

# Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis

Appendices to the Main Report

May 2007

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## A.1.0 DETAILS OF MASS FLOWS FOR COLLECTION SCENARIOS

Table 1: Collection Model Description Summaries, >200m<sup>2</sup>

	Dry recycling	Biowaste	Refuse	Costs
>200 base case	Fortnightly 240l bin, capture norm, 5% reject	None	Weekly 240l	
>200 garden only	Fortnightly 240l bin, capture high, 5% reject	Fortnightly (alternating) 240l	Fortnightly (alternating) 240l	£19/t open air windrow
>200 all biowaste	Fortnightly 240l bin, capture high, 5% reject	Fortnightly (alternating) 240l for those with gardens, kerbside bucket for those without.	Fortnightly (alternating) 240l	£35/t in vessel composting (IVC)
>200 kitchen only (kitchen AD, HWRC green windrow)	Fortnightly 240l bin, capture high, 5% reject	Weekly, kitchen caddy & kerbside bucket	Fortnightly (alternating) 240l	£60/t AD, £19 windrow → £42.70
>200 kitchen only (IVC)	Fortnightly 240l bin, capture high, 5% reject	Weekly, kitchen caddy & kerbside bucket	Fortnightly (alternating) 240l	£40/t IVC
>200 kitchen only (AD)	Fortnightly 240l bin, capture high, 5% reject	Weekly, kitchen caddy & kerbside bucket	Fortnightly (alternating) 240l	£55/t AD
>200 kitchen plus (kitchen AD, HWRC green windrow)	Fortnightly 240l bin, capture high, 5% reject	Weekly, kitchen caddy & kerbside bucket. More home composting.	Fortnightly (alternating) 240l	£60/t AD, £19 windrow → £44.05
>200 kitchen plus (IVC)	Fortnightly 240l bin, capture high, 5% reject	Weekly, kitchen caddy & kerbside bucket. More home composting.	Fortnightly (alternating) 240l	£40/t IVC
>200 kitchen plus (AD)	Fortnightly 240l bin, capture high, 5% reject	Weekly, kitchen caddy & kerbside bucket. More home composting.	Fortnightly (alternating) 240l	£55/t AD
>200 base case, vehicle sort	Weekly 44l box and kerbside sort stillage vehicle, capture basic.	None	Weekly 240l	
>200 kitchen only (IVC), kerbside sort	Weekly 44l box and kerbside sort stillage vehicle, capture plus.	Weekly, kitchen caddy & kerbside bucket	Fortnightly 240l	£40/t IVC
>200 kitchen only (AD), kerbside sort	Weekly 44l box and kerbside sort stillage vehicle, capture plus.	Weekly, kitchen caddy & kerbside bucket	Fortnightly 240l	£55/t AD

Table 2: Collection Model Description Summaries, <200m<sup>2</sup>

	Dry recycling	Biowaste	Refuse	Costs
<200 base case	Weekly sack, capture ≈norm, 5% reject	None	Weekly sack	
<200 garden only	Weekly sack, capture ≈high, 5% reject	Fortnightly (alternating) 240l	Fortnightly (alternating) 240l	£19/t open air windrow
<200 all biowaste	Weekly sack, capture ≈high, 5% reject	Fortnightly (alternating) 240l for those with gardens, kerbside bucket for those without.	Fortnightly (alternating) 240l	£35/t in vessel composting (IVC)
<200 kitchen only (kitchen AD, HWRC green windrow)	Weekly sack, capture ≈high, 5% reject	Weekly, kitchen caddy & kerbside bucket	Fortnightly (alternating) 240l	£60/t AD, £19 windrow → £42.96
<200 kitchen only (IVC)	Weekly sack, capture ≈high, 5% reject	Weekly, kitchen caddy & kerbside bucket	Fortnightly (alternating) 240l	£40/t IVC
<200 kitchen only (AD)	Weekly sack, capture ≈high, 5% reject	Weekly, kitchen caddy & kerbside bucket	Fortnightly (alternating) 240l	£55/t AD
<200 kitchen plus (kitchen AD, HWRC green windrow)	Weekly sack, capture ≈high, 5% reject	Weekly, kitchen caddy & kerbside bucket. More home composting.	Fortnightly (alternating) 240l	£60/t AD, £19 windrow → £43.64
<200 kitchen plus (IVC)	Weekly sack, capture ≈high, 5% reject	Weekly, kitchen caddy & kerbside bucket. More home composting.	Fortnightly (alternating) 240l	£40/t IVC
<200 kitchen plus (AD)	Weekly sack, capture ≈high, 5% reject	Weekly, kitchen caddy & kerbside bucket. More home composting.	Fortnightly (alternating) 240l	£55/t AD
<200 base case, vehicle sort	Weekly 44l box and kerbside sort stillage vehicle, capture ≈basic.	None	Weekly sack	
<200 kitchen only (IVC), kerbside sort	Weekly 44l box and kerbside sort stillage vehicle, capture ≈plus.	Weekly, kitchen caddy & kerbside bucket	Fortnightly 240l	£40/t IVC
<200 kitchen only (AD), kerbside sort	Weekly 44l box and kerbside sort stillage vehicle, capture ≈plus.	Weekly, kitchen caddy & kerbside bucket	Fortnightly 240l	£55/t AD

## A.2.0 COSTS OF BIOWASTE COLLECTION IN OTHER EU STATES

### A.2.1 Italy

In order to allow a comparison among different collection systems, the Research Group on Composting and Integrated Waste Management at Scuola Agraria del Parco di Monza led a survey on the costs of the different collection systems run in Italy. Collection systems were grouped mainly by the way food waste is collected. The three system groups are as follows:

