing products, and refining them for specific end-use markets. This does not always translate into higher revenues. However, the revenues are likely to be higher as the costs of gas (and other energy sources) increases, with gas being a feedstock for synthetic nitrogen fertilisers. On the other hand, in some parts of the UK, farmers' perceptions of compost are still influenced by the livestock pathogen outbreaks of the recent past, which have made farmers more risk averse in their attitude to compost use.

ADAS reports a figure for the value of nutrients of the order £10 per tonne of compost.[58] A report for the Joint Research Centre shows average values for composts obtained in different countries (see Table 33). All of these are positive with median UK figures being between €0.7- €20.00 per tonne of fresh matter. We have assumed a value of £1.25 per tonne of waste input for compost (equivalent to around £2.50 per tonne of compost, towards the lower end of the range suggested in Table 33).

A.7.2 In-vessel Composting

In-vessel composting (IVC) systems come in various shapes and sizes. They can be vertical or horizontally aligned. Unit capital costs depend upon, for example:

- Scale of facility;
- Nature of process used (and the degree to which the process is managed through 'fixed capital' rather than mobile equipment);
- Materials treated;
- Nature of exhaust air treatment; and
- Time spent in the intensive and maturation phases.

Typically, for systems in the UK, capital costs have been relatively low (of the order £150 per tonne of capacity). However, there might be reasons to expect these to be somewhat higher in cases where:

- The food waste component is higher, giving rise to a need for more thorough management of the input materials (notably to ensure adequate structural material is present through mixing), requiring more expensive treatment of exhaust air; and/or
- Concerns regarding odour are expected to be significant, again affecting exhaust air treatment.

[58] See *'Compost lowering costs for farmers'*, accessed from letsrecycle.com, 10 July 2007, http://www.letsrecycle.com/do/ecco.py/view_item?listid=37&listcatid=217&listitemid=10069

Table 33: Average Market Prices for Compost in Different Sectors across EU Member States (€/t fresh matter)

Sector	BE/FI	CZ	DE	Fi	ES	GR	HU	IE	IT	NL-bio	NL green	SE	SI	UK	EU Mean
Agriculture (food)	1.1		14.0	0.0	27.0*	-	15.0	-	3.0	-4.0	2.0	0.0	-	2.9	**6.1**
vineyards, orchards	1.1	-	-	-	-	-	-	-	12.0	-	-	-	-	2.9	**5.3**
Organic farming	1.1	-	-	-	-	42.0	-	-	-	-	-	-	-	2.9	**15.3**
Horticulture & green house production	1.1	-	15.0	-	-	42.0	-	-	-	-	-	-	-	2.9	**15.3**
Landscaping	2.5	4.5	15.0	2.0	-	-	18.0	-	25	4.0	-	-	-	6.5	**9.7**
Blends	1.1[2)]	-	-	2.0	-	-	-		-	3.5	-	-	-	2.9	**2.4**
Blends (bagged)[1]	-	-	-	-	-	-	-	90.0	200.0	-	-	-	-	-	**(145)**
Soil mixing companies	1.1	-	-	2.0	-	-	-	-	-	-	-	-	-	6.5	**3.2**
Wholesalers	1.1	-	-	-	-	-	-	-	-	-	-	-	12.0	-	**6,6**
Wholesalers (bagged)[1]	-	-	160.0	-	-	-	-	-	-	-	-	-	-	-	**(160)**
Hobby gardening	7.2	4.5	-	10.0	-	-	20.0	-	13.0	0.3	-	-	21.0	20	**12.0**
Hobby garde-ning (bagged)[1]	-	-	-	-	-	300.0	-	-	-	-	-	-	-	-	**(300)**
Mulch	-	-	-	-	-	-	-	-	-	-	-	-	-	3.6	**3.6**
Land restoration, landfill covers	1.1	-	-	0.7	-	0.0	-	-	-	-	-	-	-	0.7	**0.6**

Notes:

1) High prices because sold in small bags (5 to 20 litres)

For operating costs of IVC facilities, Jacobs suggest a figure of £18 per tonne at a 30,000 tonne plant, which we understand includes maintenance costs. We have used a figure of £10 per tonne for operating costs. Maintenance costs are not included in this operating cost figure, but are included in the annual costs, at 5% of capital cost, representing £8.25 per annum.

We assume rejects are 5% of input material and that these are sent to landfill where they attract landfill tax at the standard rate.

As with open-air facilities, we have attributed to compost a revenue of £1.25 per input tonne to the facility.

A.7.2.1 Ammonia Scrubbing

There is potential for GHG abatement through the use of scrubbers before biofilters at in-vessel compost plants. The costs of the scrubber relate to the volume of air-flow through the scrubber. For a 20,000 tonne per annum plant, the airflow would be, at a maximum, around 40,000 m^3/hr. A suitable scrubber with circulation pump, tank for sulphuric acid and tank for ammonium sulphate would cost of the order £100,000 including additional piping (somewhat less – £70,000 or so - for the scrubber alone). We therefore model on the basis of a capital cost of £5 per tonne for our 20,000tpa reference facility.

Operating costs associated with electricity use, use of concentrated sulphuric acid, use of water, maintenance, and management of residue (ammonium sulphate) have been estimated at £1.24 per tonne of waste input.

A.7.3 Anaerobic Digestion

Like IVC systems, AD facilities come in different shapes and sizes. Most digesters have vertical tanks, but some are horizontal. Mechanisms vary considerably and a number of patented processes exist. Processes may:

- Operate at high or low solids content;
- Operate at mesophilic or thermophilic temperatures;
- Be one- or two- stage in nature; and
- Be continuous or batch processes.

AD facilities can also be used to generate energy in a number of ways, the costs for which are summarised in Sections A.7.3.1 to A.7.3.4.

A.7.3.1 AD with Electricity Only

There is a dearth of experience with the anaerobic digestion of source-separated municipal wastes in the UK. The continental experience is far richer in this regard. There have been some reviews of the costs of anaerobic digestion in Europe. A recent study found considerable variation in costs across different technology suppliers. The reader is referred to the full report for details and to our earlier review.[59]

Greenfinch, whose process is currently the subject of a Demonstrator Project under Defra's New Technologies Programme, stated:

> *For a commercial operation where the boroughs are responsible for delivering the organic waste to a facility which is owned by a private operation and which derives the benefits from the by-products, the commercial gate fee would be between £40 and £50 per tonne.*[60]

Greenfinch gave figures for capital costs of £4 million for 20kt, and operating costs of £20 per tonne including rejects, but before revenues.[61] These are likely to be lower costs than would be realizable under a contractual situation.

There is some uncertainty about what contract prices might look like in the UK situation given the lack of experience here, and the fact that the UK approach to procurement appears to have the potential to increase prices significantly (through requests for comprehensive guarantees, and associated risk transfer mechanisms).

Capital costs for AD facilities used to deal with household, or industrial food wastes (and other biowastes) tend to be of the order £250 - £350 per tonne depending upon scale and the nature of the facility. We have estimated unit capital costs at £300 per tonne.

In a feasibility study for Northern Ireland, suppliers were asked about the capital costs for facilities of given sizes.[62] The results are shown in Table 34. It can be seen that the capital costs vary enormously, rather more for a given scale plant than the operating costs. This, combined with the different ways of treating capital costs, makes it difficult to generalize concerning the costs of digestion plants. Particularly when dealing with source segregated fractions of municipal waste, digesters tend to be more or less heavily engineered to deal with potential contraries. In addition, post-digestion processes vary across suppliers. It was not always clear, from the financial breakdowns offered, how suppliers had accounted for UK-specific issues in respect of planning, permitting and contracting.

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59 Leif Wannholt (1999) Biological Treatment of Domestic Waste in Closed Plants in Europe - Plant Visit Reports, RVF Report 98:8, Malmo: RVF. Hogg et al (2002) Costs for Municipal Waste Management in the EU, Final Report to DG Environment, European Commission.

60 Greenfinch (2003) Presentation by Greenfinch Ltd Based on Anaerobic Digestion: City Solutions Day, New & Emerging Technologies for Waste, February 2003.

61 Greenfinch (2003) Presentation by Greenfinch Ltd Based on Anaerobic Digestion: City Solutions Day, New & Emerging Technologies for Waste, February 2003.

62 Eunomia (2004) *Feasibility Study Concerning Anaerobic Digestion in Northern Ireland,* Final Report for Bryson House, ARENA Network and NI2000.

Table 34: Key Financial Data for Digestion Plant

CAPACITY	Capacity (ktpa)							
	10	20	25	50	50	50	75	165
Total Investment Cost (£M)	£3.13[1]	£3.00[2]	£12.68	£6.00	£17.60	£16.00	£16.00	£20.05
Unit Investment Cost (£/tonne)	£313	£150	£507	£120	£352	£320	£213.33	£121.49
Unit Operating Cost (£/tonne)	£27.14	£20.00	£20.24	£18.00	£15.72	£28.00	£22.67	£22.20

Notes:

1. Including land
2. Excluding land

A study for Remade Scotland suggested that plant treating municipal waste would have investment costs ranging from £3 million for a 5,000 tonne/year plant (£600 per tonne capex) to £12 million for a 100,000 tonne /year plant (£120 per tonne capex), with operating costs between £100,000 (£20 per tonne capex) and £900,000 per year (£9 per tonne capex).[63] These operating costs would include a revenue offset associated with energy generation and use.

The Annex to the English Waste Strategy gives the following capital costs:

1. 20 ktpa - £7.3 million (£365 per tonne capex);
2. 50 ktpa - £14.7 million (£294 per tonne capex); and
3. 150 ktpa - £28.8 million (£190 per tonne capex).

A Juniper report invited offers for a theoretical facility treating 30,000 tonnes of food waste and 10,000 tonnes of slurry. The figures obtained from respondents using extensive pre-treatment were, adjusting for recent movements in the exchange rate, of the order £4.2-5.0 million (or £140 – 167 per tonne of capex if one considers the

[63] Fabien Monnet (2003) An Introduction to Anaerobic Digestion of Organic Wastes, Remade Scotland, 2003.

food only).[64] The operating costs were quoted as £340,000, or around £11 per tonne. We assume these are net of revenues from energy sales since they are so low.[65]

Jacobs estimate capital costs for a 40,000 tonne AD facility for commercial and industrial waste at £266 per tonne of annual capacity, with operating costs of £28 per tonne before revenues.[66] Short suggests that capital and operating costs will vary as follows:[67]

- 5,000 tpa: Capex £1.8 m = £360 per tonne with Opex £20 per tonne;
- 10,000 tpa: Capex £3.0 m = £300 per tonne with Opex £17 per tonne;
- 20,000 tpa: Capex £5.0 m = £250 per tonne with Opex £14 per tonne;
- 30,000 tpa: Capex £6.8 m = £227 per tonne with Opex £13 per tonne;
- 40,000 tpa: Capex £8.4 m = £210 per tonne with Opex £12 per tonne;
- 50,000 tpa: Capex £9.9 m = £198 per tonne with Opex £11 per tonne.

One can see, therefore, a very wide variation in the capital cost figures being quoted, and the variation cannot be explained by factors such as scale alone, partly because of the variety of technical designs on offer.

We have used a figure of £300 for unit capital costs. For operating costs, we have good reason to believe that if one is seeking a figure before revenue generation from energy sales, and disposal of rejects, the figures above are all too low. We have used a figure of £30 per tonne. We believe this to be representative of facilities of scale 20-30,000 tonnes capacity, with appropriate post-treatment of the digested biowaste.

A.7.3.2 AD with CHP

The issue of CHP is discussed in this document with regard to both thermal facilities and AD facilities. Where thermal facilities are concerned, and where steam turbines are used to generate energy, there is a trade-off between the generation of electricity and the generation of heat. AD systems usually generate energy using gas engines. Where gas engines are concerned, the generation of heat incurs little penalty in terms of electricity generation, and the majority of facilities operate CHP engines, partly to ensure the provision of free heat which is needed to keep the feedstock at the required (mesophilic or thermophilic) temperature, as well as providing heat for hygienisation of the feedstock in the wake of the EU Animal By Products Regulations.

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[64] The slurry was deemed to have only 5% solids content so only 5% or so of the solids would be in the slurry.

[65] Juniper (2007) Commercial Assessment: Anaerobic Digestion Technology for Biomass Projects, Report for Renewables East, June 2007.

[66] Jacobs (2008) Development of a Policy Framework for the Tertiary Treatment of Commercial and Industrial Wastes: Technical Appendices, Report for SNIFFER / SEPA, March 2008.

[67] J Short (2008) Anaerobic Digestion and Alternative Waste Treatment Technologies, Deconstructing AD, Presentation to MRW Conference, May 2008.

There is some likelihood that AD facilities will be operated at smaller scale than incinerators and that the total heat delivered is likely to be less than in what may be larger incinerators. CHP units are also likely to be used to generate energy anyway, which makes it more likely that, where AD is concerned, heat use may be more possible, and may be possible on a more opportunistic basis. It also implies that at least as far as generation equipment is concerned, the incremental costs are low (close to zero). In addition, the associated costs of infrastructure for delivering heat may be lower in a given area (as fewer end users would need to be served).

The relatively small number of publicly available studies that have looked at the issue of CHP generation have tended to support the view that where thermal facilities are concerned, effective utilization of CHP is likely to be predicated upon district or community heating schemes. Where AD is concerned, this is less likely to be necessary. AEA argue that where sewage treatment works are concerned:[68]

> *'the option of heat recovery for additional heat (over and above what is required for the process) is generally not implemented as the value is low, there are limited opportunities for use on site (occasionally there are some works offices) and the cost of sale to other customers is too high as they will seldom be in close proximity to the water treatment works.'*

They estimate the cost of the pipes and trenching to be of the order £1 million per mile of trench.

For AD, an indication of the sort of differential between CHP and non-CHP configurations was given by Jacobs, who suggest that for a 40,000 tonne plant, the capital costs increase from £10.62M to £11.48M, or from £266 to £287 per tonne of annual throughput. The operating costs were estimated to remain constant.[69]

In this study, we have estimated capital costs for the useful deployment of CHP of an additional £1.65 million in capital terms for a 20,000 tonne per annum facility. This estimate lies between the Jacobs estimate and that implied by Ilex (see Section A.7.5.2 below) for a heat network.[70] We have also added £1 per tonne to the operating costs. Evidently, this must be treated as a 'best guess', and the specifics will vary with the location and local opportunities for heat use of any given plant. Therefore for modelling purposes, we assume a total capital cost of £382.50 per tonne, and an operating cost of £31/tonne.

A.7.3.3 AD with Gas Upgrading for Use as Vehicle Fuel

The costs of gas upgrading tend to be expressed relative to the flow rate of biogas into the cleaning process. A number of different processes exist for cleaning up

[68] AEA Energy and Environment (2008) The Evaluation of Energy from Biowaste Arisings and Forest Residues in Scotland, Report to SEPA, April 2008.

[69] Jacobs (2008) Development of a Policy Framework for the Tertiary Treatment of Commercial and Industrial Wastes: Technical Appendices, Report for SNIFFER / SEPA, March 2008.

[70] Ilex Consulting (2005) *Eligibility of Energy from Waste – Study and Analysis*, Final Report to the DTI, March 2005.

biogas (mainly for CO_2 removal, but also for scrubbing of H_2S), including chemical absorption, pressure swing adsorption, water scrubbing, and membrane separation. These processes are developing in terms of their energy use per unit of gas cleaned, and the extent to which methane is lost in the process. The aim, evidently, is to improve process efficiency without adding significantly to cost.

Much of the information offered is in terms of the cost per unit of gas cleaned, or per unit of energy in biogas delivered. However, this is not especially useful for this study as we are seeking information of the change in capital cost at the AD plant, as well as in the operating cost. It is important in this regard to note that biogas upgrading is not simply an additional cost. If the intention is to make use of biogas as vehicle fuel, there are savings to be made in terms of the avoided cost of CHP generation, and of connection to the electricity grid.

SLR estimate costs for a packaged gas engine generator set, up to about 1MWe, installed in a container ready for connection to the site switchboard, at about £600/kW (with costs per kW falling thereafter to £450-£500/kW).[71] For a 20-30,000 tonne plant, the generation is of the order 1MW, with associated capital costs of around £600,000. However, as the facility would still make use of a gas engine for heat and power, albeit a smaller gas engine, to power the process and keep the feedstock at the required temperature, any saving on CHP generation would be a proportion of the £600,000 figure. We estimate an avoided cost of £200,000 for a 20,000 tonne plant.

AEA notes:

> *The costs of connection local to the generation project will be borne by the developer of the biowaste project. These costs will include:*
>
> - *Works on the site of the generation (e.g. new transformers, switchgear etc).*
> - *Any new or upgraded cable (over or underground) from the biowaste site to the nearest suitable connection point on the network.*
> - *Additional or upgraded transformers and switchgear at the connection point.*
>
> *The size of the generator, the distance to the connection point and the voltage level at which the connection for connection will determine the scale of costs for the local connection. The costs of additional or upgraded transformers and switchgear at the connection point will depend on the level (if any) of unused capacity on the existing grid equipment.*

As this configuration of AD would not be exporting power, it is important to provide analysis of the costs of a grid connection, which would be avoided. The costs of

[71] SLR (2008) Cost of Incineration and Non-incineration Energy-from-waste Technologies, Report to the Mayor of London, January 2008.

connection and overhead lines will be specific to a given project. SLR suggests the following figures for 11, 33 and 132kV connections and overhead lines:

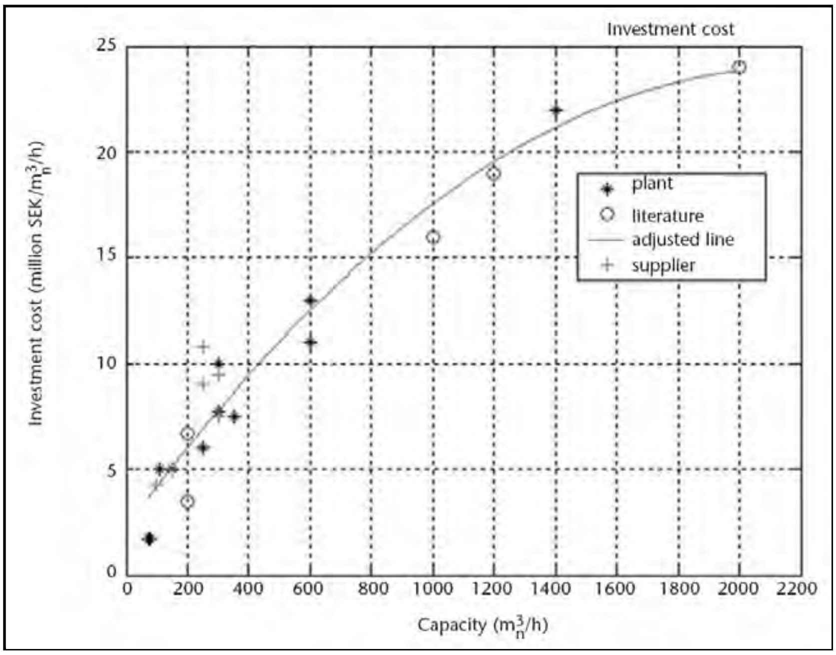
- 11kV Grid connection equipment: £20,000 - £60,000;
- Overhead line: £15,000 - £30,000/km;
- 33 kV Grid connection equipment: £120,000 - £150,000;
- Overhead line: £20,000 - £35,000/km;
- 132 kV grid connection equipment: £800,000 - £1,000,000

They also note that in addition to these figures, the time taken to get permission to connect to the grid can be important.

Small generation projects will be connected to the lower voltage distribution network. For a 20,000 tonne plant, generating some 0.5 -0.75 MW electricity, an 11kV connection should suffice. Taking into account cabling costs (which are variable depending upon distance from the grid), we estimate a total fee to be of the order £150,000.

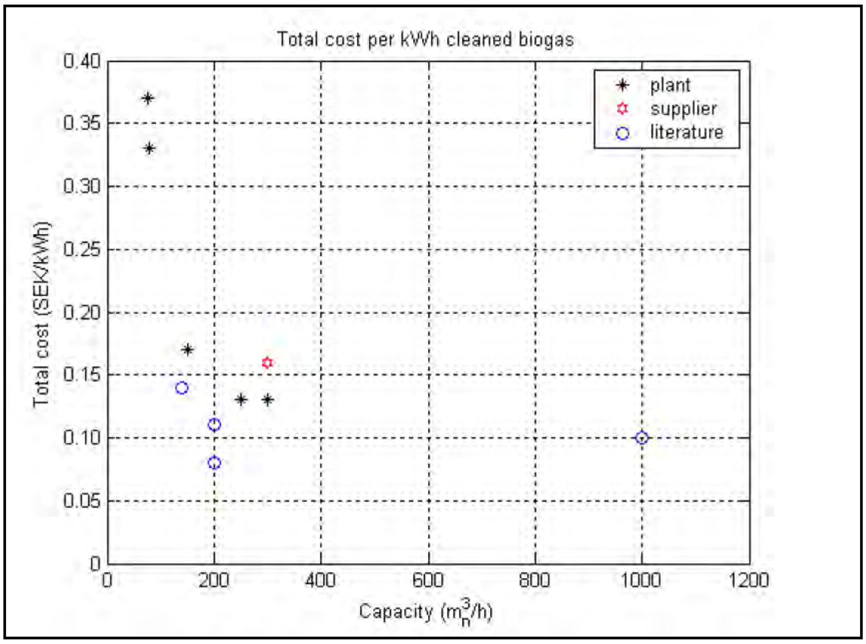
For gas upgrading, the unit costs fall as the flow rate into the clean-up system increases. This is shown in Figure 6. The implications for unit costs (under specific assumptions regarding the cost of capital and the investment life-time) are shown in Figure 7.

Figure 6: Investment Cost for Biogas Clean-up as a Function of Capacity (m^3/hr)



Source: O. Jonsson and M Persson (2003) Biogas as a Transportation Fuel, Session 1, FVS Fachtagung 2003

Figure 7: Cost per kWh of Cleaned Biogas as a Function of Plant Capacity



Source: Margaretta Persson (2007) Biogas Upgrading and Utilization as a Vehicle Fuel, Paper presented to the European Biogas Workshop, The Future of Biogas in Europe III, 14th June 2007

In our estimation, a 20,000 tonne plant, operating for around 7,500-8,000 hours per annum, would be expected to generate around 400m^3 of biogas per operating hour.

A report by the Institut Catala d'Energia (ICAEN) gave figures, which appear to be 2004 figures, of €600,000 for capital costs, and operating costs of €80,000 per annum, for pressure swing adsorption processes.[72] More recent figures from a Fraunhofer UMSICHT report gives figures for gas cleaning for different processes. At the throughputs we are interested in, investment costs are of the order €1.32 – €1.4 million. Operating costs were €327,000 – €336,000.[73]

We have used, for capital costs, an average of the Fraunhofer figures converted to UK sterling at a long-term exchange rate of £1 = €1.25. These give capital costs of £1.03 million (or around £50 per tonne), and operating costs of £249,000 (or around £12.45 per tonne). In addition, we have added a cost for pipework of £300,000 (£15 per tonne). This reflects figures for 5km of pipework given by Schulz and for a plant in Uppsala reported in an earlier study by Eunomia et al (see Table 35).[74]

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[72] ICAEN (2004) Economic Framework Report, Deliverable for the Altener Project Regulation Draft of Biogas Commercialisation in Gas Grid – BIOCOMM, 2004.

[73] Fraunhofer UMSICHT (2008) Technologien und Kosten der Biogasaufbereitung und Einspeisung in das Erdgasnetz. Ergebnisse der Markterhebung 2007-2008, report for the Bundesministerium fur Bildung und Forschung, April 2008.

[74] W. Schulz (2004) *Untersuchung zur Aufbereitung von Biogas zur Erweiterung der Nutzungsmöglichkeiten*, Bremer Energie-Konsens GmbH

Table 35: Costs of Anaerobic Digestion Facilities

Parameter	Uppsala (30ktpa)[1]	Linköping (100ktpa)[2]
Capital cost (€000s) – digestion plant	€6 000	€5 900
Capital cost (€000s) – gas cleaning and compression	€850	€2 800
Capital costs (€000s) –piping	€330	€550
Variable costs (€000s) /annum	€220	€200 – 400
Notes: 1. Constructed in 1997 2. Constructed in 1996		

Source: Uppsala municipality and Linköping municipality, cited in Eunomia et al (2002)

The additional capex, relative to generation of electricity only, for a 20,000 tpa plant is £1.03 million plus £300,000, giving £1.33 million. Savings associated with avoided costs of grid connection and a smaller gas engine are £150,000 and £200,000 respectively, totalling £350,000. Net increase in capital cost is therefore £980,000, which for a 20,000 tpa facility gives an increased unit capex of £49/tonne. This gives a total capital cost of £349 per tonne.

For operational costs we have added £12.45 per tonne, but subtracted £5 to take account of maintenance costs that we understand are included in the Fraunhofer figures. As for all processes, we calculate maintenance separately based on a proportion of the capital cost. For AD with gas upgrading for vehicle fuel these total £17.45 per tonne per annum. We have subtracted a further £1 to account for a reduction in opex for the smaller CHP plant. This gives total opex of £36.45/tonne.

A.7.3.4 AD with Gas Upgrading for Use in the Grid

To inject gas into the grid the gas must first be cleaned, upgraded, metered and a connection made. The capital cost of upgrading an anaerobic digestion plant for use of the gas in the national grid for subsequent electricity generation from a 2008 Canadian study suggests that total capital costs are in the region of $2.130 (£1.22) million, or £202 per tonne for a 6,000 tonne capacity plant.[75] Eight operational anaerobic digesters that transfer biogas to the national grid for use as electricity were

[75] Electrigaz Technologies Inc. (2008) Feasibility Study – Biogas upgrading and grid injection in the Fraser Valley, British Columbia

examined and an average of these was assessed. The result applies to a plant which produces 240 nm^3/h (approx 6,000 tonne capacity).

The standards applied to the injection of biogas to the grid are expected drive the cost of the upgrade. In Sweden the cleaning required for biogas is the same for vehicle fuel and injection into the grid, as the same standard applies to both. In the UK a regulatory barrier exists due to the oxygen content of pipeline gas which is currently too low to include renewable gas.

We have assumed that the cost of upgrading for use in the grid is similar to that of upgrading for vehicle fuels. This is due to insufficient data to assume otherwise as the technology is not yet available in the UK and there is a lack of transferable information from existing facilities. It is important to recognise, however, that this is likely to be a technically difficult option for the foreseeable future, whatever its presumed merits may be.

A.7.4 Landfill

Our landfill model is broken down into:

- The capital costs for the site. Evidently, these may vary in unit (i.e. per annual tonne treated) terms depending upon the size of the site. We have modelled on the basis, broadly, of:
    - A fill rate of 250,000 tonnes per annum and a lifetime of 12 years; Of course, fill rates and life times vary, as will the total available void of a given site. This was felt to be broadly representative of a modern site, or extension; and
    - Capex, including site assessment, acquisition, site development, restoration and aftercare, was initially estimated at approximately £23.5 million. For the modelling, we have used a figure of £115 per tonne of material accepted at the site each year (in other words, the landfill is being treated as a facility with a 250,000 tonne throughput, with capex of £115 per tonne of that annual throughput);
- Operating costs are estimated at £7 per tonne, before revenues from energy generation, whilst restoration, post-closure and aftercare are estimated to cost a further £7 per tonne;

It should be noted that where a *specific* material is being 'switched' from landfill to another process, the model picks up the relevant energy generation associated with that material.

A.7.5 Incineration

Defra, in the context of the waste strategy, estimated the following capital costs for incineration:

- 100,000 tonnes Capex £64.7 million £647.00 per tonne
- 200,000 tonnes Capex £104.7 million £502.35 per tonne
- 400,000 tonnes Capex £149.1 million £372.75 per tonne.

The costs of such facilities are sensitive to planning risks, and the nature of the procurement process.

There are two incinerators modelled in this study. These are:

- An incinerator delivering electricity only; and
- An incinerator delivering combined heat and power (CHP).

We begin by describing the basic configuration which is the first one above. We then comment on changes from this baseline model.

A.7.5.1 Electricity Only

The incineration model is broken down into:

- A capital cost element:

  Unit capital costs could be quite variable in any given situation and would depend upon scale, the nature of risk transfer, the detailed plant design, the requirements in terms of architecture, the nature of the flue gas cleaning technology etc. Quoted figures do not always include the costs of land, especially now that local authorities are encouraged to acquire sites. The figure we have chosen is felt to be broadly representative of a plant of the order of 200,000 tonnes capacity. There are likely to be some larger facilities constructed, but some smaller ones also. Ilex used a figure, in 2004 prices, of £58.6 million for a 200kt plant.[76] This figure seems very low in the current context of UK municipal waste contracts, which is the context in which most incinerators have been built. SLR looked at plants already built and found that capital costs varied with scale. It is important, in this context, to recognise that costs have escalated quite significantly in recent years. Jacobs suggest a figure of £86.5 million for a 250,000 tonne facility, though this seems low relative to the same company's estimates in the context of a recent procurement in Leeds (see below). [77] Given the recent cost inflation in construction projects, these figures are probably rather low for new-build facilities.

  Fichtner, looking at a one-line 182kt plant, considered that capital costs would be of the order £77.2 million, excluding land acquisition, costs of grid connection, and legal and advisory services, increasing to £98.6 million where the plant had two lines. Design enhancements were thought to be of the order £1.5 million, with grid connection and enabling works of the order £3.5 million. For all costs, including contingency, but excluding land acquisition, the figure was £87.25 million. However, a key factor for incinerators and other capital projects, is the effect of inflation. Particularly in recent years, the costs

[76] Ilex Consulting (2005) *Eligibility of Energy from Waste – Study and Analysis*, Final Report to the DTI, March 2005.

[77] Jacobs (2008) Development of a Policy Framework for the Tertiary Treatment of Commercial and Industrial Wastes: Technical Appendices, Report for SNIFFER / SEPA, March 2008.

of construction and of material have run ahead of conventional indices of inflation. The indexation cost implied for this project owing to inflation over the construction period was £26.5 million.[78] This inflation figure appears to be a figure quoted in nominal, rather than real terms. For the purposes of the analysis, we have assumed capital costs today would be of the order £90 million, with the real effects of indexation likely to be of the order £10 million. We have used a figure of £100 million capex, or £500 per tonne;

- Operating costs:

  For operating costs, before revenues from electricity generation and costs of dealing with residues, we have used a figure of £20 per tonne;

- Revenues from electricity generation:

  These are estimated on the basis of net delivered energy (calculated from the environmental analysis) and using the wholesale price for electricity contained within the most recent updated energy projection (UEP) published by DECC. [79]

- Revenues from ROCs:

  The ROC-able element for the 'electricity only' incinerator is assumed to be zero, so ROC revenues are always zero;

- Costs of dealing with residues, which are estimated as follows:
  - For fly ash, the waste is assumed to be landfilled at a hazardous waste landfill. We have not modelled this explicitly but have used a fixed pre-tax figure for the costs of landfilling, inclusive of haulage.
  - For bottom ash, we assume that on average, around two-thirds of material is put to some form of use in the construction industry. The remaining third is assumed to be landfilled at non-hazardous waste sites, and attracting lower rate landfill tax. There is currently a consultation process ongoing to consider whether bottom ash should attract the standard rate of landfill tax, and in future years it may even attract the higher rate.[80] [81]

A.7.5.2 Incineration with CHP

It is difficult to estimate, with any degree of accuracy, exactly what could be the costs of a CHP system given that so many variables exist. Costs will depend upon the

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[78] Fichtner (2007) Jacobs Leeds Energy-from-Waste: Validation of EFW Costs, 7 September 2007.

[79]DECC(2009) Energy and emissions projections webpage, Table E: price assumptions, available at http://www.decc.gov.uk/en/content/cms/statistics/projections/projections.aspx (accessed 3rd November 2009)

[80] HM Treasury and HMRC (2009) Modernising Landfill Tax Legislation, April 2009. Available at http://www.hm-treasury.gov.uk/d/Budget2009/bud09_landfill_tax_964.pdf (accessed June 2009).

[81] We note that recent sampling by the Environment Agency suggests that in a relatively large minority of samples, bottom ash fails to meet some of the limit values. Bottom ash may, in future, have to be treated as hazardous waste dependent upon the outcomes of tests.

specific design of a given CHP scheme. Not only are there differences related to the nature of the infrastructure required, but also, there will be differences in the impact on the plant itself, depending upon whether the intention is merely to use low grade heat for district heating, or medium or high pressure steam extraction. The former will have little impact on power generation, the latter will have a more significant effect.

Ilex estimated the costs of a CHP system on behalf of BERR.[82] The estimated costs of CHP were based around the development of a 400,000 tone per annum plant, partly because a previous report had suggested that larger plants of this size were likely to be developed.

Costs of CHP relate to:

- Costs of providing heat from the facility (relative to costs of providing electricity only);
- Costs of securing a market for the heat; and
- Loss of revenue from power sales.

The first of these includes the costs of tapping into the steam turbine where the initial design allowed for this (and several have done so, or are planning to do so), provision of heat exchangers, and (depending on the nature of the recipients) provision of back-up boilers. In addition, the infrastructure for heat supply to the users has to be put in place if it does not already exist. The nature of the heat consumer(s) is likely to be a key determinant of these network-related costs. It is difficult to generalise these costs, given the wide variation in the possible networks. In principle, co-location alongside a major industrial heat user would be likely to give lower costs, but in practice, the likelihood of this occurring at conventional incinerators may be low. There might be a higher likelihood of merchant facilities being developed for the off-take of SRF, especially where the heat user is involved in the project itself.

Ilex estimated costs for different CHP plant as shown below in Table 36. These figures were intended to be indicative of costs. The 43MW capacity relates to a heat generation efficiency of around 24%. This is the only CHP option considered by Ilex which seems likely to qualify as 'good quality CHP' as the net efficiency, however measured, is relatively low for the other options considered. The figures in the Table show that the main costs are related to the provision of the network and customer connections, and that in the Ilex assumptions, these show some clear increase with scale, which might not be the case depending upon the nature of customers.

In reviewing the Leeds scheme, Fichtner comments on Jacobs' costs associated with a CHP system which, it is claimed, have been taken directly from a scheme considered for a 250 ktpa EfW facility.[83] The capital costs for the CHP scheme were £33.8 million with an annual operating cost of £320,951. Fichtner comment: '*We understand that these costs are taken from a report completed by ILEX Energy*

[82] Ilex Energy Consulting (2005) *Extending ROC Eligibility to Energy from Waste with CHP*, a supplementary report to the DTI, September 2005.

[83] Fichtner (2007) Jacobs Leeds Energy-from-Waste: Validation of EFW Costs, 7 September 2007.

Consulting and Electrowatt Ekono for the DTI.' The 250 ktpa facility is, in fact, a 400ktpa facility, and one with a power efficiency of 20% and a heat efficiency of 12%. This would imply efficiency of heat generation of the order 20% for a 250ktpa facility, which is quite a low figure (and would, arguably, imply a very poor use of capital in the investment in the heat network).

Table 36: Capex and Opex Assumptions for 400ktpa Incinerator with CHP Plant

Thermal Capacity, MWth	Capex (£s)			Annual Opex (£s)			
	Heat Exchanger	Heat Network	Customer Connections	Pumping	Heat Exchanger	Heat Network	Customer Connections
3	0.19	2.39	0.95	0.01	0.00	0.01	0.02
11	0.62	6.80	3.08	0.02	0.01	0.03	0.06
20	0.88	14.83	5.59	0.03	0.01	0.07	0.11
23	0.90	15.81	6.43	0.04	0.01	0.08	0.13
28	0.90	19.25	7.83	0.05	0.01	0.10	0.16
30	0.95	20.62	8.39	0.05	0.01	0.10	0.17
34	0.95	23.37	9.51	0.06	0.01	0.12	0.19
43	0.98	29.56	12.03	0.07	0.01	0.15	0.24
66	1.00	45.37	18.46	0.11	0.01	0.23	0.37

Note:

1. Costs for heat networks, to a lesser extent, customer connections, will be very site specific and these numbers are intended only to be illustrative

Source: Ilex Energy Consulting (2005) Extending ROC Eligibility to Energy from Waste with CHP, a supplementary report to the DTI, September 2005.

The CHP option which most closely reflects our technical options is that with the higher thermal capacity (even though we are considering a smaller plant). The heat network and the customer connections would, using Ilex's figures, imply additional capital costs of the order £43-£65 million. Perhaps unsurprisingly, in Ilex's analysis, these scenarios – where the heat generation is greatest – are those which appear least favourable from a financial perspective given the power penalty implied by the increase in heat demand. Interestingly, the bottom row of the Table implies a heat generation efficiency of only 24%, with power generation at 17%, implying a much higher power to heat ratio than might be expected in many CHP systems which had what one might call a 'serious' focus on heat provision.

In a report carried out at the turn of the decade for the European Commission, investment costs for power and CHP schemes in Finland were as set out in Table 37.

What this shows is the relative costs of power only schemes to those generating heat and power. The differentials are not trivial. Given that these figures are expressed in Euros in 2000, then accounting for exchange rate movements and for inflation over the last eight years, the figures do not seem so different to those provided by Ilex.

Table 37: Investment Costs for CHP in Finland (in €000, year 2000 base)

	Heating	CHP	Heating	CHP
Capacity, tons/year	40	40		300
Investment	13 336	24 248	52 490	95 437

Source: Eunomia (2002), Costs for Municipal Waste Management in the EU: Annexes, Final Report to DG Environment, European Commission

In another report, Jacobs suggest that at 25,000 tonnes capacity, unit capital cost figures increase by £135 per tonne (or around a 40% increase in costs relative to their power only estimate).

In our analysis, we have used Ilex's figures at the 43 MW size, for a 400,000 tonnes per year plant, implying heat generation at around 30% of input energy. For such a scheme, the following applies:

- Additional capital costs of £43 million;
- Additional operating costs of £0.47 million; and
- ROC-eligibility of incineration depends upon the definition of good quality CHP which is to be used. Plants which meet the criterion of good quality CHP are eligible for ROCs, but on the electricity generation only. We have assumed that the plants operate above the relevant threshold and, as a result, ROCs are received for the electricity.

We therefore model on the basis of additional capex of £107.50 per tonne, and additional opex of £1.18 per tonne. This gives a total capital cost of £607.50 per tonne, and an operational cost of £21.18 per tonne.

A.7.6 Mechanical Biological Treatment

Mechanical-biological treatment (MBT) facilities can be configured in various different ways. Generally, outputs include more than one of the following:

- Recyclable materials;
- A stabilised biowaste, which may find use as a 'compost like output', but which may have to be landfilled;
- A fraction to be sent to landfill;
- A refuse derived fuel.

In the UK procurement and regulatory context, the capital costs for MBT facilities have been difficult to estimate as the regulatory environment has been so fluid.

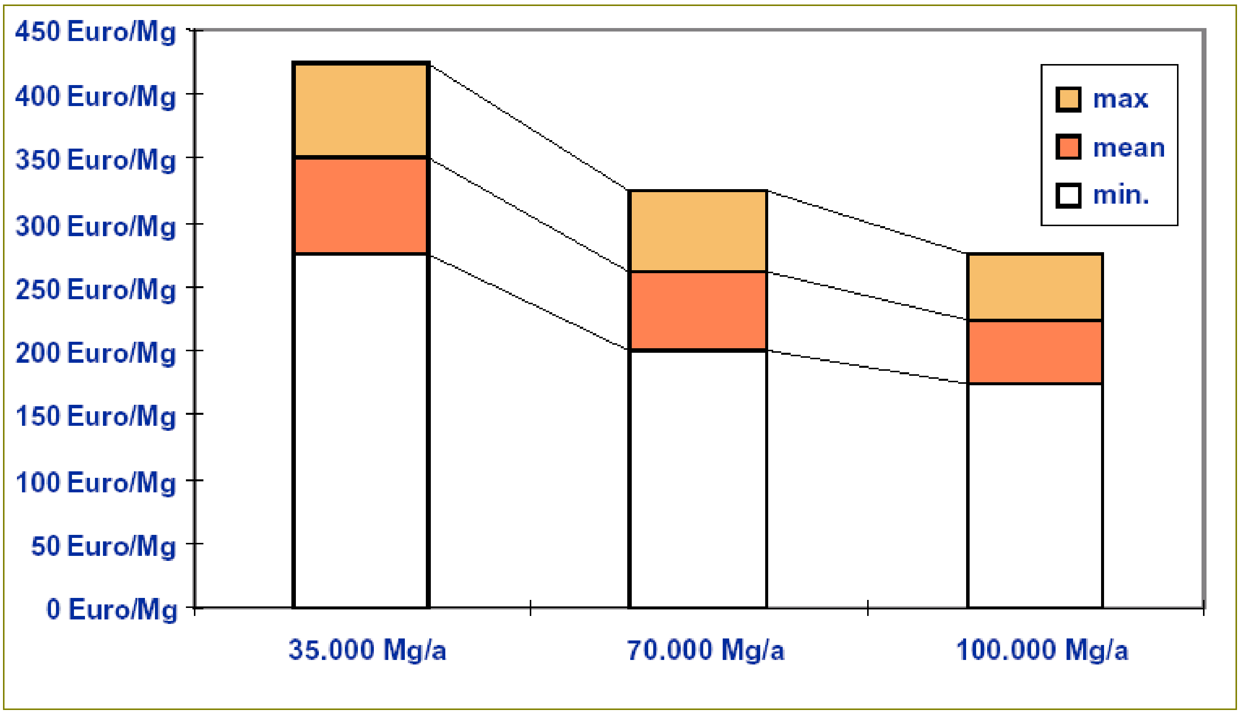
An Annex to Waste Strategy for England 2007 gives capital costs for MBT facilities which are configured to produce an RDF with the RDF, presumably, combusted in a dedicated waste incinerator. The capital costs quoted were as follows:

- 50,000 tonnes Capex £29.4 million (£588 per tonne);
- 100,000 tonnes Capex £44.4 million (£444 per tonne); and
- 200,000 tonnes Capex £67.1 million (£335 per tonne).

For some facilities of this nature, particularly lower capital cost MBT processes based on aerobic treatment, 60,000 tonnes or so is believed to be a near-optimum scale from a *technical* (if not a project) perspective. Figure 8 showing the results of analysis of tenders for German plants over one year, suggests that economies of scale may already be limited at a capacity of 100,000 tonnes.

It should also be noted that this review covered a range of plant sizes with 60,000 tonne facilities falling in the middle of this range. Figure 8 shows that this type of capacity is far from unusual for MBT plants. Indeed, the average size for the German facilities listed is around 70,000 tonnes.

Figure 8: Range of Unit Capital Costs Reported in German Tenders for MBT Facilities



Our analysis focuses on an aerobic biodrying treatment process, with the resulting low biomass solid recovered fuel (SRF) sent for:

- Gasification, with a steam turbine in CHP mode;
- Gasification, with a gas engine in CHP mode; and
- Combustion, generating electricity only.

Consequently, one needs to understand the costs of a range of different pieces of equipment. These include:

- An aerobic biodrying facility preparing SRF;
- A gasifier, with a steam turbine in CHP mode;
- A gasifier with a gas engine in CHP mode; and
- A dedicated combustion facility generating electricity.

These are discussed in Sections A.7.6.1 to A.7.6.3. For each system, where more than one of the above technologies are used in one system, we have simply multiplied the quantity of material to be treated by the unit capital cost to understand the total capital costs for treating one tonne of waste in the overall process.

A.7.6.1 Aerobic Biodrying Facility Preparing SRF

In principle, the costs of this type of system will be different depending upon whether the SRF which is being prepared is to be of 'higher' or 'lower' quality with regard to biomass content. We have used figures for the capital cost of £200 per tonne, with operating costs of £17 per tonne before residue disposal. It should be noted that the reality is that both the capital costs and the costs of dealing with residues will depend upon the detailed configuration of the system and the specification to which SRF is being produced.

A.7.6.2 Gasifier with a Steam Turbine

It is very difficult to give any clear figures for gasification costs. There is no commercial experience with waste gasification in the UK other than for small amounts of clinical waste. There are some demonstration projects in construction, as well as some merchant facilities now being planned. Some of these merchant facilities claim low unit capital costs and the ability to operate their technology coupled to a gas engine. Such a configuration has proved technically difficult to deliver in a form which is reliable.

The quoted sources publicly available suggest enormous variation in the figures used for various analyses. For example, AEA quotes indicative gate fees in North London of £37 per tonne (excluding disposal of residues), but quoted operating costs alone of £20-55 per tonne in a report for the Environment Agency.

Fichtner make the point:

> *Reasonably accurate costs are only likely to come from real quotations to detailed specifications. Even costs obtained from tenders must be treated with a degree of caution since tender prices can vary dramatically from one supplier to the next even for almost identical technologies.*

Though Fichtner seem to making this point regarding the capital costs of gasification and pyrolysis in particular, the comment has more general applicability.

Our own view is that the tendency to generalize costs across 'landfill' and 'grate incineration', both relatively mature technologies in which some variation exists, but for which the associated cost variation is tolerably well understood, has led to a situation in which stakeholders have tried to generalize across technologies which are quite varied. It should not be expected that all gasification technologies, nor all MBT configurations, will cost the same. Different variants are patented processes which may exhibit quite significant variation in design, performance and cost.

The Annex to the English Waste Strategy gives the following figures for capital costs and gate fees:

- 30 ktpa - capex £21.7 million (£723 per tonne), gate fee £93.6 per tonne;
- 100 ktpa - capex £27.9 million, (£279 per tonne), gate fee £69.2 per tonne;
- 150 ktpa - capex £67.2 million, (£448 per tonne), gate fee £51.56 per tonne.

These figures seem somewhat strange, implying as they do a massive drop in unit capital costs as plants increase in scale from 30ktpa to 100ktpa, but then a significant increase in unit capital cost as the scale increases to 150ktpa. Indeed, it is rather difficult to understand how the gate fees would exhibit the suggested trend if the capital costs really were as the document states.

We have used a figure for capital costs which assumes the gasifier has a capital cost of £550 per tonne where a steam turbine is used and £600 per tonne where a gas engine is used (our assumption being that requirements in terms of gas clean up will be greater). These seem higher than many of the above figures, but in our view, most of the above figures are rather low.

Operating costs are no more straightforward to determine as the variation in the the above Defra figures indicates. We have used a figure of £25 per tonne for all configurations.

A.7.6.3 Gasifier with a Gas Engine

As per the above discussion, we have used a figure for capital costs which assumes the gasifier has a capital cost of £600 per tonne where a gas engine is used.

For operating costs, as previously stated, we use a figure of £25 per tonne. It is worth, however, emphasising the point that there are no working examples of a gasifier with a gas engine being used to treat residual waste in the UK, and therefore any assumptions about capital and operating costs must be considered speculative at best.

A.7.7 Autoclaving

Autoclaving (also know as Mechanical Heat Treatment (MHT)) is a term used to describe configurations of mechanical and thermal, including steam, based processes. The purpose of these processes is to separate a mixed waste stream into a number of component parts, to give further options for recycling, recovery and in some instances biological treatment. The processes also sanitise waste by destroying bacteria present, and reduce its moisture content.

MHT technologies have a limited track record. The most common system being promoted for the treatment of MSW using MHT is based around a thermal autoclave. Autoclaving has been long been used in hospitals to sterilise surgical equipment via the application of heat and pressure. The technology is also commonly used to sanitise clinical wastes and for certain rendering processes for animal wastes, prior to sending to landfill. However, application to MSW is a recent innovation and there is limited commercial experience with this feedstock material. [84]

Another type of MHT system is a non-pressurised heat treatment process, where waste is heated in a rotating kiln prior to mechanical separation.

Our analysis focuses on an autoclave treatment process, with plastics recovered for reprocessing, and with the resulting high biomass solid recovered fuel (SRF) sent for:

- Gasification, with a steam turbine in CHP mode;
- Gasification, with a gas engine in CHP mode, and
- Combustion, generating electricity only.

Consequently, one needs to understand the costs of a range of different pieces of equipment. These include:

- An autoclave;
- A gasifier, with a steam turbine in CHP mode;
- A gasifier with a gas engine in CHP mode; and
- A dedicated combustion facility generating electricity only.

The gasifier options have been discussed in sections A.7.6.2 and A.7.6.3, and the combustion facility in A.7.5.1. The autoclave costs are outlined below.

For each system, where more than one of the above technologies are used in one system, we have simply multiplied the quantity of material to be treated by the unit capital cost to understand the total capital costs for treating one tonne of waste in the overall process.

There is little published information on the cost of MHT facilities. A Defra report identifies that technology suppliers suggest between £25-£45 per tonne operating costs for the autoclave/separation components of the process. Capital costs are estimated at around £15 million for a 100,000 tpa facility, which equates to a capital cost of £150/tonne. The lifetime of the facilities are anticipated to be not less than 10 years and more usually 20 years. [85]

We feel these figures to be on the low side, and model on the basis of £270/tonne capital costs, and £13/tonne operating costs for the autoclave. We assume that for each tonne input, 325kg goes to a gasifier using a steam turbine, or a gas engine with capital costs of £550/tonne and £600/tonne respectively, and operating costs of £25/tonne. Therefore, in our modelling, these capex and opex figures are attributed to the MHT process in proportion to the 325kg/tonne output.

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[84] Defra (2007) Mechanical Heat Treatment of Municipal Solid Waste. Prepared by Enviros Consulting on behalf of Defra as part of the New Technologies Supporter Programme.

[85] Defra (2007) Mechanical Heat Treatment of Municipal Solid Waste. Prepared by Enviros Consulting on behalf of Defra as part of the New Technologies Supporter Programme

A.7.8 Materials Recovery Facilities

Obtaining reliable data on MRF costs, or gate fees, is complicated by a number of factors:

- The reluctance of operators to divulge information perceived as commercially sensitive;
- The different configurations of MRFs, whether fully co-mingled (single stream), or a 2 stream mixture of fibres and containers, and variations in these basic categories;
- The range of different types of MRFs, in terms of the source of their inputs (Household/Commercial & Industrial/Construction & Demolition);
- The age of the facility, as new facilities are becoming larger, more capital intensive, and with lower relative labour costs;
- The nature of the contract that the MRF operator has entered into, which might include an element of sharing in material revenues; and
- The value of materials recovered.

The approach taken from this study is to use illustrative capital and operational costs taken from WRAP's Single Stream MRF Cost Model, for two configurations at the largest annual capacity available in the model;[86]

- Single stream bagged with glass, with 85,000 tonnes annual capacity; and
- Single stream bagged without glass, with 85,000 tonnes annual capacity.

For both we assume an operational lifetime of 14 years, with a 15% cost of capital

Uplifting these figures by approximately 8% to convert them to 2009/2010 prices, and then subsequently attributing an uplift to reflect higher land and labour costs in London is felt to deliver a reasonable approximation of MRF costs in London.[87] These assumptions have been presented to MRF operators in and around London for verification.

A.7.8.1 Single Stream Bagged with Glass

For single stream bagged with glass, the model shows, for a facility of 85,000 tonnes capacity, a total equipment cost of £4.9 million, with a building cost of £1.7 million. This gives a total capital cost of £6.6 million, or £78/tonne. Inflated to 2009-10

[86] WRAP (2007) Materials Recovery Facilities Cost Model Single Stream, Entec Consulting Ltd., Canada, for WRAP, 26 January 2007.

[87] HM Treasury GDP Deflators webpage, available at http://www.hm-treasury.gov.uk/d/gdp_deflators.xls. Prices were uplifted from 2006-07 prices to 2009-10 prices. With index levels at 94.803 and 102.11 respectively, this represents an increase of 7.59%

prices, this is £84/tonne. To account for the higher cost of land in London, the capital cost is uplifted to £93/tonne.

Labour costs total £1.57million per annum, with total variable operating costs minus residue disposal and equipment maintenance at £450,000 per annum. Summing these cost components gives a total annual operating cost of £2.04 million, or £24/tonne. Inflated to 2009-10 prices, this is £26/tonne. To account for the higher cost of labour in London, the operating cost is uplifted to £29/tonne

We assume maintenance to be 5% of capital cost, at £4.20/tonne. Rejects to landfill are modelled at 15%. The calculated gate fee is £34/tonne.

A.7.8.2 Single Stream Bagged without Glass

For single stream bagged without glass, the model shows, for a facility of 85,000 tonnes capacity, a total equipment cost of £4.6 million, with a building cost of £1.8 million. The equipment cost is lower as there is no need for a glass line, and there is a reduced cost associated with the mixed container receiving line. However, the building costs are slightly higher, as the building is assumed to be larger, reflecting the lower density, and thus higher volume requirements of the throughput as glass is not processed. This gives a total capital cost of £6.4 million, or £75/tonne. Inflated to 2009-10 prices, this is £81/tonne. To account for the higher cost of land in London, the capital cost is uplifted to £90/tonne.

Labour costs total £1.7million per annum. These are higher due to an increased number of fibre sorters and container sorters. Total variable operating costs minus residue disposal and equipment maintenance are £478,000 per annum. These are slightly higher than for a glass accepting MRF due to increased baling wire requirements, although savings are assumed in terms of requirements for spare parts. Summing these cost components gives a total annual operating cost of £2.2 million, or £26/tonne. Inflated to 2009-10 prices, this is £28/tonne. To account for the higher cost of labour in London, the operating cost is uplifted to £32/tonne

We assume maintenance to be 5% of capital cost, at £4.05/tonne. Rejects to landfill are modelled at 13%. The calculated gate fee is £27/tonne.

A.7.8.3 Single Stream Bagged with Glass but without Paper

We also model the cost of sending a composition that includes glass, but excludes paper. The capital and operating costs are the same as for Single Stream Bagged with Glass, but the revenues are amended accordingly. The calculated gate fee is £33/tonne.

A.7.8.4 Composition

Composition data from WDF is shown in Table 38. For paper/card, the assumed split is 25% card, 75% paper.

Table 38: Composition of MRF Inputs

MRF Type	Paper/Card	Cans	Plastic	Glass	Total
Bagged with Glass	67%	5%	5%	23%	100%
Bagged without Glass	88%	6%	6%		100%
Bagged with Glass no Paper	n/a	15%	15%	70%	100%

A.7.8.5 Revenue Assumptions

Table 23 above shows assumed values for clean recyclate. To account for increased contamination of MRF outputs, these values are lowered by 32% for the MRF without glass, and 35% where glass is included, and 32% where there is glass but no paper.

A.8.0 Environmental and Technology Performance Assumptions

This section describes the underlying assumptions used in the analysis. We start by considering some framing assumptions common to all treatment technologies under consideration within this study.

A.8.1 Energy Generation and Use

A.8.1.1 Electricity

The carbon intensity of an energy source is the quantity of GHG emissions associated with generating the energy. Where emissions are avoided as a result of generating energy from waste, or where energy is used by a process, assumptions regarding which source of energy is considered to have been avoided, or utilised, are important in determining the overall GHG benefit associated with power generation.

With a growing demand for electricity (unfortunately, most would add), where new facilities are being built to generate energy, and where these operate more or less continuously, it seems reasonable to argue that the avoided source of generation is the source, or mix of sources, deemed most likely to have been built in the absence of capacity arising through energy from waste infrastructure. Across the UK at present, this might be a mix of sources, including gas, renewables such as wind, nuclear, and some coal, though possibly equipped with some element of carbon capture and storage.

Defra has suggested that for the purposes of policy evaluation, the marginal source of electricity should be taken to be CCGT gas plant, representing the trend in terms of recently commissioned power generation technology.[88] The carbon intensity figure used within the current analysis is based around electricity generated by a modern CCGT power station. We have calculated the carbon intensity using an assumed efficiency of generation of 55% (the levels achieved by modern power stations today), and assumed natural gas has a calorific value of 39 MJ/m^3.[89] The carbon intensity associated with electricity generation in this form is 0.330 kg CO_2 equivalent per kWh from the process itself with some 0.057 kg CO_2 equivalent per kWh from the pre-combustion process, giving a total of 0.387 kg CO_2 equivalent per kWh.

A.8.1.2 Heat

The carbon intensity of displaced heat generation was estimated from the calorific value of natural gas of 39 MJ/m^3. Emissions are 0.258 kg CO_2 equivalent per kWh of heat energy generated, taking into account the efficiency of heat generation

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[88] Defra (2006) Greenhouse Gas Policy Evaluation and Appraisal in Government Departments, April 2006

[89] CV of natural gas: DECC (2009) Digest of UK Energy Statistics: National Statistics, 2009

(assumed to be 90%) and the pre-combustion emissions as was the case with the electricity emissions figure.

A.8.1.3 Diesel

We have used a figure of 2.63 kg CO_2 equivalent per litre of diesel (including 0.46 kg CO_2 equivalent pre-combustion emissions).[90]

A.8.2 Dry Materials Recycling

Impacts associated with the following materials are considered within this section:

- Paper and card;
- Dense plastic;
- Glass;
- Steel;
- Aluminium;
- Wood;
- Textiles;
- WEEE; and
- Furniture.

Figures are given in terms of avoided CO_2 equivalent emissions per tonne of material recycled. We use data taken from a range of recent studies - including work undertaken within the UK, Europe and the US - as a basis for modelling the environmental impacts associated with recycling.

The principal UK-based sources considered within our analysis are:

- ERM (2006a) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006;
- ERM (2006b) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006; and
- WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006.

The WRAP analysis reviewed a number of studies, incorporating results from the UK, Europe and the US.

The relevant European and American sources are those by:

[90] Diesel emissions: Defra (2005) Guidelines for Company Reporting on Greenhouse Gas Emissions, July 2005

- AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001;
- USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002.[91]

Of these, the AEA report also reviewed data from a variety of European studies. Data from these studies was included within the WRAP review cited above for some of the materials considered within the current analysis.

Where possible, the information provided by the above sources has also been cross referenced against updated industry specific data provided by such bodies as the European Aluminium Association.

This Section discusses the range of values for each of the material provided by the various literature sources and confirms the value chosen for the current analysis. A summary of the values used is provided at the end of the Section.

A.8.2.1 Paper

The international review of recycling studies undertaken on behalf of WRAP looked at a range of life-cycle scenarios. Their analysis also evaluated impacts associated with the method of disposal in the situation where paper is produced from virgin materials.[92] Table 39 summarises the results from the study with respect to paper and card.

Table 39: Emissions Savings – Paper Recycling Compared to Disposal (WRAP)

Material	Disposal method	No. of scenarios considered	Average saving across scenarios (tonnes CO_2eq)	Range of savings across scenarios (tonnes CO_2eq)
Paper (all types)	Incineration	35	0.73	$-0.1 < x < 4.6$
Paper (all types)	Landfill	13	1.34	$-1.1 < x < 3.4$
Notes: 1. Negative numbers here represent scenarios that lead to a net contribution to climate change as a result of recycling that material				

WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006

[91] A revised version of this report was subsequently published in 2006 and incorporated data on new materials and an updated energy mix for the US. However the major part of the analysis did not significantly change from the 2002 version.

[92] WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report for WRAP, May 2006

Their analysis indicated that the most sensitive assumptions surrounded the type of energy used for both the virgin paper and recycled paper production processes. The report indicated that producing paper from recycled input results in an energy reduction in comparison to production from virgin materials. There is however a variation in the type of energy used by the different processes. In cases where wood is used to generate thermal energy – which is the case for some virgin paper production processes – the emissions associated with that energy use may be reduced, even though the actual energy requirement is greater. The energy requirements are dependent on the type of paper being processed, and in particularly whether it is electrical or thermal energy that is required. Paper produced using thermo-mechanical pulp or chemical-thermo-mechanical pulp (e.g. newsprint) usually requires a supply of electricity, and this is less frequently produced from wood fuel. In contrast paper produced from craft pulp requires thermal energy, and this was much more frequently provided from wood fuel. The WRAP study found that in all cases where incineration was favoured over recycling some form of wood energy was assumed to be used within the virgin paper production process (but not within the recycled paper production process).

The AEA (2001) report reviewed the estimates associated with a number of studies. For paper, these are shown in Table 40.

Table 40: Life-cycle Emissions for Paper Production (AEA)

Paper Type	Source	Production emissions (kg CO_2eq / t paper)	
		Virgin Materials	Recycled Materials
Newsprint	Swedish study	1,755	849
Newsprint	US study	2,222	1,535
Newsprint	BUWAL database		291 (68% recycled)
Kraft paper unbleached	BUWAL database	1,080	633
(Swiss Kraft) Graphic paper	BUWAL database	436 (uncoated) 730 (coated)	586 with de-inking 380 without de-inking
Corrugated board	BUWAL database	644 (25% recycled)	522-556

Source: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001

The report carried forward the figure from the Swedish study, which converted energy use into emissions using EU average power mix. The study assumed that 1 tonne of recycled paper could produce 700kg of recycled newsprint. Actually, this estimate may be quite low for newsprint, and is more representative of other paper grades, but since the study was looking at other forms of paper and card (using 'paper' as one category), 70% could be a reasonable figure to use. As such, the estimated GHG savings associated with recycling paper were 0.7 x 906 kg CO_2 equivalent per tonne, or 0.634 tonnes CO_2 equivalent per tonne of paper.

These savings are much lower than are estimated by the USEPA (2002). However, the USEPA modelling included some quite sophisticated modelling of the US forest sector, and the implications of not harvesting forests:[93]

> *'When paper and wood products are recycled or source reduced, trees that would otherwise be harvested are left standing. In the short term, this reduction in harvesting results in a larger quantity of carbon remaining sequestered, because the standing trees continue to store carbon, whereas paper and wood product manufacture and use tends to release carbon. In the long term, some of the short-term benefits disappear as market forces result in less planting of new managed forests than would otherwise occur, so that there is comparatively less forest acreage in trees that are growing rapidly (and thus sequestering carbon rapidly).*
>
> *Considering the effect of forest carbon sequestration on U.S. net GHG emissions, it was clear that a thorough examination was warranted for this study. The complexity and long time frame of carbon sequestration in forests, coupled with the importance of market dynamics that determine land use, dictated the use of best available models.'*

Close inspection shows that these are extremely important in the modelling outcomes, albeit (as the study itself admits) subject to considerable uncertainty. Importantly, the study claims that these benefits are potentially transferable to other countries: [93]

> *'Although the goal of this analysis is to estimate the impact of paper recycling and source reduction on GHG emissions in the United States, the actual effects would occur in Canada and other countries as well.'*

The caveats under which these sequestration effects might be deemed a) accurate and b) directly transferable are quite numerous (and the reader is directed to the study for more detailed discussion). Suffice to say that the effect is potentially important, being far greater than the total savings estimated by the AEA report.

The enormous significance of the sequestration effect in the total outcomes can be appreciated by reference to Table 41. These figures take into account the loss rates of material in the recovery process and in the production process. It can be seen that for newspaper, the recycled input credit – which represents GHGs saved through using recovered fibre as opposed to virgin materials - is close to the estimate used by AEA. The enormous difference in the reported outcomes is entirely associated with the sequestration effects modelled in the US study. It is also noteworthy that the relative performance of the different materials recovered in respect of the credits for

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[93] USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002

using recycled inputs tends to reflect what was suggested in the studies reviewed by AEA. For example, the credit for corrugated cardboard is less than the GHGs emitted in using virgin materials, whilst the AEA review suggests a much reduced credit relative to virgin material production.

Table 41: GHG Emissions for Recycling, $MTCO_2eq$ per tonne of Paper (USEPA)

(a)	(b)	(c)	(d)	(e)	(f)
Material	**Recycled Input Credit*: Process Energy**	**Recycled Input Credit*: Transportation Energy**	**Recycled Input Credit*: Process Non- Energy**	**Forest Carbon Sequestration**	(f = b + c + d + e) **GHG Reductions From Using Recycled Inputs**
Corrugated Cardboard	0.147	-0.037	0.000	-2.677	-2.603
Magazines/Third-class Mail	0.000	0.000	0.000	-2.677	-2.713
Newspaper	-0.770	-0.037	0.000	-2.677	-3.483
Office Paper	0.220	0.000	0.000	-2.677	-2.493
Phonebooks	-0.660	0.000	0.000	-2.677	-3.337
Textbooks	-0.037	0.000	0.000	-2.677	-2.750
Dimensional Lumber	0.073	0.000	0.000	-2.530	-2.457
Medium-density Fiberboard	0.037	0.000	0.000	-2.530	-2.457

Source: USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002

ERM give a figure of 496 kg CO_2 equivalent avoided per tonne of material recycled.[94] A more recent study takes figures from the Swiss Ecoinvent database, giving values maximum and minimum figures of 0.62 tonnes and 0.28 tonnes respectively, with the difference attributable to de-inking processes and the grade of paper being produced (the minimum value assumes the paper recovered is of a low grade).[95]

We have used the upper value given by the latter ERM study of 0.62 tonnes CO_2 equivalent per tonne of paper recycled in our analysis. This is close to the value taken forward in the previously cited AEA study, and is also consistent with the average WRAP value if the contribution given by the avoided landfill emissions is removed.

A.8.2.2 Glass

Glass commonly constitutes around 25-35% by weight of dry recyclables collected. Tonne for tonne savings of greenhouse gases resulting from recycling glass therefore have a significant impact with respect to the overall emissions associated with the waste management system of an authority.

The values presented in Table 42 are taken from the WRAP review.

Table 42: Emissions Savings – Glass Recycling Compared to Disposal (WRAP)

Material	Disposal method	No. of scenarios considered	Average savings across scenarios (tonnes CO_2eq)	Range of savings across scenarios (tonnes CO_2eq)
Glass (closed loop)	Incineration	9	0.45*	$0.0 < x < 2.1$
Glass (closed loop)	Landfill	16	0.44	$0.3 < x < 1.1$
Glass (open loop)	Landfill	5	0.01	$0.0 < x < 0.1$
Notes 1. The study did not find any scenarios that compared open loop recycling with incineration 2. Negative numbers here represent scenarios that lead to a net contribution to climate change as a result of recycling that material				

WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006

[94] ERM (2006) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006

[95] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006

WRAP's review found that:[96]

> *...the assumptions that were found to have the highest influence on LCA outcomes were those related to the interdependency of the glass waste handling system on the energy system of the surrounding technosphere, including:*
>
> - *the type of energy used for manufacture of primary glass;*
> - *the type of energy used for manufacture of secondary glass from recycled cullet;*
> - *the type of recycling process applied (closed loop recycling appeared to be preferable to open loop recycling processes).*

As part of their study, WRAP reviewed a report by Enviros undertaken on behalf of British Glass. This attributed benefits of 0.314 tonnes of CO_2 equivalent per tonne of material recycled in closed loop processes where the avoided disposal route was landfill.[97] The study also considered the impact of reprocessing glass in overseas facilities; a value of 0.290 tonnes CO_2 equivalent per tonne of glass emerged from this scenario.

The AEA report used the EA / Chem Systems life cycle inventory. From this data, it was estimated that the carbon dioxide savings through the use of an additional tonne of cullet were 301 kg CO_2. 1,049 tonnes of raw cullet are needed for 1000 tonnes of processed cullet, giving a net GHG savings of 0.287 tonnes CO_2 equivalent per tonne of recycled cullet. This is almost identical to the figure reported by the USEPA (0.28 tonnes CO_2 equivalent per tonne).

We have used the closed loop value from the Enviros study, assuming that the glass is processed within the UK. This is slightly higher than the net savings attributed by AEA and USEPA, but lower than the average WRAP value.

A.8.2.3 Steel

Data from the WRAP study is presented in Table 43.

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[96] WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006

[97] Enviros (2003) Glass Recycling – Life Cycle Carbon Dioxide Emissions, internal report for the British Glass Public Affairs Committee

Table 43: Emissions Savings – Steel Recycling Compared to Disposal (WRAP)

Material	Disposal method	No. of scenarios considered	Average saving across scenarios (tonnes CO_2eq)	Range of savings across scenarios (tonnes CO_2eq)
Steel	Incineration	11	-0.90	$-0.1 < x < 3.1$
Steel	Landfill	8	-1.34	$0.0 < x < 3.0$
Notes 1. Negative numbers here represent scenarios that lead to a net contribution to climate change as a result of recycling that material				

WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006

The study found that the assumptions which had the highest influence on the results were those related to the interdependency of the steel waste handling system with the energy system of the surrounding technosphere – particularly with regard to the type of energy used within the primary and recycled scrap manufacturing systems. The study also cited assumptions regarding the effectiveness of steel reclamation from incineration processes as a further potential source of variation between studies.

ERM give a figure of 0.43 tonnes CO_2 equivalent per tonne of material recycled,[98] while a later report by the same company gives minimum and maximum figures of 0.58 tonnes CO_2 equivalent and 0.83 tonnes CO_2 equivalent, respectively, albeit reportedly using the same database as in the previous study.[99]

The AEA report used the datasets from BUWAL 250 for production of tin plate from raw materials and from non-detinned scrap. This data includes all emissions associated with transport of materials, energy used in processes etc. It was assumed that 0.84 tonnes of tinplate were manufactured from 1 tonne of scrap. This gave a figure of 1.521 tonnes CO_2 equivalent savings per tonne of steel collected for recycling. This figure is close to that reported in the USEPA report, which is slightly higher at 1.79 tonnes saved. The IWM2 model gives a figure of 1.75 tonnes CO_2 equivalent saved per tonne steel recycled.

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[98] ERM (2006) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006

[99] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006

We have used the WRAP figure of 1.34 tonnes CO_2 per tonne of steel recycled for the current analysis. This value is marginally higher than the mean of the other studies previously cited.

A.8.2.4 Aluminium

Table 44 presents data from the WRAP study.

Table 44: Emissions Savings – Aluminium Recycling Compared to Disposal (WRAP)

Material	Disposal method	No. of scenarios considered	Average savings across scenarios (tonnes CO_2eq)	Range across scenarios (tonnes CO_2eq)
Aluminium	Incineration	10	6.92	$-2.9 < x < 15.1$
Aluminium	Landfill	6	6.33*	$-0.4 < x < 15.1$
Notes: 1, Negative numbers here represent scenarios that lead to a net contribution to climate change as a result of recycling that material 2. Excluding one outlier result (50.32 tonnes CO_2eq)				

WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006

Almost all studies reviewed by the WRAP analysis attributed a clear benefit from aluminium recycling with regard to climate change. One outlier scenario considered very poor recycling rates and compared this to an incineration process where a very high recovery for the extraction of aluminium from the slag was assumed. The outlier scenarios were not, however, regarded as either typical or representative.

Two ERM studies gave similar values - a range from 12.3 tonnes of CO_2 equivalent avoided to 13.1 tonnes of CO_2 equivalent avoided per tonne of aluminium, and a figure of 11.6 tonnes of CO_2 equivalent avoided per tonne of aluminium.[100]

The AEA report used the datasets from BUWAL 250 for production of aluminium ingots from raw material and from recycled aluminium, and for production of tin plate from raw materials and from non-detinned scrap have been drawn from the BUWAL 250 data set. This data includes all emissions associated with transport of materials, energy used in processes etc. For primary aluminium production, emissions of the potent greenhouse gas carbon tetrafluoride (CF_4), which has a global warming potential of 6,500, are included. Table 45 confirms the GHG emissions for the

[100] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006; ERM (2006) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006

production of virgin and recycling aluminium indicated within the AEA study. It is further assumed that 0.93 tonnes of aluminium are produced from 1 tonne of recycled cans, and 0.84 tonnes of tinplate from 1 tonne of scrap. This gives a net savings figure per tonne of aluminium recycled of 9.108 tonnes CO_2 equivalent.

Table 45: GHG Emissions for Production of Virgin and Recycled Aluminium (AEA)

Material	CO_2 (kg)	CF_4 (kg)	Total kg CO_2eq
1,000 kg aluminium ingot (virgin)	7,640	0.4	10,240
1,000 kg aluminium ingot (recycled)	403	0	403

Source: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001

The USEPA report gives a figure of 15.07 tonnes CO_2 equivalent per tonne of aluminium recycled. For aluminium, the USEPA and AEA data was incorporated into the dataset considered by the WRAP review.

Recent data produced by the European Aluminium Association (EEA) suggests the total global warming potential for ingot production in Europe to be 9,677 kg CO_2 equivalent per tonne of aluminium, whilst comparable emissions for producing ingot from recycled aluminium were given as 506 kg CO_2 equivalent.[101] This suggests avoided emissions of 9.17 tonnes CO_2 equivalent per tonne of aluminium recycled.

We have used the EEA value in the current analysis, which is slightly lower than an average obtained from the average ERM and WRAP values.

A.8.2.5 Plastics

Different studies split out plastics fractions in different ways. Some give values of plastics by polymer, others simply split out materials by whether or not they are rigid or films. Where one is examining only household waste, one can reasonably consider recycling of dense plastic, with less attention paid to plastic film. The same is not the case for commercial wastes. Here, the value of plastic film may be significant, and it becomes more important to understand the GHG benefits of film recycling.

Table 46 provides data from the WRAP review, for all types of plastic.

[101] European Aluminium Association (2008) Environmental Profile Report for the European Aluminium Industry: Life Cycle Inventory Data for Aluminium Production and Transformation Processes in Europe, April 2008

Table 46: Emissions Savings – Plastics Recycling Compared to Disposal (WRAP)

Material	Disposal method	No. of scenarios considered	Average across scenarios (tonnes CO_2eq)	Range across scenarios (tonnes CO_2eq)
Plastics (all types)	Incineration	29	-1.25	$-3.8 < x < 4.1$
Plastics (all types)	Landfill	15	-1.08	$-1.5 < x < 2.4$
Notes 1. WRAP's analysis compared a range of different plastics 2. Negative numbers here represent scenarios that lead to a net contribution to climate change as a result of recycling that material				

WRAP (2006) Environmental Benefits of Recycling: An International Review of Life cycle Comparisons for Key Materials in the UK Recycling Sector, Final Report to WRAP, May 2006

Results for scenarios comparing the recycling of plastic to the incineration of the same material were particularly variable, and the WRAP review identified a number of sensitive assumptions from the range of studies it had considered. Particularly important were the following:

- Whether washing or cleaning of the material was required - where this was the case incineration was suggested to be environmentally preferable in the majority of cases (as a result of the use of hot water);
- Whether the recycled material was assumed to substitute (on a tonne for tonne basis) virgin material of the same kind. In cases where the quality / grade of the recovered plastic implied a less favourable substitution ratio (worse than 1:1), the scenarios dealing with this issue demonstrated that a ratio of 1:0.5 was about the break-point at which recycling and incineration with energy recovery were environmentally equal.

WRAP's analysis found that results were less sensitive to the type of polymer that was recycled. Their study featured one dataset evaluating the benefits of recycling film plastic from farms which was assumed to require washing as one of the reprocessing stages. The benefits attributed to this were in the region of 1.60 – 2.60 tonnes CO_2 equivalent per tonne even including the washing step (the analysis did not confirm which of the values taken from the study were attributed to the film plastic as opposed to the container plastic). Beyond this, the review did not provide any data on the recovery of film plastic from commercial waste streams.

ERM attributed low and high values to recycling of dense plastic and plastic film.[102] In both cases, the low values represent the impacts associated with the use of plastic as plastic lumber, whilst the high value represents the effect of displacing granulate, PET in the case of dense plastic, and LDPE in the case of plastic film. The values for dense plastic range from -0.85 tonnes CO_2 equivalent saved (i.e. a net contribution to climate change) to +1.82 tonnes CO_2 equivalent saved per tonne of material. For plastic film, the figures range from -0.85 tonnes CO_2 equivalent saved (i.e. a net contribution to climate change) to +1.47 tonnes CO_2 equivalent saved per tonne of material.

ERM's earlier study gave figures for dense plastic of 2.324 tonnes CO_2 equivalent saved per tonne of material recycled, and for plastic film, 1.586 tonnes CO_2 equivalent saved per tonne of material recycled.[103] ERM do not indicate the source of the feedstock for the recycled film. The benefits of reprocessing clean packaging film from a commercial stream are likely to be greater than those associated with recycling food packaging from households, given that the energy used to wash the material will be significantly reduced for film obtained through the first of these streams.

In the AEA Report, data on the emissions associated with plastics production were taken from the BUWAL 250 LCA data set which is based on data from the Association of Plastics Manufacturers in Europe (APME), except for HDPE where data used was taken from the Chem Systems work for the UK Environment Agency. Data on recycling of HDPE plastic bottles into flakes which are then extruded into pellets which can substitute for virgin material is available for a plant in the UK, and gives a value of 341 kg CO_2 per tonne of recyclate due to a much lower energy demand. Similarly data on PET bottle recycling to produce PET flakes at a Swiss plant gives a value of 114 kg CO_2 per tonne of flakes due to a low energy demand. AEA's life cycle CO_2 emissions associated with the production of different types of plastics are given in Table 47.

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[102] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, December 2006

[103] ERM (2006) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006

Table 47: Avoided Emissions Associated with Recycling Plastics (AEA)

Plastic type	Emissions in kg CO_2eq / tonne material			
	EU virgin	EU recycled	US virgin	US recycled
PE granules (general)	2,200			
HDPE granules	1,000	341	700	280
LDPE granules	2,320		890	330
LDPE granules	1,910			
PVC powder	1,940			
PET granules	2,200	114	1,160	450
PP granules	1,800			

Source: AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001

The AEA study used figures for HDPE and PET of savings of 0.53 tonnes CO_2 equivalent per tonne material recycled, and 1.8 tonnes CO_2 equivalent per tonne material recycled.

Updated information published by the APME suggests that typical emissions associated with HDPE manufacture are higher than those given by Chem Systems, implying that the benefits associated with recycling this polymer are greater than those cited by the AEA study.[104] The APME suggest typical emissions from a HDPE production process to be 1.9 tonne CO_2 equivalent per tonne of HDPE produced. Other figures are comparable to those presented by AEA.

The USEPA study also gives quite different figures for HDPE compared to the AEA study, as is shown in Table 48.

[104] The most recent updates were made in 2005. See http://www.plasticseurope.org

Table 48: Avoided Emissions Associated with Recycling Plastics (USEPA)

Material	Avoided emissions, tonne CO_2eq / tonne of material
HDPE	1.40
LDPE	1.71
PET	1.55

Source: USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002

For the current study, an average value for benefits associated with recycling dense plastic was derived - 1.40 tonnes CO_2 equivalent per tonne of material. The average is based on the values obtained for HDPE, LDPE and PET from the AEA, ERM and USEPA studies. For film plastic, we use the upper value given by the ERM study of 1.47 tonnes CO_2 equivalent per tonne of film (which assumes a closed loop recycling process).

A.8.2.6 Textiles

The WRAP review did not consider textiles recycling. AEA considered the process of recycling textile fibers into wool and acrylic garments. Energy savings result from the avoidance of raw wool scouring, the removal of contaminants and dyeing (less estimated energy usage from rag pulling). Overall CO_2 savings were calculated at 3.031 tonnes per tonne.[105]

ERM (2006a) reported figures of 0.93 to 1.75 tonnes CO_2 equivalent saved per tonne material recycled.[106] Figures at the lower end of the spectrum refer to the recycling of poor quality material into rags or fillers. Higher figures refer to combinations of synthetic and natural fibres therefore cannot be classified as 'maximum' values, which is how they are reported. ERM (2006b) report a figure of 7.869 tonnes CO_2 equivalent saved per tonne material recycled, suggesting that the 'maximum' figure in ERM (2006a) is underestimated.[107]

Data incorporated within the WRATE model suggests emissions benefits associated with the recycling of donated textiles to be 4.29 tonnes of CO_2 equivalent per tonne of

[105] Based on Energy Efficiency Office Best Practice Programme, Good Practice Case Study 181, 'A Novel Use for Recycled Textile Fibres', (undated) ETSU, Oxfordshire, UK

[106] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, Final Report for Defra, December 2006

[107] ERM (2006) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006

textiles, based on information supplied by Oxfam and WasteSavers. The model assumes that 70% of the clothing donated is resold, with 3% being rejected (subsequently landfilled) and a further 27% recycled into rags. Impacts are calculated on the basis of a UK specific mixture of textiles. No information is provided on the source of emissions reductions data.

It is clear that the benefits associated with recycling textiles vary enormously depending on the type of fibres and the end use of the recovered material. We have used the data provided by WRATE as the central assumption for the benefits associated with recycling textiles in our analysis, as this used a UK specific mix of materials to calculate the benefits.

A.8.2.7 Wood

There is a lack of robust data with regard to the benefits attributable to wood recycling, as was acknowledged in WRAP's international review of life cycle studies associated with recycling materials.

We use the value given by ERM 0.001 tonne of CO_2 equivalent per tonne of wood recycled. Following the same approach as for paper and card, we also include, in the spirit of sensitivity analysis, the non-fossil carbon emissions associated with carbon sequestration. These are attributed as 2.53 tonnes of CO_2 equivalent per wood recycled, as given by USEPA.[108]

A.8.2.8 WEEE

The benefits from the recycling of WEEE are estimated from the avoided emissions values for steel, aluminium and plastics, applied to the assumed composition of WEEE. The composition data is derived from a survey of recyclable WEEE undertaken in London, as shown in Table 49. This gives benefits of 1.078 tonnes CO_2 equivalent per tonne of WEEE.

Table 49: Composition of Recyclable WEEE in London, 2006

Steel	Aluminium	Plastics
53%	2%	16%
Notes: 1. The remaining proportion of the material is assumed non-recyclable.		

Source: Axion Recycling (2006) WEEE Flows in London: An Analysis of Waste Electrical and Electronic Equipment within the M25 from Domestic and Business Sectors, Report for the Environment Agency, September 2006

[108] USEPA (2002) Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, EPA530-R-02-006, May 2002

A.8.2.9 Furniture

Very little data quantitative data exists on the composition of furniture waste in the UK, or on the benefits that can be attributed to the recycling or reuse of furniture items. However the European Furniture Manufacturers Federation (UEA) has produced an estimated composition of European furniture wastes (excluding large WEEE items and white goods). These estimates are based on:

- European furniture manufacture statistics produced by the UEA and its partners;
- The expected lifetime of the furniture items, and
- An assumed replacement rate of 70%.

The UEA provides only limited data on the proportion of furniture waste that is recycled and reused across Europe.

The UEA provides only limited data on the proportion of furniture waste that is recycled and reused across Europe.

Table 50 indicates the estimated composition of furniture waste in Europe, using the UEA dataset together with that from a separate LCA study on mattresses.

Table 50: Estimated Composition of Furniture Waste (Excluding Large WEEE Items)

	Type of furniture				Proportion of total furniture waste, by material
	Household furniture[1]	Upholstered furniture	Mattresses[2]	Office furniture	
Wooden products	68%	66%		51%	62%
Metal products	6%	6%	36%	38%	13%
Plastics	9%	0%		1%	6%
Fittings	7%	0%		6%	6%
Foams	0%	2%	14%	0%	1%
Textile coverings	0%	15%	50%	0%	5%
Glass	3%	0%		0%	2%
Others	6%	12%		4%	6%
Proportion of total furniture waste, by type	*65%*	*14%*	*5%*	*16%*	*n/a*

Notes

1. Items such as dining tables / chairs, kitchen and bedroom fittings (excluding mattresses)
2. Data provided by Deliege et al as this is more typical for UK mattresses than the high level data supplied by the UEA

Sources: UEA (u.d.) Furniture Waste and its Treatment, available from http://www.ueanet.com/furniturewaste/; Deliege E, Nijdam D and Vlaanderen A (2008) European Ecolabel Bed Mattresses: LCA and Criteria Proposals, Final Report for the EC

In addition to the materials identified above that have been previously considered in other sections of the current analysis (i.e. wood, plastics, glass and textiles), flexible polyurethane foam and latex foam rubber can also be recovered as manufacturing scrap or recovered from post consumer use as bonded carpet cushion (although no life cycle inventory data is available to quantify potential emissions reductions).[109]

[109] See http://www.pfa.org/intouch/new_pdf/lr_IntouchV.8.pdf

The composition analysis indicates that where biogenic CO_2 emissions are excluded from the analysis, the most significant benefits are likely to be attained from the recycling or re-use of office furniture, given its relatively high metal content. The UEA suggests that up to 70% of replaced office furniture in the EU is already being reused as second hand products in Europe and Africa.

If the avoided emissions values for wood, steel, glass, textiles and plastics are applied to the assumed composition of furniture, this gives estimated benefits for furniture recycling of 0.40 tonnes of CO_2 equivalent per tonne of furniture where biogenic CO_2 emissions are excluded from the analysis. Inclusion of the biogenic CO_2 impacts (through sequestration) results in an additional 1.56 tonnes CO_2 equivalent per tonne of furniture.

The benefit that can be attributed to the reuse of household furniture items are not necessarily greater than those that are assumed to occur from the recycling of the composite materials. A recently published analysis of furniture reuse schemes attempted to evaluate the environmental, social and financial benefits associated with such schemes. The study incorporated data derived from interviews with scheme participants, including those who donated furniture to the scheme as well as those who obtained furniture from it.[110] The interview data suggested that only 10% by weight of furniture sold through the scheme would have been bought in the absence of the scheme.

A.8.2.10 Summary of Values Used for all Materials

Table 51 provides a summary of the assumptions used within the current study along with the relevant literature sources.

[110] Alexander C and Smaje C (2008) Evaluating Third Sector Reuse Organisations in the UK: Case-Studies and Analysis of Furniture Reuse Schemes, Resources, Conservation and Recycling, 52, pp719–730

Table 51: Summary of Values Used and their Literature Sources

	Avoided emissions, t CO_2eq / t recycled material
Paper and card[1]	0.62
Dense plastic[2]	1.40
Film plastic[3]	1.47
Glass[4]	0.31
Steel[5]	1.34
Aluminium[6]	9.17
Textiles[7]	3.03
Wood[8]	0.001
WEEE[9]	1.08
Furniture[10]	0.40

Notes:

1. ERM 2006 (upper value)
2. Average of HDPE, LDPE and PET taken from ERM 2006a and 2006b, AEA 2001 and USEPA 2002
3. ERM 2006 (closed loop process)
4. Enviros 2003 (value for UK re-processing, assuming closed loop process)
5. WRAP 2006 (average landfill value)
6. European Aluminium Association 2008
7. AEA 2001
8. ERM 2006
9. Uses Axion WEEE composition and the above avoided emissions for each of the separate components
10. Uses UEA furniture composition and the above avoided emissions for each of the separate components

A.8.3 Materials Recycling Facilities

Our model considers the energy requirements of MRF facilities. We have based our assumptions in this regard on data provided within the Environment Agency's software tool WRATE.

The tool uses data provided by operating plant where available, although closer inspection of the facilities data provided within the model indicates that the electricity consumption is often estimated. Table 52 outlines the energy requirements for these facilities as indicated by WRATE.

The data provided in Table 52 suggests that the size of facility is important in defining energy requirements, indicating that larger facilities require less electricity per tonne of waste treated. The extent of the separation carried out by the plant is also important. As indicated in Table 52, we have used average values across all these facilities for modelling MRFs in London.

Table 52: MRF Energy Requirements

Facility Description[1]	Throughput (tonnes / year)	Electricity requirement (kWh / tonne)	Diesel requirement (kg / tonne)
'Small dirty (paper only)'	25,000	14	0.29
'Refuse Derived Fuel (RDF) for cement kiln (front end)'	60,000	30	0.33
'V screen, semi automated'	75,000	9	0.58
'Including infra red equipment'	50,000	15	0.93
Average	n/a	**17**	**0.53**

Notes:

1. Descriptions as expressed within WRATE

Source: WRATE

A.8.4 Location of Materials Reprocessing

Based on Defra's latest Packaging Recovery Note (PRN) and Packaging Export Recovery Note (PERN) data, reprocessing is assumed to take place broadly 50% in the UK, and 50% overseas, depending upon material type. For example, it is assumed that 90% of glass is reprocessed in the UK. As discussed in Section 8.2 of the Main Report, the destination and subsequent reprocessing of materials is relevant to carbon reporting mechanisms. Under IPCC guidance only carbon emitted, or saved, in the UK is counted in national emissions inventories. Hence the greater the reprocessing of materials in the UK, the greater the reported carbon benefit. The impact of this variable is tested within the sensitivity analysis in Section 9.6 of the Main Report.

A.8.5 Open Air Composting of Green Waste

Source-separated green waste can be treated by either IVC or open-air windrow composting facilities. Open-air windrow composting processes are those which occur in the open, usually in piles of triangular cross-section, these being turned periodically to introduce air into the process. In the UK, food waste cannot be treated in uncovered (open-air) facilities.

A.8.5.1 Climate Change Emissions to Air from Composting Process

The quantity of emissions to the atmosphere of any given gas from a given composting process is related to the degree to which the composting process is allowed to proceed towards a theoretical 'final' point at which all the carbon dissimilable in the composting process has been degraded.

In practice, different processes may facilitate more or less rapid degradation of the available biomass, so that over a given period of time, different processes may lead to differing levels of emissions. Other things being equal, however, and subject to proper management of the composting process, a longer retention time would be expected to lead to greater 'raw gas' (i.e. before biofiltration / scrubbing) emissions.

Depending upon the nature of the input materials and the market outlets, compost producers may seek to produce more or less mature products. The former is typically used in higher value horticultural applications; the latter is typically used on agricultural land. In terms of the overall emissions profile, it is important to understand whether fresh or mature composts generate more or less emissions overall.

This linkage – between end products, retention times and process emissions – has to be approached carefully. Fresh compost would produce fewer process emissions. However, the question arises as to what might happen once it is applied to land. Would emissions of nitrogenous gases continue (and be relatively more harmful because of the absence of any biofiltration)? Would the potential for methane generation be increased as a consequence of the less stable nature of the material, and the likelihood of the material being less well aerated?

To some degree, it could be argued that process emissions from compost, where they are less because of the lower retention time, are likely to be compensated for when the material is added to the soil. We probably do not have the evidence base to make this assumption; however the assumption is likely to be more applicable when considering open air composting than when considering in-vessel systems, with biofiltration. Arguably, the longer the period of treatment in a system that uses biofiltration, the less will be the difference in emissions from the 'short duration' and 'long duration' processes.

Within the modelling of composting processes carried out for this study, organic waste is assumed to contain carbon in the form of cellulose, lignin, protein, sugar / starch and fats. The proportion of these constituent types of carbon varies depending on the composition of the organic stream - green waste contains a greater proportion of cellulose and lignin whilst food waste contains more protein. Whilst sugar, starch and fat will degrade completely during aerobic digestion processes, lignin degrades much more slowly, such that only 15% is assumed to be degraded.

Table 53 outlines the key assumptions used in this study, which were developed in a previous study for WRAP by Eunomia.[111] The principal climate change impacts are associated with release of biogenic CO_2 emissions which are not reported in the majority of studies that use a life-cycle assessment approach to analyse the emissions of greenhouse gases.

Table 53: Assumptions for Windrow Composting

Parameter	Assumption
CH_4 emissions from process	0.05 kg / t input
N_2O emissions from process	0.117 kg / t input
Non-degraded carbon (retained in biomass)	30%
Electricity requirement	0 kWh / t input
Diesel use by process	1 l / t input
Mineralisation rate of readily available organic matter[3]	20%
Mineralisation rate of stable humus	1%
% of organic matter from compost becoming humus	25%

Notes:

1. No action of biofilter is assumed for windrow facilities.
2. These avoided emissions equate to the amount of energy required to produce fertiliser. The fertiliser requirement is assumed to be is displaced as a result of applying the compost to land.
3. The mineralisation rate is the rate at which carbon contained within the organic matter (or humus) is assumed to become atmospheric CO_2.

Nitrous oxide emissions are determined by temperature, ventilation, nitrogen content, the C/N relation, and other factors.[112] Maximum N_2O formation rates are observed if the supply of oxygen during decomposition is insufficient. This may occur, for

[111] Eunomia (2006) Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis, Final Report for WRAP, May 2006

[112] Hüther L (1999) Entwicklung Analytischer Methoden und Untersuchung von Einflußfaktoren auf Ammoniak-, Methan- und Distickstoffmonoxidemissionen aus Flüssig- und Festmist. Landbauforschung Völkenrode, Sonderheft 200, Braunschweig (FAL) 1999, 225 S.

example, if the partial pressure of oxygen in the rotting material drops to zero due to very high rates of biological activity.[113]

Aeration and the C:N ratio are believed to have an important effect on the nitrogen conversion processes. Where composting processes have included manures, intensive aeration in connection with low C-content has been shown to give rise to nitrite accumulation in slurry (up to 33% of the total nitrogen content) and incomplete ammonium oxidation. Low ventilation rates and sufficient carbon supply support the formation of nitrous oxide during nitrification and denitrification processes.

Gronauer et al suggest that around 12% of total nitrogen escapes from the material in the form of ammonia and that 0.15 kg N_2O per tonne waste would be emitted.[114] A further Swedish study assumed the nitrogen leakage to air was 7.5% of the nitrogen content of the feedstock.[115] Of this leakage, it was assumed 89% was emitted as NH_3, 9% as N_2O and 2% N_2. A study for the Danish EPA assumed that of the total amount of nitrogen lost as gaseous emission, 98 % was volatilised as NH_3, 0.5 % as N_2O and 1.5 % as N_2.[116] These are figures for raw gas (as opposed to gas which has been scrubbed).

Our model assumes that 10% of the total nitrogen content of the waste is released in some form as a result of the composting process. Of this 10%, we further assume that 10% is released as N_2O (with the remainder being released as NH_3 and N_2).

A.8.5.2 Benefits Associated with the Use of Compost

50% of the compost produced is assumed to be used in agriculture. Our model considers the following benefits associated with the use of compost in this way:[117]

- The displacement of alternative nutrient sources otherwise applied through the use of synthetic fertiliser, including the avoided energy use associated with this;

[113] Hellebrand H J (1998) Emission of Nitrous Oxide and Other Trace Gases During Composting of Grass and Green Waste (Emission von Lachgas und anderen Spurengasen während der Grüngutkompostierung). J. Agric. Engng Res. 69, S. 365-375; Zhou S, Zaeid H, and Van den Weghe H (1999) Kompostierung tierischer Exkremente - Einfluß der Sauerstoffkonzentration auf Reaktionskinetik und Emissionsverhalten, *Agrartechnische Forschung* 5, S. 2-10

[114] Gronauer A, Helm M, Schon H (1997) Verhafen und Konzepte der Bioabfallkompostierung – Vergleich – Bewertung – Empfehlungen, Bayerische Landesanstalt fur Landtechnik der TU Munchen-Weihenstephan

[115] Finnvenden G, Johansson J, Lind P and Moberg A (2000) Life Cycle Assessments of Energy from Solid Waste, Forskningsgruppen for Miljostrategiska Studier, FMS 137, August 2000

[116] Beck-Friis B (2001) Emissions of Ammonia, Nitrous Oxide and Methane during Composting of Organic Household Waste, Agraria 266, Doctoral Thesis, SLU, Sweden, cited in Baky A and Eriksson O (2003) Systems Analysis of Organic Waste Management in Denmark, Environmental Project No. 822, Copenhagen: Danish EPA

[117] For a detailed description of the methodology used to calculate these estimates see: Eunomia (2007) Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis, Appendices to the Main Report, Report for WRAP, May 2007

- N_2O emissions avoided as a result of the reduced application of nitrogenous fertiliser.

The remaining 50% of the compost is assumed to displace the use of peat in horticulture and hobby gardening applications. Here the avoided impacts are principally the slow release of CO_2 from the aerobic degradation of peat after its removal from the peat-land.[118]

A.8.6 In-Vessel Composting of Mixed Green and Food Wastes

Our model for the in vessel composting processes is largely the same as that previously described for the open air facilities. Emissions to air are, however, managed differently at IVC facilities and this has an impact on the greenhouse gas emissions from the process.

This study considers two types of aerobic digestion process for source-separated organic wastes – In-Vessel and Open Air Composting. Whilst garden waste can be treated by either process, food waste can only be treated through IVC facilities as a consequence of the Animal By-Product Regulations (ABPR) in the UK.

Emissions from IVC facilities vary depending on the composition of the organic material being treated by the facility. Food waste requires structural material (i.e. green waste) to be added to it prior to treatment within an IVC facility. Although it is possible to treat a feedstock of up to 70% food waste using IVC, the proportion is usually optimised at closer to 50-60%.[119] For the purposes of the current analysis, 50% of the material treated at IVC facilities is assumed to be food waste.

In in-vessel composting systems, the ammonia released from the composting process is usually treated in biofilters. In biofilters, the nitrogen in the ammonia is converted to, in varying proportions, N_2, NO and N_2O. The last of these is a potent greenhouse gas. The N_2O emissions from in-vessel composting systems are thus associated with:

- The process itself, through the release of nitrogenous gases to the atmosphere as a result of degradation processes – as has been previously described in Section A.8.5.1 with respect to emissions from windrow composting; and
- The workings of the biofilter, which are likely to include conversion of nitrogen in the form of ammonia to nitrogen in the form of N_2O.

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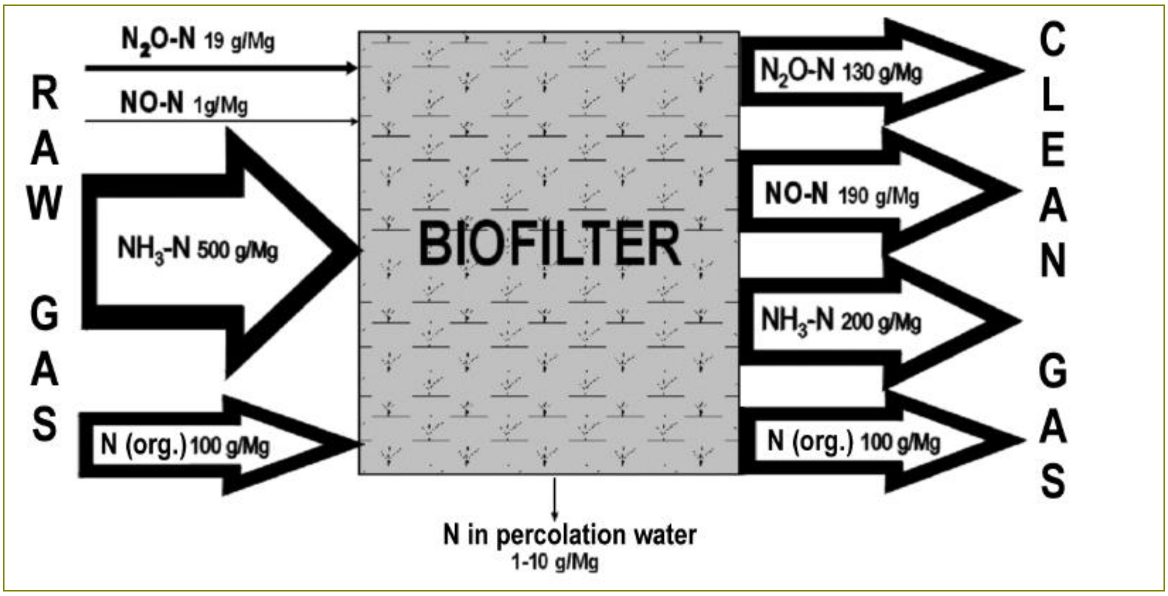
[118] This follows the methodology described in AEA Technology (2001) Waste Management Options and Climate Change: Final Report, European Commission: DG Environment, July 2001. Emissions associated with the aerobic decomposition of peat are modelled using the emissions factors provided by Cleary J, Roulet N T and Moore T R (2005) Greenhouse Gas Emissions from Canadian Peat Extraction, 1990-2000: A Life-cycle Analysis, Ambio, 34(6) pp456-461

[119] It should be noted, however, that this percentage will largely depend upon the level of sophistication of each particular IVC facility. These range from cheap, usually static clamp systems with 'temporary' polymer textile-type roofs, which have a high propensity to generate odours, to more expensive, housed or tunnel systems with generally lower likelihood of problematic odours

Estimates vary as to the proportion of N in NH_3 which follows the conversion pathway, but best estimates are that conversion efficiencies are of the order 25%.

One study looking at MBT processes suggests a mass balance as shown in Figure 9. This suggests that, as regards N, for every 500 g of N entering the biofilter as NH_3, an additional 111 g of N is emitted as N_2O. This would imply a conversion ratio of 22%. On the other hand, the above figure suggests a low overall rate of destruction of NH_3.

Figure 9: N-balance of a One-step Biofilter at the MBP Plant in Bassum, Germany



Trimborn et al conclude that independent from the level of NH_3 load in the raw gas ca. 29% of the transformed NH_3 is released as N_2O and ca. 9% to NO.[120]

Amlinger et al assume that:[121]

> *the continuous aerobic conditions in the biofilter supports the microbial oxidation of NH_4^+ to NO_2^-. High concentrations of NH_3 and NO_2^- can inhibit further oxidation to NO_3^- (Spector, 1998a,b). NO_2^- can be directly denitrified to NO and N_2O. It is likely that caused by a high NH_3^- concentration the microbial community in the biofilter is shifted in a way that deoxidising, denitrifying enzymatic activities become predominant.*

Literature suggests range of removal efficiencies for different compounds using biofilters. Vogt et al assumed a removal efficiency of 96% for NH_3, 50% for methane and 50% for total organic carbon. Omrani et al site removal efficiencies of 97-99% for

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[120] Trimborn M, Goldbach H, Clemens J, Cuhls C, Breeger A (2003) Endbericht zum DBU-Forschungsvorhaben Reduktion von klimawirksamen Spurengasenin der Abluft von Biofiltern auf Bioabfallbehandlungsanlagen (AZ: 15052)

[121] Amlinger F, Cuhls C and Peyr S (2008) Greenhouse Gas Emissions from Composting and Mechanical Biological Treatment, Waste Management and Research, 26, pp47-60

a biofilter using peat, soil and sand, whilst one of sawdust, clay and sand achieved 94% abatement.[122]

A more recent study by Amlinger and Cuhls presented data on the efficiency of CH_4 removal from biofilters as part of a wider study on emissions from composting processes.[121] Their data – which included measurements taken from currently operating facilities - suggested much lower efficiencies of removal than those indicated above. They concluded that biofilter system could be expected to reduce CH_4 concentrations by a maximum of 20%, with typical removal efficiencies in the order of 15%.

A specific recommendation, in relation to the operation of exhaust air treatment, so as to reduce N_2O emissions is – logically – to deploy acid scrubbers to eliminate NH_3 prior to treatment at the biofilter. This reduces the amount of NH_3 arriving at the biofilter, and hence, its conversion to N_2O. For example, ORA report 100% removal through the combined use of biofilter and scrubbing.[123] However, the use of a scrubber alongside the biofilter is not yet standard practice in UK IVC facilities.

In this study, we assume only a biofilter is used. Of the NH_3 generated in the first instance (through the composting process), 25% of the N in the NH_3 is converted to N_2O. This lies between the estimates derived from Doedens et al, and that of Trimborn et al. We further assume that 95% of the NH_3 and 20% of the CH_4 is removed by the biofilter, and that the remaining VOC emission not removed by the biofilter does not cause any environmental damage.

[122] Omrani G, Safa M and Ghaghazy L (2004) Utilization of Biofilter for Ammonia Elimination in Composting Plant, Pakistan Journal of Biological Sciences, 7, pp2009-2013

[123] ORA (2005) Development of a Dynamic Housed Windrow Composting System: Performance Testing and Review of Potential Use of End Products, Report for Canford Environmental, Dorset

Table 54: Assumptions for In-Vessel Composting

Parameter	Assumption
CH_4 emissions from process	0.816 kg / t input
N_2O emissions from process – garden waste	0.361 kg / t input
N_2O emissions from process – food waste	0.477 kg / t input
Non-degraded carbon (retained in microbial biomass)[2]	30%
Electricity requirement	40 kWh / t input
Diesel use by process	0.3 l / t input
Mineralisation rate of readily available organic matter[4]	20%
Mineralisation rate of stable humus	1%
% of organic matter from compost becoming humus	25%

Notes:

1. Assumes that a biofilter converts 95% of the available NH_3 to N_2O. 88% of the total nitrogen is assumed to be released as NH_3, whilst 10% is assumed to be released as N_2O without the action of the biofilter.
2. This carbon is assumed to be used for cell reproduction and growth of the microbiological organisms carrying out the degradation process.
3. These avoided emissions equate to the amount of energy required to produce fertiliser. The fertiliser requirement is assumed to be is displaced as a result of applying the output from AD to land.
4. The mineralisation rate is the rate at which carbon contained within the organic matter (or humus) is assumed to become atmospheric CO_2.

A.8.7 Anaerobic Digestion of Food Wastes

Emissions associated with the anaerobic digestion (AD) process itself are likely to occur at the following treatment phases:

1. During the digestion phase, described in Section A.8.7.1;
2. During the stabilisation process used to treat the solid residue, described in Section A.8.7.2.

The ultimate emissions to atmosphere and the emissions associated with the compensatory system are dependent in each case upon the utilisation of the biogas. Our model considers the following uses for the biogas produced by AD systems:

1. On-site combustion and energy generation using a gas engine generating both electricity and heat, described in Section A.8.7.4;

2. Biogas upgrading and its subsequent use as a vehicle fuel offsetting the use of diesel, described in Section A.8.7.5; and
3. Biogas upgrading and its subsequent injection into the gas grid offsetting the use of natural gas, described in Section A.8.7.6.

We also consider the environmental impacts associated with the use of the digestate. The benefits associated with this are described in Section A.8.7.7.

A.8.7.1 Emissions to Air from the Digestion Phase

The emissions from anaerobic digestion processes vary with input materials. They may also vary with the degree to which digesters approach a theoretical maximum biogas yield from the input materials. This theoretical yield depends upon the efficiency of the process, and the retention time within the digester (and for some processes, the difference between the hydraulic retention time and the solid retention time may be important).

CO_2 emissions resulting from the AD of source-separated organic waste are based on the carbon content of the input waste, assumed to 100% food waste for the purposes of this study.[124] The carbon content is calculated on the basis of the total organic content of the waste and its volatile solids (VS) content. A proportion of the total carbon content will be converted to biogenic CO_2 as a result of biogas combustion for energy generation (in whatever form this takes). Table 55 outlines key assumptions used within the modelling for this study.

Table 55: Assumptions Relating to AD Process and Generation of Biogas

Parameter	Assumption
Dry matter content of food waste	30%
Organic matter content of VS	93%
Carbon content of VS	45%
VS content of organic matter	45%
VS loss during digestion	70%
Methane content of biogas	60%

A.8.7.2 Emissions to Air from the Stabilisation Process

The residues can either be dewatered, creating a solid and a liquid fraction, or used directly on land as a slurry, sometimes using flocculants in the process. Whilst there

[124] These emissions are non-fossil in origin and therefore excluded in the majority of LCA analyses

may be some arguments for direct spreading, not least that of cost, it is considered better practice to stabilise the solid residues (following dewatering depending upon the materials and the process) through an aerobic stage so as to produce a compost.

We assume that the digestion process is followed by an aerobic treatment phase similar to that modelled for the composting of food waste at an in-vessel composting facility.

A.8.7.3 Energy Use

Unlike composting plant, AD facilities can potentially utilise some of the energy generated within the process to meet their requirements, although the literature suggest that some plant supplement this with electricity taken from the grid. AD facilities typically use both electricity and heat, although the extent to which both are required can vary considerably between different facilities.

Data from the UK biogas technology supplier Greenfinch suggests that between 3 and 28 kWh of electricity per tonne of input was required by the process, depending on the feedstock (although a lower electricity generation efficiency was indicated for this process).[125]

The significant heat requirement is confirmed by data provided from plant in Germany and the UK. Data from Bavaria suggests that a maximum two thirds of the heat produced by agricultural biogas plants can be used in some way under practical operating conditions, suggesting a heat requirement of 125 kWh per tonne of input.[126] Mass balance information provided by the UK operator Greenfinch suggests that up to 50% of the heat may be required (equivalent to 216 kWh of heat per tonne of input) although 130 kWh is more typical.[127]

The current study assumes that the AD process utilises 30 kWh of electricity and 118 kWh of heat per tonne of input to the process, equivalent to 10% of electricity generated, and 33% of the heat generation. The additional electricity requirement for upgrading is assumed to be 28 kWh of electricity (equivalent to 0.2 kWh per Nm^3 of biogas). These energy requirements are assumed to be supplied by the AD process itself – i.e. a smaller CHP unit is assumed to fuel the vehicle fuel and gas to grid

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[125] Greenfinch (2005) Mass and Energy Balance: Ryegrass and Pig Slurry Biogas Plant, Produced for DTI, September 2005; University of Glamorgan, The Wales Centre of Excellence for Anaerobic Digestion and the Sustainable Environment Research Centre (2007) Ludlow (Greenfinch) Trial Scale Kitchen Waste Treatment Plant; Biogen Greenfinch (2009) Renewable Energy Tariffs for Biogas, Presentation given at: Developing UK Biogas, Stoneleigh, June 2009

[126] Bachmaier H , Effenberger M and Gronauer A (2008) Agricultural Biogas Production: What About the Climate Balance?, 17th Annual Convention of Fachverband Biogas e.V, 15th-17th January 2008, Nuremberg

[127] Greenfinch (2005) Mass and Energy Balance: Ryegrass and Pig Slurry Biogas Plant, Produced for DTI, September 2005; University of Glamorgan, The Wales Centre of Excellence for Anaerobic Digestion and the Sustainable Environment Research Centre (2007) Ludlow (Greenfinch) Trial Scale Kitchen Waste Treatment Plant

applications. The energy content of the biogas that is assumed to be available for the upgrading process is therefore reduced accordingly.

No emissions are directly attributed to these energy requirements, as they are included within the total emissions attributed to the AD process.

There will be an additional electricity requirement associated with the gas upgrading process where the intention is to use the biogas as a vehicle fuel or to inject it into the gas grid. These demands may vary depending on which upgrading process is used. One study suggests that the upgrading steps require an input of electricity amounting to around 6% of the energy produced.[128] More recent data from both Germany and Sweden suggests that this is typically in the order of 0.2 kWh per Nm^3 of biogas.[129]

A.8.7.4 On-site Combustion of Biogas

Data from Greenfinch suggests a gross electrical generation efficiency of 30% together with a heat generation efficiency of more than 50%.[130] More recent data from the same technology producer suggests higher electrical generation efficiencies along with a lower efficiency of heat generation.[131] This more in line with data from Germany, which suggests a gross electrical generation efficiency of 40% together with a gross heat generation efficiency of 45%.[132]

Our study assumes a gross electrical generation efficiency of 37% and a gross heat generation efficiency of 45%. A proportion of the energy generated is assumed to be consumed by the process, as was discussed in Section A.8.7.3. 60% of the *net* heat generated is assumed to be utilised, again was discussed in Section A.8.7.3.

The principal climate change impact resulting from the energy generation phase relates to the biogenic CO_2 emissions from the combustion of the biogas in the gas engine.

Additional CH_4 emissions result from the non-combusted gas (known as the "slip") from gas engine. Data from five Bavarian agricultural biogas facilities suggests this

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[128] Baky A and Eriksson O (2003) Systems Analysis of Organic Waste Management in Denmark, Environmental Project No. 822, Copenhagen: Danish EPA.

[129] Urban W (2008) Methods and costs of the generation of natural gas substitutes from biomass – presentation of results of latest field research, 17th Annual Convention of Fachverband Biogas e.V, 15th-17th January 2008, Nuremberg

[130] Greenfinch (2005) Mass and Energy Balance: Ryegrass and Pig Slurry Biogas Plant, Produced for DTI, September 2005; University of Glamorgan, The Wales Centre of Excellence for Anaerobic Digestion and the Sustainable Environment Research Centre (2007) Ludlow (Greenfinch) Trial Scale Kitchen Waste Treatment Plant

[131] Biogen Greenfinch (2009) Renewable Energy Tariffs for Biogas, Presentation given at Developing UK Biogas, Stoneleigh, June 2009

[132] Scholwin F (2008) Present State of the Treatment of Biogas for Feeding into the Natural Gas Network in Germany, 17th Annual Convention of Fachverband Biogas e.V, 15th-17th January 2008, Nuremberg

"slip" results in emissions of 21-37 g CO_2 equivalent per kWh electricity.[133] N_2O emissions from the combustion of biogas are assumed to be negligible.

As previously indicated in this section, biogas combusted in a gas engine is assumed to result in the net generation of 265 kWh of electrical energy, and 97 kWh of heat (taking into account the utilisation factor). This results in avoided emissions of 128 kg of CO_2 equivalent where both the heat and electricity are exported.

A.8.7.5 Biogas Used a Vehicle Fuel

The utilisation of biogas as vehicle fuel uses the same engine and vehicle configuration as natural gas. There are reportedly more than 1 million natural gas vehicles in use across the world, which demonstrates that there is a receptive market to the use of biogas as vehicle fuel.[134]

The size of vehicle has a considerable impact on its emissions. Fuel consumption is far greater for heavier vehicles such as lorries and buses in comparison to cars, resulting in higher emissions per km. The sections that follow assume the upgraded biogas is used to fuel a fleet of heavy vehicles (such as buses or lorries) from a central re-fuelling point, such as already occurs in Sweden and France.

Both liquid fuel and gas operated heavy goods vehicles have seen considerable improvements in emissions over the past decade. There remains, however, considerable variation in performance between currently the available vehicles using either fuel.

For the purposes of our analysis, what is most important is the 'differential impact' of using CNG derived from biogas as opposed to conventional fuels. We are interested in the direct emissions and the 'displacement effect' associated with the use of the fuel to generate transport energy.

Gas quality demands are strict so as to provide a consistent high calorific gas containing low levels of contaminants and corrosive gases. The raw biogas produced in AD plants contains CH_4 and CO_2, smaller amounts of H_2S and NH_3, and trace amounts of H_2, N_2, CO, and O_2. Across different countries the minimum CH_4 content specification is between 95% and 97%, the permissible remainder being mostly CO_2. Typically also, the vapour content must be lower than 15 mg/Nm^3, the H_2S content should not exceed 100 mg/Nm^3 and the particle size is limited at 40 microns. The typical sequence for gas preparation is:[135]

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[133] Bachmaier H, Effenberger M and Gronauer A (2008) Agricultural Biogas Production: What About the Climate Balance?, 17th Annual Convention of Fachverband Biogas e.V, 15th-17th January 2008, Nuremberg

[134] IEA Bioenergy (u.d.) Biogas Upgrading and Utilisation, Task 24: Energy from Biological Conversion of Organic Waste

[135] Urban W (2008) Methods and costs of the generation of natural gas substitutes from biomass – presentation of results of latest field research, 17th Annual Convention of Fachverband Biogas e.V, 15th-17th January 2008, Nuremberg

A two step biogas desulphurisation process involving firstly a coarse desulphurisation method such as sulphide precipitation, followed by a fine desulphurisation step typically using activated charcoal filters;

- Gas drying;
- Gas compression;
- Removal of the CO_2 from the biogas (sometimes called CO_2 sequestration). This is most commonly achieved by scrubbing the gas with water under pressure, although other methods such as Pressure Swing Adsorption (PSA) are also used.

The CO_2 removal step results in the loss of some CH_4 from the biogas. These losses are typically in the order of 1% for the water scrubbing methods, although they can be as much as 3% if PSA is used. However some technology providers claim they can reduce this amount to close to zero.[136] The clean up process is also associated with an additional energy requirement, as was previously discussed in Section A.8.7.3.

The size of vehicle has a considerable impact on its emissions. Fuel consumption is far greater for heavier vehicles such as lorries and buses in comparison to cars, resulting in higher emissions per km. Both liquid fuel and gas operated vehicles have seen considerable improvements in emissions over the past decade. There remains, however, considerable variation in performance between currently the available vehicles using either fuel.

Natural gas has a lower carbon content than diesel, which results in a reduction in greenhouse gas emissions where these are calculated on the basis of the amount of energy consumed. Gas also has a high octane number, enabling a high compression ratio to be used, further reducing emissions.[137] However gas-fuelled vehicles emit more CH_4 than diesel vehicles. In addition, differences in fuel consumption between the two types of vehicles may reduce the benefits seen when emissions are calculated per km of travel. Data from France suggests the fuel consumption for the biogas buses is 65 m^3 per 100 km, whilst diesel buses use 50 litres per 100 km.[138] This gives fuel consumption for biogas buses of 23.4 MJ per km, whilst that of diesel vehicles is 17.9 MJ per km.

A study in Finland by VTT compared the emissions performance between a number of diesel and CNG buses, as part of a comprehensive national programme investigating bus emissions. Their analysis considered vehicles in prime condition manufactured during 2002-4, representative of Euro III technology. Data from that study with regard to the greenhouse gas emissions is presented in Table 56.

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[136] See http://www.haase-energietechnik.de

[137] See http://www.whatgreencar.com/cng.php

[138] Lille Metropole Communaute Urbaine (u.d.) Lille Metropolis, Urban Community: Biogas Buses Project, presentation to the US Department of Energy

Table 56: Emissions Data from Diesel and Gas Buses

Fuel and Vehicle Type		Emissions, g / km	
		CO2	CH4
Diesel	Euro III	1,150	0.01
	Euro III + OC	1,200	0.01
	Euro III + CRT	1,230	0.05
Gas	Euro III + LB CNG	1,230	0.60
	EEV LB CNG + OC	1,420	1.90
	EEV LM CNG + TW/OC	1,300	0.30
	EEV SM CNG + TW	1,050	1.20

Source: Nylund N, Erkkilä K, Lappi M and Ikonen M (2004) Transit Bus Emission Study: Comparison of Emissions from Diesel and Natural Gas Buses, VTT Processes, October 2004

The VTT dataset suggests that the use of gas to fuel buses does not necessarily result in a reduction in greenhouse gas emissions (although the emissions from the biogas fuelled vehicles are biogenic in origin, unlike those from the diesel fuelled vehicle). The VTT study suggests that CH_4 emissions accounted for around 2% of the total CO_2 equivalent emissions.

However, an earlier report detailing tailpipe emissions from Swedish buses suggests much lower emissions from gas buses, giving values of 524 g CO_2 per km for a natural gas bus and only 223 g / km for a biogas bus.[139] It is not clear whether the last figure includes the biogenic CO_2 emissions; if this is not the case, the CH_4 emission is far larger than anything seen within the VTT test data. The same study suggested emissions from a diesel bus of 1,053 g / km, which is more in line with the VTT dataset.

Other data produced by car manufacturers suggests that greenhouse the use of dual fuelled cars operating with a mixture of natural gas and diesel results in emissions reductions of 10-15% in comparison to comparable petrol fuelled vehicles.[140] Those vehicles fuelled solely by gas are anticipated to achieve greater emissions reductions. More recently published data associated with a planned trial for dual fuelled buses in

[139] Plombin C (2003) Biogas as Vehicle Fuel: A European Overview: Trendsetter Report No 2003:3, Stockholm

[140] See http://www.whatgreencar.com/cng.php

the UK indicates that carbon dioxide emissions reductions of 14% are expected as a result of the shift to the dual fuel vehicles.[141]

In this study, we assume that the use of bio-methane results in emissions reductions of 15% for the greenhouse gases (in terms of CO_2 equivalent emissions). We further assume that 2% of the CH_4 in the biogas is emitted during the upgrading process.

A.8.7.6 Biogas Injected to Grid

Cleaned and upgraded biogas (bio-methane) can also be injected into the gas distribution grid as a substitute for natural gas. Injection into the gas grid may require additional gas cleaning, although the extent to which this is necessary will depend on the requirements of the gas grid within each country.

At present there is only limited data associated with the environmental impacts of the gas to grid option. Depending upon the requirements of the grid, the following additional clean up steps may be required:

- Up to 4.6% propane is added to the gas to improve the Wobbe Index; and
- Oxygen may also need to be removed, further adding to the cost of the option. The environmental implications of this step (e.g. in terms of any additional energy requirement) are unclear.

Biogas injected into the gas network is assumed to offset emissions associated with a similar quantity of natural gas on a calorie for calorie basis. The plant is assumed to produce 2,114 MJ or 587 kWh of compressed biogas per tonne of food waste to the facility excluding the biogas required by the process for energy generation purposes.

Offset emissions for climate change impacts are based on the calorific value of natural gas, assumed to be 0.238 kg CO_2 per kWh (including emissions associated with extraction and transport).

We assume similar environmental impacts associated with the gas clean up process as was the case where the biogas is upgraded for use as a vehicle fuel. Our analysis is based on data from German facilities where additional clean up steps (such as propane addition and oxygen removal) are not required for grid injection as the grid accepts gas of a lower quality than that currently supplied to the UK grid. The environmental implications of the additional clean up steps likely to be required for grid injection within the UK are unclear as only limited data exists on these processes at present.

A.8.7.7 Use of Digestate

For the purposes of this study, we have assumed that the output material behaves in the same way as compost from the same feedstock produced through aerobic means. However, we have assumed a lower mass of compost produced of 300 kg per

[141] See http://www.letsrecycle.com/do/ecco.py/view_item?listid=37&listcatid=217&listitemid=53457&section=waste_management

tonne of waste input. The nutrient content is deemed to be the same as in the case of composting of kitchen and garden waste.

The benefits associated with the use of the composted digestate are calculated on a similar basis to those attributed to the compost produced from open air windrow operations (described in Section A.8.5.2). Since some of the material has already been degraded by the AD process, there is less solid material to enter the post-digestion composting process, and therefore less compost will be produced (typically less than half of that produced from a similar quantity of material sent to an in-vessel composting process). 90% of the compost produced from AD facilities is assumed to be used in agriculture, with the remainder used for horticulture and amateur gardening. The environmental impacts associated with the diesel required to spread the compost on land are calculated in the same way as for the composting facilities.

A.8.8 Landfill

A.8.8.1 Landfill Gas Generation

In order to capture the relationship between degradation and residence time, our model links the nature of the constituent organic compounds to the release of greenhouse gases through time-dependent 'first order decay' functions, as is done in both the Land Quality Management (LQM) landfill model (used for inventorying the UK's greenhouse gas emissions for waste), and in the IPCC default model.[142] Emissions of methane from landfill are allocated to specific years over a 150 year period. The degradation factors within the model have been validated to some extent through assessing the implied methane emissions from the materials and cross-checking against work undertaken in the United States and by the UK Environment Agency.[143]

The constituent carbon fractions degrade at different speeds as a result of variations in their chemical and physical structure. Our model uses three degradation speeds to represent the varying speeds at which carbon degrades within the landfill.[144] The simplified grouping of carbon fractions used within the model is shown in Table 57.

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[142] Land Quality Management (2003) Methane Emissions from Landfill Sites in the UK, Report for Defra, January 2003; IPCC (2006) Guidelines for National Greenhouse Gas Inventories: Chapter 3 – Solid Waste Disposal

[143] Barlaz M (1997) Biodegradative Analysis of Municipal Solid Waste in Laboratory-scale Landfills, EPA 600/R-97-071, Washington, DC: USEPA; Gregory R and Revans A (2000) Part One, in Environment Agency (2000) Life Cycle Inventory Development for Waste Management Operations: Landfill, Project Record P1/392/3, Bristol: Environment Agency

[144] The same approach is taken in modelling other anaerobic processes, including landfill (see, for example, Land Quality Management (2003) Methane Emissions from Landfill Sites in the UK, Report for Defra, January 2003). The method is not usually applied to aerobic processes, though some work of a similar nature has been undertaken for degradation of organic matter in soil (including the work by DU)

Table 57: Simplification of Carbon Fractions for Landfill

Speed of Decay	Carbon Fraction(s)
Fast	Sugars
Medium	Fats, Proteins, Cellulose
Slow	Lignin and some Cellulose[1]
Notes 1. Some cellulose is bound within the lignin and is therefore similarly resistant to degradation	

Source: Dalemo M (1996) The Modelling of an Anaerobic Digestion Plant and a Sewage Plant in the ORWARE Simulation Model, Rapport 213, Swedish University of Agricultural Sciences, Uppsala 1996

Table 58 shows the impact of these differential degradation rates, and confirms the outputs of our model in terms of proportion of each type of material degraded after a *50* year period. The model assumes that all of the organic matter will degrade during 150 years.

Table 58: Degradation of Organic Matter in Landfill Over 50 Years

Material	Proportion degraded after 50 years
Food waste	83%
Garden waste	71%
Office paper	75%
Newspaper[1]	70%
Textiles (natural fibres)	79%
Wood	79%
Notes 1. Newspaper contains a greater proportion of lignin than other forms of paper	

To take account of the time profile of these emissions over the 150 year period, the damage costs for landfill emissions are discounted using a declining long-term discount rate as recommended in the UK Treasury's Green Book. Table 59 shows the rates at which damage costs are discounted for the relevant time periods.

Table 59: Declining Long Term Discount Rate Applied to Landfill Emission Damage Costs

Period of years	0 - 30	31 - 75	76 - 125	126 - 200	201 - 300	301 +
Discount rate	3.5%	3.0%	2.5%	2.0%	1.5%	1.0%

Source: The UK Treasury Green Book

A.8.8.2 The Issue of Gas Capture

There is some debate with regard to both the efficiency landfill gas capture and the proportion of the gas that is used for energy generation. Of these, the gas capture rate is both the most sensitive and the most contested component.

A previous assessment undertaken by Eunomia used a gas capture rate of 50%, an approach based upon two studies conducted on behalf of Defra by LQM and Enviros.[145] A study conducted by ERM on behalf of Defra, however, assumed a 75% capture rate over the 100 year timeframe assessed.[146] A subsequent ERM report acknowledged that if one moved the analysis beyond this (somewhat arbitrary) timeframe, lifetime capture rates might be around 59%.[147] Documentation supplied with the Golders model indicates that the expert review group formed as part of that study considered that 85% of the gas would be collected during the gas utilisation phases, and a lifetime 75% gas capture rate appears to have been suggested upon that basis.[148]

The wider literature suggests a range of estimates for the efficiency of gas collection with a distinction being made between instantaneous collection efficiencies and the proportion of gas that can be captured over the lifetime of the landfill.[149] Whilst

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[145] Eunomia (2006) A Changing Climate for Energy from Waste? Final report to Friends of the Earth, May 2006; LQM (2003) Methane Emissions from Landfill Sites in the UK, Report for Defra, January 2003; Enviros, University of Birmingham, RPA Ltd., Open University and M. Thurgood (2004) Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes, Final Report to Defra, March 2004

[146] ERM (2006) Impact of Energy from Waste and Recycling Policy on UK Greenhouse Gas Emissions, Final Report for Defra, January 2006

[147] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, Defra R&D project WRT 237. December 2006

[148] Golder Associates (2005) Report on UK Landfill Methane Emissions: Evaluation and Appraisal of Waste Policies and Projections to 2050, report for Defra, November 2005

[149] Anderson P (2005) The Landfill Gas Recovery Hoax, Abstract for 2005 National Green Power Marketing Conference; USEPA (2004) Direct Emissions from Municipal Solid Waste Landfilling, Climate Leaders Greenhouse Gas Inventory Protocol – Core Module Guidance, October 2004; Brown K A, Smith A, Burnley S J, Campbell D J V, King K and Milton M J T (1999) Methane Emissions from UK Landfills, Report for the UK Department of the Environment, Transport and the Regions

instantaneous collection rates for permanently capped landfilled waste can be as high as 90%, capture rates may be much lower during the operating phase of the landfill (35%) or when the waste is capped with a temporary cover (65%).[150] In addition, gas collection is technologically impractical towards the end of the site's life. The Intergovernmental Panel on Climate Change (IPCC) has recently stated that lifetime gas capture rates may be as low as 20%.[151] We would consider, however, that landfills in the UK are somewhat better engineered than in the general (global) case, although a recent report by the European Environment Agency uses the IPCC figure.[152]

Our model assumes that waste which has been pre-treated (e.g. through an aerobic stabilisation process) will behave differently in landfill with respect to the generation of landfill gas, and that pre-treated wastes will therefore ultimately require a different form of gas management in landfill.

We have assumed a landfill gas capture of 50% for untreated wastes, in line with the lifetime capture rates suggested in the wider literature for well-managed landfills (such as those currently operating in the UK). Assumptions for pre-treated wastes are outlined in Section A.8.8.5.

A.8.8.3 Energy Generation from Landfill Gas

Energy is generated from a variable proportion of the recovered gas. At times of high flux, emissions can be greater than the capacity of the engines and thus a proportion of the gas must be flared. At times of low flux, i.e. towards the end of the site lifetime, emissions may be too small for the gas engines to function effectively. In such a situation, the usual practice of the landfill operator is to flare the gas.

LQM carried out a survey of landfill operators to estimate the total flare capacity across UK landfills.[153] They noted within their analysis that:

> *There are difficulties in ascertaining the actual volumes of LFG burnt as detailed records, if they exist at all, will be held by individual site operators. It is rare to find a flow stack with a flow measurement device installed, even though the capital cost of such a device is relatively small.*

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[150] Spokas K, Bogner J, Chanton J P, Morcet M, Aran C, Graff C, Moreau-Le Golvan Y and Hebe I (2006) Methane Mass Balance at 3 Landfill Sites: What is the Efficiency of Capture by Gas Collection Systems? Waste Management, 5, pp515-525

[151] IPCC (2007) Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (Metz B, Davidson O R, Bosch PR, Dave R, and Meyer L A (eds)), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., pp 60o

[152] Skovgaard M, Hedal N, Villanueva A, Andersen F and Larsen H (2008) Municipal Waste Management and Greenhouse Gases, ETC/RWM Working Paper 2008/1, January 2008

[153] Land Quality Management (2003) Methane Emissions from Landfill Sites in the UK, Final Report for Defra, January 2003

LQM did not consider the amount of energy generated from LFG within their analysis, although they estimated the total flaring back-up capacity to be around 60% of generation capacity. It is usual for landfill operators to maximise energy generation as this represents a revenue stream. We assume within the current analysis that 40% of the recovered gas will be flared. Although it is acknowledged that there is some uncertainty here, the impact of this uncertainty (in terms of CO_2 equivalent offsets associated with energy generation from landfill) is relatively small.

A.8.8.4 Oxidation of Landfill Gas

Some of the uncaptured landfill gas will be oxidised as it passes through the cap to the surface, the proportion being dependent upon the nature of the cap. The USEPA suggests a range of 10% to 25%, with clay soils at the lower end of the range and top-soils being at the higher end. The lower value reflects what was proposed by Brown et al in 1999 in a study on behalf of what was then the DETR.[154] The IPCC similarly suggested 10% as the default oxidation rate.[155]

However, a recently published review of the wider literature on this subject suggests that the mean fraction of methane oxidised was 36% (an average across 42 studies taken in a variety of locations).[156] We have assumed an oxidation rate of 20% for untreated waste sent to landfill, taking into account both the range of results suggested by the USEPA along with data from the literature review.

A.8.8.5 Landfill of Pre-Treated Waste

Under the very low fluxes of landfill gas assumed to occur when pre-treated wastes are landfilled, the methanotrophic bacteria within the soil cover can oxidise a much larger portion of the methane delivered them, oxidising up to 95-100% of the emission. Fugitive emissions of methane are therefore minimal in this case. Landfill gas capture is not necessary (the low flux makes this technically infeasible, as was previously discussed) and therefore no energy is generated from the landfill gas.

We assume that 90% of the methane is oxidised by the landfill cover when pre-treated waste is landfilled. This reflects the likely management of landfill gas in a situation where a ban on untreated waste to landfill has been put in place

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[154] Brown K A, Smith A, Burnley S J, Campbell D J V, King K and Milton M J T (1999) Methane Emissions from UK Landfills, A Report for the UK Department of the Environment, Transport and the Regions

[155] IPCC (2007) Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (Metz B, Davidson O R, Bosch PR, Dave R, and Meyer L A (eds)), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., pp 60*0*

[156] Chanton J P, Powelson D K and Green R B (2009) Methane Oxidation in Landfill Cover Soils, is a 10% Default Value Reasonable? Journal of Environmental Quality, 38, pp 654-663

A.8.8.6 Summary of Assumptions for Landfill

Table 60 summarises our assumptions with regard to the management of landfill gas for untreated waste. The totals include emissions associated with energy used at the landfill (although these are insignificant in comparison to the direct emissions from the process).

Table 60: Landfill Gas Management – Untreated and Pre-treated Wastes

Parameter	Assumption
Proportion of methane captured (untreated waste)	50%
Proportion of methane captured (waste pre-treated at MBT facility)	0%
Proportion of captured methane used for energy generation	60%
Proportion of captured methane that is flared	40%
Efficiency of electricity generation, landfill gas engine	35%
Rate of oxidation of methane within the landfill cover (untreated waste)	10%
Rate of oxidation of methane within the landfill cover (pre-treated waste)	90%
Electricity requirement	1% of generated
Diesel requirement	1.65 l / tonne

A.8.9 Incineration

A.8.9.1 Direct Emissions to Air

Greenhouse gas emissions occurring as a result of the incineration of waste will be dependent upon the carbon content of the dry material, along with the overall efficiency of energy generation that results from the combustion of that material. Table 61 details the carbon content of waste components together with their energy and moisture content.

Table 61: Carbon Contents and Energy Content of Materials in the Waste Stream

	Total C (% fm)	Proportion of C that is non fossil	Energy content (lower heating value as received) MJ per kg	Typical moisture content
Paper	41%	100%	13	10%
Card	32%	100%	12	24%
Dense plastic	77%	0%	35	10%
Plastic film	72%	0%	33	15%
Textiles	49%	50%	15	19%
Glass	0%	0%	0	2%
Ferrous metal	0%	0%	0	3%
Non ferrous metal	0%	0%	0	5%
Wood	32%	100%	12	30%
Garden waste	26%	100%	8	45%
Food waste	14%	100%	4	70%
Misc. combustibles	40%	50%	15	41%
Misc. non combustibles	7%	0%	0	6%
Fines	30%	100%	5	41%

N_2O emissions are modelled based on previous research undertaken by Eunomia on behalf of WRAP.[157] The considerable uncertainty with respect to these emissions is acknowledged within the EU BREF note, which provided a range of 5.5 – 66 g N_2O per tonne of waste treated by the facility. We use the mid point of these values within the current analysis. CH_4 emissions are negligible from incineration facilities.

[157] Eunomia (2007) Emissions of Nitrous Oxide from Waste Treatment Processes, Report to WRAP, July 2007

Total climate change impacts associated with incinerating one tonne of residual waste are typically in the order of one tonne of CO_2 equivalent per tonne of residual waste (including the biogenic CO_2 emissions). The exact figure will vary, depending on the composition of residual waste treated by the plant.

A.8.9.2 Energy Use at Incineration Facilities

The energy usage of the plant depends upon the scale of plant, and the nature of the flue gas cleaning system. It also depends upon the presence or otherwise of:

- Mechanical pre-treatment systems;
- Incineration air preheating;
- Equipment for re-heating of flue gas;
- Waste water evaporation plant;
- Flue gas treatment systems with high pressure drops (which demand more powerful fans); and
- Changes in the energy content of input waste (necessitating use of fuel to maintain minimum combustion temperatures).

ERM's analysis suggests 3.9 kWh electricity is consumed per tonne of waste treated at an incinerator, with process diesel use indicated as 1.2 kg of per tonne of waste.[158] They arrived at these figures using Environment Agency data collected for the development of the waste model WRATE. However, they note in their report that:

> *These process data were used as a substitute for all thermal treatment processes. In reality the ancillary requirements of each will differ, but within the context of the research the more important parameter relates to the energy conversion efficiency of the process.*

ERM's energy consumption figures appear to be very low in comparison to values given in the wider literature. It is certainly true that far greater than use, but this does not make the figures for energy use less important to the extent that the range of values in the literature spans a small percentage of the energy in the waste, but the choice of efficiencies used is also, typically, discussed in terms of whether the figures used should be a small percentage higher or lower than some central figure (in other words, in deriving net generation figures, the figures regarding use are actually quite significant). The Draft BREF note for Incineration gives figures of:[159]

- Electricity use - 62 - 257 kWh per tonne, average 142 kWh per tonne; and

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[158] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, Defra R&D Project WRT 237)

[159] A BREF note is a note prepared by the Joint Research Centre of the European Commission to give guidance to Member States as to what is implied by 'Best Available Techniques' under the Directive on Integrated Pollution Prevention and Control. See: European Commission (2005) Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration, Final Draft, May 2005

- Heat demand - 72 - 3,366 GJ thermal energy per tonne, average 433 GJ thermal energy per tonne.

These, in turn, are far higher than figures suggested in, for example, reports by Erichsen and Hauschild (46 kWh electricity per tonne) though this figure reflects only the operation of gas cleaning equipment. [160] The Flemish Institute for Technological Research (VITO) gave the following consumption of energy for processes with and without SCR (these were based upon incinerators operated by Seghers Better Technology):[161]

- Natural gas: 7.2 m3 per tonne
- Oil: 4 kg per tonne (or 4.7 litres per tonne)
- Electricity use (per tonne): 80 kWh with Selective Non-Catalytic Reduction (SNCR) pollution abatement, 85 kWh with Selective Catalytic Reduction (SCR) abatement.

To ensure the catalyst is not contaminated by other elements within the flue gas the SCR abatement system is typically located just prior to the emissions stack, which requires the gas to be reheated using additional electrical energy.[162]

CEWEP's survey of 97 facilities during 2001-2004 suggested the average electricity used by incineration processes was 78 kWh per tonne of waste input.[163] We use the CEWEP figure for electricity consumption with SNCR and VITO's figures for energy use assuming SCR within the current analysis. We have also used VITO's data for the diesel usage, but have assumed no natural gas is used by the process. These figures appear appropriate to UK incinerators.

A.8.9.3 Energy Generation

The efficiency of generation of electricity by an incinerator may be quoted gross, or net of any energy used in the plant itself. The energy use in the plant depends partly upon the nature of the flue gas cleaning system used, but also upon a range of other factors. The relationship to flue gas cleaning is important since it seems likely that as standards for abatement have improved, so the energy used in achieving those levels of abatement has increased also.

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[160] Hanne L, Erichsen L and Hauschild M (2000) Technical Data for Waste Incineration - Background for Modeling of Product Specific Emissions in a Life-cycle Assessment Context, Elaborated as part of the EUREKA project EUROENVIRON 1296: LCAGAPS, sponsored by the Danish Agency for Industry and Trade, April 2000

[161] VITO (2000) Vergelijking van Verwerkingsscenario's voor Restfractie van HHA en Niet-specifiek Categorie II Bedrijfsafval, Final Report

[162] Note that this is not always the case – see European Commission (2005) Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration, Final Draft, May 2005

[163] Riemann I (2006) CEWEP Energy Report (Status 2001-2004): Results of Specific Data for Energy, Efficiency Rates and Coefficients, Plant Efficiency Factors and NCV of 97 European W-t-E Plants and Determination of the Main Energy Results, updated July 2006

ERM suggested gross efficiencies of 20-27% for conventional incineration with steam cycle electricity generation in a recent report for Defra.[164] Fichtner quotes a 'realistic range' for net electrical efficiency of 19-27%.[165] The highest figures we have seen quoted are those quoted in the context of the Belvedere Inquiry where it was claimed that a net efficiency of 27% would be achieved. This was based around assumptions of a thermal efficiency of 84% and an electrical efficiency of 35%. These are optimistic in the context of efficiencies currently achieved and are likely to be deliverable only at large operating scales. The Draft BREF note gave no case where the net export of electricity exceeded 18%.[166] A survey of 25 incinerators across Europe generating electricity only reported a maximum *gross* energy efficiency of 27.9% with a weighted mean efficiency of 21.8% across the 25 facilities (the mean *net* efficiency was given as 17.7%).[167] This study uses a *gross* efficiency of 27% for facilities generating only electricity, reflecting the top end of the range quoted by ERM and the CEWEP survey.

Whilst CEWEP supplies maximum values for heat and electricity generation for facilities operating in CHP mode, the survey data does not directly supply any information regarding the ratio of heat to electricity produced at each of the facilities concerned. Where thermal facilities are concerned, and where steam turbines are used to generate energy, there is a trade-off between the generation of electricity and the generation of heat.

In its submission to the DTI as part of a review of the Renewables Obligation, ILEX assumed electrical output would be reduced at an approximate rate of 1 MW of electrical energy for every 4 MW of heat off-take.[168] Data from CEWEP gives the maximum heat output from surveyed facilities surveyed producing only heat as 92.7%, suggesting a theoretical ratio of 3.3 MW heat for every MW of electricity.[169] The maximum heat output for any of the surveyed facilities operating in CHP mode was 83.9%, whilst the maximum electricity output for the CHP facilities was 26.9%. This suggests a ratio of 3.1 MW heat for every MW of electricity. However, the German Waste Incineration Association suggests that the ratio should be rather lower at 2.3 MW heat for each MW of electricity, based on the data from German facilities (the

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[164] ERM (2006) Carbon Balances and Energy Impacts of the Management of UK Wastes, Defra R&D Project WRT 237

[165] Fichtner Consulting Engineers Limited (2004) The Viability Of Advanced Thermal Treatment Of MSW In The UK, ESTET, March 2004

[166] European Commission (2005) Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration, Final Draft, May 2005

[167] Riemann I (2006) CEWEP Energy Report (Status 2001-2004): Results of Specific Data for Energy, Efficiency Rates and Coefficients, Plant Efficiency Factors and NCV of 97 European W-t-E Plants and Determination of the Main Energy Results, updated July 2006

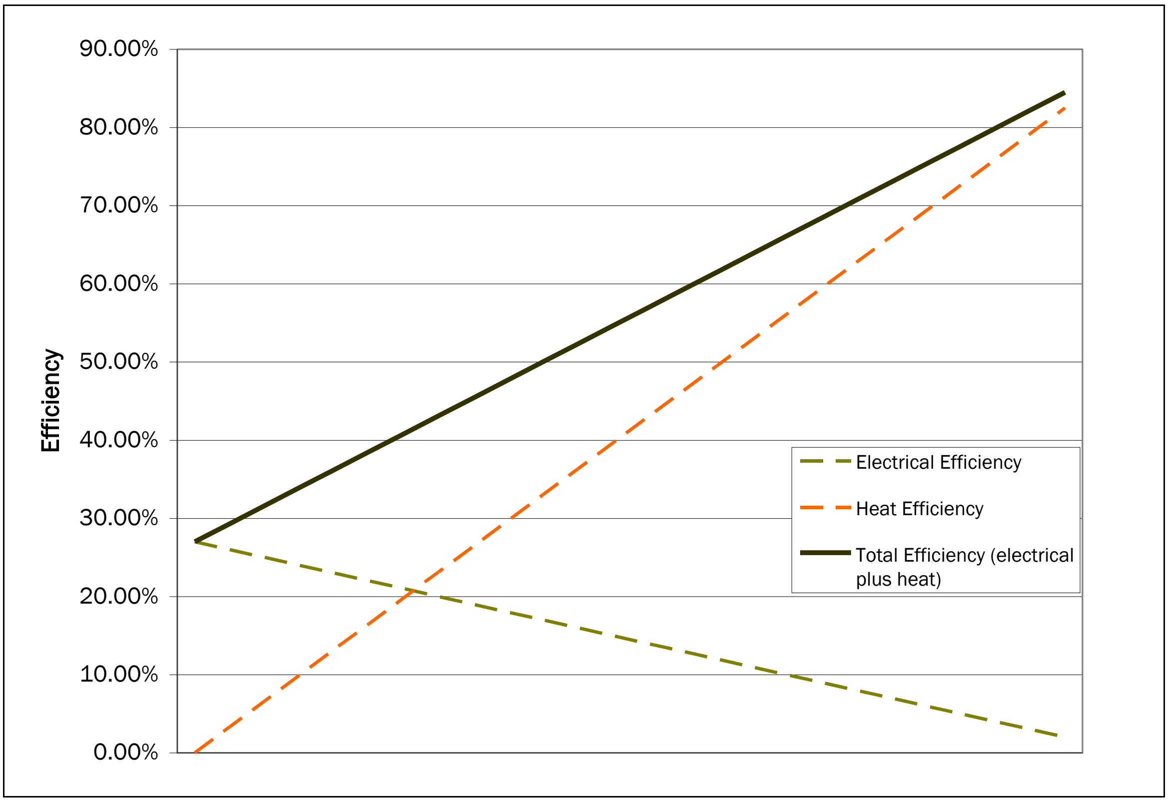
[168] ILEX Energy Consulting (2005) Extending ROC Eligibility to Energy from Waste with CHP, Supplementary Report to the Department of Trade and Industry, September 2005

[169] This is simply calculated as the ratio of the maximum gross efficiency of heat generation relative to the maximum gross electrical generation efficiency of 29.7%

majority of which operate in CHP mode).[170] It is not clear, though, whether the German figures speak in terms of gross or net generation, or indeed, whether they take into account the heat load effect (in other words, these figures may relate to the electricity and heat actually put to a useful purpose).

The relationship between the electrical efficiency, heat generation efficiency and total generation efficiency (as outlined above) is shown graphically in Figure 10.

Figure 10: Electricity, Heat and Total Efficiency – Facilities Operating in CHP Mode



Our energy generation efficiencies for facilities operating in CHP mode are based on the average electricity production for CHP facilities surveyed by CEWEP, using the higher ratio in the CEWEP report of 3.3 MW heat per MW electricity to calculate the heat production. We assume a total system generation efficiency of 66%, with electrical and heat generation efficiencies of 10% and 56% respectively.

A.8.9.4 Recovery of Metals

The efficiency with which metals are recovered from incineration facilities is modelled based on a survey of Dutch facilities.[171] The survey suggested that 70% of the ferrous

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170 Available from www.itad.de

171 Muchova L and Rem P (2008) Wet or Dry Separation: Management of Bottom Ash in Europe, Waste Management World Magazine, 9(3)

metal could be recovered as well as 30% of the non ferrous. The materials recovery is assumed to result in offset emissions as previously described in Section A.8.2.

A.8.9.5 Summary of Assumptions

Table 62 summarises the assumptions for incineration discussed within this section.

Table 62: Assumptions for Incineration

Parameter	Assumption
Gross electrical generation efficiency	27%
Gross electrical efficiency (CHP mode)	10%
Gross heat efficiency (CHP mode)	56%
Electricity demand for flue gas cleaning	78 kWh / t input
Diesel use by process	4.7 l / t input
Recycling of bottom ash	50%
CH_4 emissions from process	0 kg CH_4 / t
N_2O emissions from process	0.04 kg N_2O / t
Recovery rate for ferrous metals	70%
Recovery rate for non-ferrous metals	30%

A.8.10 MBT (Aerobic ‘Stabilisation’ Systems)

A.8.10.1 Emissions to Air from the Stabilisation Process

The central aim of aerobic stabilisation processes is to produce an output which has a reduced biodegradability thereby decreasing the environmental impacts associated with landfilling this material. Our assumptions for the landfill of pre-treated (stabilised) material are presented in A.8.8.

The approach for modelling the impacts of stabilisation processes draws upon work by Eunomia on behalf of WRAP, which was based upon a raft of published research.[172] The body of research included work by Baky and Eriksson, Sonneson,

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[172] Schleiss K (1999) Grüngutbewirtschaftung im Kanton Zürich aus betriebswirtschlaftlicher und ökologischer Sicht: Situationsanalyse, Szenarioanalyse, ökonomische und ökologische Bewertung sowie Synthese mit MAUT, Dissertation ETH No 13,746, 1999; Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC

and Komilis and Ham, all of whom investigated the link between the biochemical composition of the waste and the release of CO_2 within composting processes. This research, together with data sourced from technology suppliers, was used to model the degradation of carbon fractions within our model and the subsequent release of biogenic CO_2 from the process.

The release of biogenic CO_2 is the most significant environmental impact resulting from the stabilisation process - as is the case with the previously described composting processes. Relatively small emissions of CH_4 and N_2O (assumed to 0.01 and 0.04 kg of pollutant per tonne of waste) also occur.

Energy use at facilities also results in environmental impacts although the energy requirements are usually less than those of incineration facilities. Typical requirements are 50 kWh of electricity per tonne of waste to facility along with 1 litre of diesel.

A.8.10.2 Recovery of Materials for Recycling

Stabilisation facilities typically recover plastics in addition to ferrous and non ferrous metals, with the following recovery rates:

- Dense plastics 40%;
- Ferrous metals 65%;
- Non ferrous metals 60%.

These recovery rates for materials are considered typical of the better performing facilities currently operating in the UK. Although some MBT facilities occasionally target other materials for removal from the residual stream (such as paper), we include within our model only those for which a market currently exists within this context.

This materials recovery results in emissions offsets as previously described in Section A.8.2.

A.8.10.3 Summary of Assumptions

Table 63 outlines the key assumptions within the model for stabilisation processes.

Table 63: Assumptions for Stabilisation Process

Research & Consulting (2002) Economic Analysis of Options for Managing Biodegradable Municipal Waste, Final Report to the European Commission; Komilis D P and Ham R K (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, Journal of Environmental Engineering, 130(11), pp.1390-1400; Baky A and Eriksson O (2003) Systems Analysis of Organic Waste Management in Denmark, Environmental Project No. 822, Copenhagen: Danish EPA; Sonesson U (1996) Modelling of the Compost and Transport Process in the ORWARE Simulation Model, Report 214, Swedish University of Agricultural Sciences (SLU), Department of Agricultural Engineering, Uppsala Sweden

Parameter	Assumption
Electricity requirement	50 kWh / t input
Diesel use by process	1 l / t input
CH_4 emissions from process	0.01 kg / t input
N_2O emissions from process	0.04 kg / t input
Recovery rate for ferrous metals	65%
Recovery rate for non ferrous metals	60%
Recovery rate for dense plastics	40%

A.8.11 MBT (Aerobic 'Biodrying' Systems)

A.8.11.1 Emissions to Air

Biodrying systems involve the use of the heat from the process of biodegradation to reduce the moisture content of waste prior to its being mechanically refined (including using material separation technologies) for use as fuel. During this process degradation of some of the carbon fractions will occur, but the amount of degradation is relatively limited in comparison that occurring during aerobic decomposition (stabilisation) processes. Key differences between biodrying and stabilisation processes are the air-flow used to drive the process, the different water management systems (in stabilisation processes, the waste is kept wet to maintain biodegradation, whilst in biodrying processes, the material is allowed to dry), and the retention times (which are much shorter in the case of biodrying). Biodrying processes are modelled using an analysis of data from technology suppliers.

The central aim of biodrying processes is to produce a fuel. A reject stream is also produced, which is assumed to be stabilised before being sent to landfill, using the process previously described in Section A.8.10 of this Appendix. Our assumptions for the landfill of pre-treated (stabilised) material have been previously discussed in Section A.8.8.5.

Emissions associated with the biodrying phase of the process are relatively small – as is the case with the stabilisation facilities. However the environmental impacts associated with the combustion of the fuel are more significant. These impacts are described in Section A.8.11.2.

The energy requirements for the biodrying process are similar to those of stabilisation facilities. There will be additional energy requirements associated with the combustion of the fuel.

A.8.11.2 Energy Generation

Our assumptions regarding the nature of the SRF produced are detailed in Table 64.

Table 64: Model Parameters for Residual Waste to SRF

Parameter	Assumption
Amount of SRF produced by biodrying process (per tonne to facility)	0.4 tonnes
Energy content of SRF (lower heating value as received)	16.5 MJ / kg
% of carbon that is non-fossil	46%

We assume that the SRF is combusted in an incineration facility generating electricity as was previously described in Section A.8.9.[173] This energy generation results in avoided climate change impacts that would otherwise have occurred through the generation of energy by other means. Typically emissions of 170 kg of CO_2 equivalent are avoided through the generation of electricity when impacts are considered on the basis of one tonne of waste to the MBT facility.

A.8.11.3 Recovery of Materials for Recycling

MBT biodrying facilities use similar separation techniques to those employed at the stabilisation facilities. However plastics are not removed for recycling where the aim is to produce a fuel with a relatively high calorific value. We therefore assume the same rates of recovery for the metals as previously described in Section A.8.11.3, with no recovery of dense plastics.

A.8.11.4 Summary of Assumptions

Table 65 outlines key assumptions used to model the biodrying phase.

[173] Combustion facilities accepting solely SRF will typically require a different type of incinerator than is the case with facilities accepting untreated waste – due to the higher calorific value and the more homogenous nature of the fuel. SRF is however a suitable fuel for fluidised bed incineration facilities, such as that operating in Allington, Kent. We have not modelled a differential between the two types of facility, given the relative lack of data regarding the operation of fluidised bed facilities within the UK.

Table 65: Assumptions Used to Model Biodrying Systems

Parameter	Assumption
Residence time in biodrying phase	12 days
Residence time of rejects in maturation (stabilisation) phase	7 weeks
Electricity requirement[1]	40 kWh / t input
Diesel use by process[1]	0.5 l / t input
CH_4 emissions from process[2]	0.01 kg / t input
N_2O emissions from process[2]	0.02 kg / t input
Recovery rate for ferrous metals	65%
Recovery rate for non-ferrous metals	60%
Notes: 1. Per tonne input to the MBT facility. 2. Per tonne input to the biodrying process.	

A.8.12 Autoclaving

A.8.12.1 Emissions to Air

Autoclave (or MHT) facilities use either steam or direct heat to treat the waste. Where the heat treatment is carried out under pressure, the technology is referred to as autoclaving. The energy requirement to heat the waste is a key parameter for autoclaving and heat treatment technologies, and this varies according to the exact approach undertaken. As is the case with the MBT Biodrying Systems, the aim is to produce a fraction of dry recyclables, a fuel fraction (sometimes called 'floc') and a residue fraction which is usually landfilled.

Autoclaving is essentially a sterilisation technology and is commercially proven in a variety of other industries. There are, however, very few such facilities treating *MSW* on a commercial scale.[174] Our model is based on data provided by technology suppliers, a number of whom are currently operating demonstration facilities throughout the country.

[174] A 100,000 tonne per annum autoclave facility opened in Yorkshire at the beginning of 2008 - see http://www.letsrecycle.com/do/ecco.py/view_item?listid=37&listcatid=217&listitemid=9110.

Our analysis assumes that the fuel fraction is subsequently gasified, with the syngas used to generate energy via a steam turbine. A residue fraction, which will also contain appreciable amounts of organic material, is assumed to undergo a stabilisation process prior to being landfilled, in order to reduce its biodegradability.

Emissions from the autoclaving process itself are relatively trivial, being limited to some NMVOC emissions associated with waste handling. Much of the environmental impact of the overall treatment process is associated with the energy generation phase.

However the energy requirements of MHT and autoclave plant are usually more substantial than that of the MBT facilities, as energy is required to heat the waste. We have assumed an energy demand of 75 kWh of electricity and 15 m^3 of gas per tonne of waste to the MHT facility. There will be additional energy requirements for the energy generation phase.

A.8.12.2 Recovery of Materials for Recycling

MHT and autoclave facilities aim to recover a wider range of materials than is typically seen for MBT facilities. Operational data from MHT and autoclave plant suggests that recovery rates are similar to those seen at MBT plant for those materials recovered by both types of facility, although the recovery rate for ferrous metals is higher at the former. Table 66 outlines the recovery rates used in our model of MHT facilities. The total mass of material removed for recycling from one tonne of waste is dependent upon the composition of material entering the facility. Typically, recovery rates such as those shown in the table will result in the removal of approximately 100 kg of recyclate per tonne of waste to the facility.

Table 66: Recovery of Materials for Recycling at Autoclave Facilities

Parameter	Assumption
Plastic film	20%
Dense plastic bottles	25%
Other dense plastic	25%
Glass	80%
Ferrous metal	90%
Non-ferrous metal	60%

It is unlikely that glass removed by the MHT facilities will be suitable for the closed loop process previously described in Section A.8.2.2. We have therefore attributed no environmental benefit for the recycling of this material, as the benefits associated with the use of this material as aggregate are minimal.[175]

A.8.12.3 Summary of Assumptions

Table 67 presents a summary of our assumptions used to model MHT and autoclave facilities, whilst Table 68 outlines those used to model the gasification of the SRF produced by the MHT process.

Table 67: Summary of Assumptions Used to Model MHT

Parameter	Assumption
Fuel parameters Moisture content Proportion of biomass by weight (fresh matter basis) Calorific value (LHV, fresh matter basis) Amount of SRF (per tone to facility)	 15% 70% 15.0 MJ / kg 320 kg
Electricity use by MHT process	75 kWh / t
Gas used by MHT process (for heating purposes)	15 m^3 / t
Total recovery of materials for recycling (from one tonne of residual waste)	Circa 10%

A.8.13 Gasification of SRF from MBT and Autoclave facilities

Gasification is a process in which materials, when heated, are exposed to some oxygen, but not a sufficient amount to lead to combustion. Facilities usually carry out some pre-treatment of the raw MSW prior to gasification, which typically involves the removal of metals and shredding of the waste. The output of the gasification process is a syngas which is combusted to generate energy, with the calorific value of this syngas being dependent upon the composition of the input waste to the gasifier. The other main product produced by gasification is a solid, non-combustible 'char'.

If the syngas is sufficiently cleaned it can be used in a gas engine, where the efficiency of generation is improved. However, the majority of *existing* commercially operating facilities use a steam turbine or boiler for the generation of energy as this requires less clean up of the syngas. Efficiencies of generation in this case are likely to be similar to those seen for conventional incinerators, with the smaller gasification facilities generating less energy than the larger plant.

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[175] See Enviros (2003) Glass Recycling – Life Cycle Carbon Dioxide Emissions, Internal Report for the British Glass Public Affairs Committee

Our central assumption is that the syngas is used in a steam turbine operating in CHP mode with efficiencies of energy generation the same as those assumed for an incinerator operating in CHP mode (see Section A.8.9). The gross electrical generation efficiency is therefore assumed to be 10% and the gross heat generation efficiency 56%. The overall benefits associated with heat generation include the heat 60% utilisation factor described in Section A.8.1.2.

The climate change impacts associated with the gasification of the SRF are dependent upon the carbon content of the fuel, as is the case with combustion processes. The biogenic (or non-fossil) CO_2 content is dependent not only on the composition of waste received at the MBT or autoclave facility, but also on the requirements of the fuel consumer.

As was the case with the incineration facilities, estimates of the use of energy within the facility (parasitic load) vary considerably. In gasification facilities, energy use depends not only on the gas clean up techniques, but also on how much energy is used to heat the waste within the gasification process. The assumptions used within the current analysis are based on those developed by Hellweg.[176]

Table 68: Summary of Assumptions Used to Model Gasification

Parameter	Assumption
Efficiency of electricity generation	10%[1]
Efficiency of heat generation	56%[1]
Electricity used by process	97 kWh / tonne[2]
Diesel used by process	4.7 litre / tonne[2]
Notes: 1. These are gross generation efficiencies, which do not take into account energy use by the process 2. Energy demand figures are described per tonne of waste to the gasifier	

[176] Stefanie Hellweg (2000) Time- and Site-dependent Life-cycle Assessment of Thermal Waste Treatment Processes, Diss. ETH No.13999, Zurich

A.9.0 Comparison of Assumptions with WRATE

Although some relevant information on the environmental impacts of waste management systems is available within the Environment Agency's Waste Resources Assessment Tool for the Environment (WRATE) tool, we have based our analysis of the GHG impacts of most of the waste treatment facilities on data provided by our own proprietary models of residual and organic waste treatment facilities. As detailed above, these are based upon a range of data sources, which are totally transparent to the user, in this case, the GLA.

WRATE considers the environmental impacts of facilities wherever possible on actual data obtained from facilities currently operating in the UK, although many process models contain information extrapolated from other facilities and theoretical values supplied by literature sources to fill the gaps that exist in the operational data supplied by the facilities.

The tool offers the user some flexibility in the modelling process. Those holding an expert license for the software can modify much of the data contained within the models of individual treatment processes through the creation of so-called 'user defined' processes. It is also possible to create new bespoke models of processes not already included within WRATE.[177] However, it is not possible to make changes to the landfill module – this is considered part of the background database, and modification of this data is not possible even for 'Expert' license holders (of which Eunomia is one such holder). Eunomia believes that the most significant deficiencies of the tool relate to the information associated with the landfill module.

WRATE substantially underestimates the amount of methane emissions that result from the degradation of most wastes sent to landfill; for example, the emission of methane assumed to occur over 150 years from landfilled food waste and paper is about half of what we would expect even given the same landfill gas capture rate (currently fixed at 75% in WRATE).

The model that underpins WRATE with regard to the behaviour of landfills is called GasSim. This model produces similar results to another model which is used to prepare the UK's greenhouse gas inventory submitted to the IPCC. Inaccuracies associated with the latter model have now been acknowledged by Defra, although at the time of writing it appears there are no immediate plans to revise either model.

Further fundamental errors occur in the tool's treatment of the stabilised output from MBT facilities. The model assumes a proportion of the carbon is degraded within the biological part of the MBT process. However, when this stabilised material is subsequently landfilled, the methane emission is assumed to be exactly the same as that of the non-stabilised material - WRATE only accounts for the reduction in mass which occurs in material which is biologically pre-treated (occurring as a result of moisture loss). The model, therefore, significantly underestimates the extent to which

[177] Eunomia holds an expert license for WRATE, and has used this to create user-defined processes for both the New Earth Solutions facility and the ATAD composting process.

the biological component of the MBT process reduces the biological activity of material subsequently sent to landfill.[178]

Errors in the model's current datasets have been acknowledged by the Environment Agency, and a programme of updates – updating both the functionality of the model as well as some of the data contained with it – commenced in 2009. Although the first phase of updates was originally intended for release during 2009, these did not reach the user community until 2010.[179]

Information provided to date by the Environment Agency suggests that improvements to the front end of the tool have provided the focus for these initial updates, with the aim being to provide standard users with more control over the models of treatment processes. We note that substantial revisions have been made to the incineration (Energy from Waste) module in this regard. The updated model will also include a number of new MBT and gasification processes, although no revisions are planned for the existing Autoclave model.[180] The new version of WRATE will also include data from the latest version of the Swiss life cycle database ecoinvent.

The second set of updates now scheduled for release later during 2010 may allow users to modify assumptions with regard to landfill gas capture. However no other amendments to the landfill module are scheduled at present.

We feel that the current fundamental errors inherent within WRATE's approach to modelling the landfill impacts (for both treated and un-treated wastes) are such that the tool is currently an unreliable source of information with regard to the analysis of the GHG emissions from residual waste treatment. As such we have based our analysis on outputs from our own model, which we feel more closely reflects data provided within the wider literature on this subject.

We acknowledge, however, that use of WRATE alongside our own model can provide organisations with a cross-comparison of results, and we have undertaken such analysis previously on behalf of a number of public bodies and private sector companies. As a principle, Eunomia supports the ongoing development of WRATE, such that it is further improved to become a more useful life-cycle assessment tool.

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[178] The basis for our model of landfill impacts is discussed in Section A.8.8

[179] This was confirmed in presentations by the Environment Agency at the first WRATE user conference held in Birmingham on the 18th November 2009

[180] We understand that this is because the technology suppliers did not want to provide the Environment Agency with data to be included within the updated tool

A.10.0 Assumptions relating to Monetisation of CO_2

We apply the approach detailed in the latest guidance from DECC on the valuation of carbon in policy appraisal. [181] Under this new approach, the precise valuation methodology differs according to the specific policy question being addressed:

- For appraising policies that reduce/increase emissions in sectors covered by the EU Emissions Trading System (ETS), and in the future other trading schemes, a 'traded price of carbon' will be used. This will be based on estimates of the future price of EU Allowances (EUAs) and, in the longer term, estimates of future global carbon market prices;
- For appraising policies that reduce/increase emissions in sectors not covered by the EU ETS (the' non-Traded Sector'), the 'non-traded price of carbon' will be used, based on estimates of the marginal abatement cost (MAC) required to meet a specific emission reduction target;
- In the longer term (2030 onwards) consistent with the development of a more comprehensive global carbon market, the traded and non-traded prices of carbon converge into a single traded price of carbon.

The new values, in real terms, are as follows:

- A short term traded price of carbon of £25/tonne CO_2e in 2020, with a range of £14-£31. [182]
- A short term non-traded price of carbon of £60 per tonne CO_2e in 2020, with a range of +/- 50% (i.e. central value of £60, with a range of £30 -£90.
- A long term traded price of carbon with a value of:
    - £70 per tonne CO_2e in 2030, with a range of +/- 50% (i.e. £70 central estimate, £105 high estimate and £35 low estimate).
    - £200 per tonne CO_2e in 2050, with a range of +/- 50% (i.e. £200 central estimate, £300 high estimate and £100 low estimate).
- Linear interpolation is used to form a price series between 2020 and 2030, and 2030 and 2050.

These traded and non-traded carbon values are shown in Table 69. Because our analysis affects both traded and non-traded sectors, both traded and non-traded values of carbon are used, as appropriate, in this assessment.

By way of illustration, we have provided the following example:

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[181] DECC (2009) Carbon Valuation in UK Policy Appraisal: A Revised Approach. Climate Change Economics, Department of Energy and Climate Change, July 2009.

[182] 'CO_2e' refers to carbon dioxide equivalent. Non- CO_2 greenhouse gases have a global warming potential (GWP) ascribed to them. This describes their warming potency relative to carbon dioxide. For example, methane has a 100 year GWP of 21, and nitrous oxide has a 100 year GWP of 310.

- For an incinerator producing electricity, the GHG emissions from the process are valued at the higher non-traded price (incineration is not included in the EU-ETS), while the emissions reductions attributed to displacement of counterfactual generation of electricity are priced at the lower, traded value (electricity generation is covered by the EU-ETS);
- Likewise, for landfill, the GHG emissions (primarily methane) will be valued at the higher non-traded price (landfill is outside the EU-ETS), while the emissions reductions attributed to displacement of counterfactual generation of electricity are priced at the lower, traded value;
- For anaerobic digestion with biogas upgrading for use as a vehicle fuel, the emissions reductions attributed to displacement of diesel will be valued at the higher non-traded price (transport is not within the EU-ETS). If, however, the output of the process is electricity for supply to the grid the emissions reductions attributed to displacement of counterfactual generation of electricity are priced at the lower, traded value;
- For paper recycling, where both recycling, and the manufacture of paper using virgin materials are assumed to occur overseas, the emissions savings will be valued at the Shadow Price of Carbon.[183]

[183] The IPCC approach considers GHG emissions only insofar as they affect the UK's inventory as reported to the IPPC. In this case, any increase or reduction in GHG emissions overseas as a result of UK waste management is ignored. Under the Global approach, all emissions would be considered, irrespective of the location of their generation.

Table 69: Traded and non-traded carbon values (2008-2050) (£ at 2009 prices)

	Traded			Non-Traded			Shadow Price of Carbon
	Low	Central	High	Low	Central	High	
2008	12	21	26	25	50	75	27
2009	12	21	27	25	51	76	28
2010	12	22	27	26	52	78	28
2011	12	22	27	26	52	79	29
2012	13	22	28	27	53	80	29
2013	13	23	28	27	54	81	30
2014	13	23	29	27	55	82	31
2015	13	23	29	28	56	84	31
2016	13	24	29	28	57	85	32
2017	14	24	30	29	57	86	32
2018	14	24	30	29	58	87	33
2019	14	25	31	30	59	89	34
2020	14	25	31	30	60	90	34
2021	16	30	39	31	61	92	35
2022	18	34	46	31	62	93	36
2023	20	39	53	32	63	95	36
2024	23	43	61	32	64	96	37
2025	25	48	68	33	65	98	38
2026	27	52	76	33	66	99	39
2027	29	57	83	34	67	101	40
2028	31	61	90	34	68	102	40
2029	33	66	98	35	69	104	41
2030	35	70	105	35	70	105	42
2031	38	77	115	38	77	115	43
2032	42	83	125	42	83	125	44
2033	45	90	134	45	90	134	44
2034	48	96	144	48	96	144	45
2035	51	103	154	51	103	154	46
2036	55	109	164	55	109	164	47
2037	58	116	173	58	116	173	48
2038	61	122	183	61	122	183	49
2039	64	129	193	64	129	193	50
2040	68	135	203	68	135	203	51
2041	71	142	212	71	142	212	52
2042	74	148	222	74	148	222	53
2043	77	155	232	77	155	232	54
2044	81	161	242	81	161	242	55
2045	84	168	251	84	168	251	56
2046	87	174	261	87	174	261	58
2047	90	181	271	90	181	271	59
2048	94	187	281	94	187	281	60
2049	97	194	290	97	194	290	61
2050	100	200	300	100	200	300	62

A.11.0 Air Quality Impacts

A.11.1 Impacts of Plant on Air Quality

Measures outlined in the existing Air Quality Strategy were such that it was estimated that London would not achieve targets for annual nitrogen dioxide (NO_2) target set for 2005, and daily particulate matter (PM_{10}) target set for 2004.[184]

A previous study carried out by Eunomia and EMRC on behalf of the GLA looked at the air quality impacts of the installation of additional waste and wood energy facilities in London.[185] Two types of residual waste facility were considered:

1. A biodrying process with an SRF sent for gasification (with the syngas assumed to be combusted in a steam turbine); and
2. An AD facility, with the stabilised reject stream assumed to be sent to landfill.

The study combined dispersion modelling with baseline maps of pollution to identify areas of London that would be suitable for development of the different types of plant considered in the report, without causing a significant increase in pollutant levels in areas where air quality objectives are most difficult to meet. The analysis focused on NO_2 and PM_{10} as these are currently present in London at concentrations that exceed air quality objectives. An initial review of baseline maps showing existing exceedances, however, demonstrated that problems of compliance with air quality objectives were most severe for NO_2 (converted from NOx) and that all areas with PM_{10} exceedance also had NO_2 exceedance. To simplify the analysis and its subsequent interpretation, therefore, the bulk of the analysis then focussed on NO_2.

Information on the pollution abatement techniques for the waste treatment technologies was developed from the Integrated Pollution Prevention and Control (IPPC) BAT reference (BREF) document for waste treatment.[186] The analysis also used data from the London Atmospheric Emissions Inventory (LAEI), and maps of pollutant concentrations generated by ERG for the GLA.[187]

Dispersion modelling for individual plant was performed using the ADMS-Screen 3 Model. This requires information on various parameters in addition to the emission factors, including stack height and diameter, gas temperature and exit velocity, all of which were derived from reference facilities.

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[184] GLA (2002) Cleaning London's Air: The Mayor's Air Quality Strategy, September 2002

[185] Eunomia / EMRC (2008) Air Quality Impacts of Waste Management and Wood Energy Infrastructure in London, Summary Report to the Greater London Authority, November 2008 (unpublished)

[186] European Commission (2006) Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for the Waste Treatment Industries, August 2006

[187] http://www.airquality.co.uk/archive/laqm/laqm.php

Since emissions from new plant are likely to be carried into an area where objectives are not met, there is a need to identify concentrations that are likely to cause a *significant* added problem. As a starting point or central assumption, a significant additional burden was considered in the analysis to be an increase in NO_2 concentration of 0.04 µg.m^{-3} of NO_2 (or 0.1% of the existing air quality objective). The influence of stack height on the results was also investigated for plant that had a more significant impact on air quality.

Table 70 shows the proportion of London that was considered potentially suitable for plant development for the two type of waste facility assessed as part of the study. These results suggest that from the perspective of impacts on air quality, over half of the area of London might be suitable for a MBT (biodrying) and gasification plant whilst just over a third would be suitable for a MBT (AD) plant assuming a stack height of 20 metres. Increasing the stack height has the effect of making a greater proportion of London suitable for plant development.

The study concluded that there are large areas of London where waste treatment plant could be located with minimal effect on air quality objective attainment. Isolated plant of the types considered - if managed and operating as designed - were therefore considered to be unlikely to have a significant effect on air quality where objectives are not forecast to be exceeded in the future.

Table 70: Proportion of London Potentially Suitable for Plant Development

Plant type	Stack height (m)	Permitted increment in NO_2 relative to objective in areas with current exceedance		% of London suitable for plant development
		Concentration µg.m^{-3}	% of objective	
MBT (Biodrying) and Gasification	20	0.04	0.1%	63%
MBT (AD)	20	0.04	0.1%	36%
	10	0.04	0.1%	29%

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