- **traditional source separation** based on the use of plastic bags or road containers (up to 3.3 m<sup>3</sup>) for mixed MSW and *source separation through road containers only for dry recyclables (paper, glass, plastics)*. The food waste is not sorted and is delivered along with the residual waste. Residual waste therefore remains to a large degree fermentable (especially because the food waste element is further ‘concentrated’ in residual waste due to the sorting of paper, cardboard, glass, plastics, etc.) and has to be collected frequently.
- **intensive source separation, including that of food waste**, based on *road containers (120-240 litres, up to 3.3 m<sup>3</sup>) both for food waste and dry recyclables*; the collection of residual waste is through road containers. This is usually referred to as the ‘double container’ collection (besides that for residual waste, households can use one for food waste). This system is quite widely diffused in Central Italy (Emilia, Tuscany) and has been the most common system, so far, also in Spain (e.g. Cordoba, Catalunya).
- **intensive source separation, including that of food waste, with collection at the doorstep – otherwise worded as door-to-door (DtD) – of food waste and residual waste**. In general, also some high-yield dry recyclables are collected with a DtD system (usually paper and cardboard, due to the much higher capture per person than with road containers). This is the most common system in those Italian Municipalities and Provinces where the highest recycling rates are achieved (up to 70% in single Municipalities, and above 50% in Districts, with Lecco, the first Province to reach 50%). Also Catalunya lately reported some successful attempts where this scheme has been implemented (see below).

The outcomes of the survey follow.

#### A.2.1.1 Traditional Collection Systems

Table 3 reports on the costs of traditional collection (with £ = roughly 3000 ITL<sup>1</sup>). The data show that the total waste management costs (including disposal) fluctuate widely because of the different disposal fees charged in different regions.

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<sup>1</sup> Italian Lire

Therefore, in order to evaluate the competitiveness and draw reliable conclusions *it is necessary to focus on collection and transport costs*, disregarding disposal costs, at least until National and European Regulations (e.g. the EC landfill Directive) effectively make the cost of disposal more uniform across different sites.

The results also indicate that data expressed in cost per unit mass (ITL/kg, or equivalently, £/tonne) tend to penalise those municipalities whose specific (i.e. per household or per inhabitant) production of waste is lower. The average collection and transport costs of the three municipalities with waste arisings below 350 kg.person<sup>-1</sup>year<sup>-1</sup> is ITL 253/kg, while municipalities with more than 500 kg.person<sup>-1</sup>year<sup>-1</sup> have costs of ITL 134/ kg. But in absolute terms, *these must dispose of more waste, and overall waste management cost tends to be higher*. Generally, in systems which capture higher quantities of material per pick up, the costs per unit weight can be expected to be lower. Arguably, this is one reason why UK refuse collections look so cheap when compared in this way with some other EU countries. Costs per household probably show less variation, but with quantities per household collected as refuse being lower in many other countries.

The per capita cost of collection + transport (without disposal) for traditional collection of mixed MSW averages some ITL 66.000 (or around £22 per inhabitant).

### A.2.1.2 Collection Systems with Source Separation of Food Waste

As mentioned above, these systems can be grouped into two categories:

- door to door (DtD) collection systems
- road container collection systems

The study focused on mature systems (run for at least two years), mainly concentrated in Northern Italy, even though during last 2 years many schemes have been implemented also in Southern Regions. Table 4 and Table 5 summarise the costs of the service. As previously noted, what matters is the average cost for collection + transport per person; we have highlighted it in both tables with an emboldened font.

The results indicate that collection schemes based on the use of road containers (whether for mixed MSW or separate food waste) show a higher specific waste production than schemes where small waste bins and buckets are given to single households (DtD collection). This is partly due to industrial waste being delivered into big containers, but also to a much higher quantity of garden waste that can enter big road containers than in DtD schemes where food waste is collected – at detached houses - though small buckets which are owned by the householder

Table 3: Municipalities with a 'Traditional' Source Separation System Only for Dry Recyclables

Municipality/District	Population	Average yearly MSW production (kg/inh.y)	Collection + transport cost (ITL/inh. y)	Disposal cost (ITL/inh. y)	Total cost (ITL/inh. y)	Collection + transport cost (ITL/kg)	Total cost (ITL/kg)
Venezia 4 District (3 municipalities)	n.a.	408	62,157	46,286	108,443	152	266
Priula District (3 municipalities)	36,575	412	45,064	54,203	99,267	109	241
Verona province (38 municipalities)	n.a.	439	61,090	51,287	112,377	139	256
Verona town	254,000	470	n.a.	n.a.	159,123	n.a.	339
Caravaggio (before implementing the scheme)	14,180	453	112,065	75,609	187,674	247	414
Bergamo province (3 municipalities)	8,224	536	63,405	96,095	159,499	118	298
Cinisello B.	78,000	n.a.	59,751	n.a.	n.a.	n.a.	n.a.
Pescara	122,236	436	73,743	48,006	121,749	169	279
Cepagatti	7,870	478	65,082	51,970	117,052	136	245
Popoli	5,855	443	44,309	18,043	62,352	100	141
Vasto	5,000	409	45,000	n.a.	n.a.	110	n.a.
Cupello	3,500	275	63,000	n.a.	n.a.	229	n.a.
Macerata	41,936	407	63,338	40,101	103,439	156	254
Termoli	30,100	520	65,620	18,765	84,385	126	162
Campobasso	51,518	412	79,310	34,532	113,842	193	277
Alghero	40,477	508	104,726	54,352	159,078	206	313
Quartu	61,500	505	87,138	46,732	133,870	172	265
Guspini	13,400	349	45,522	20,896	66,418	130	190
Montagnareale	1,800	194	52,633	9,779	62,412	271	321
Librizzi	2,020	379	73,855	12,376	86,231	195	227
S. Piero Patti	3,664	396	62,901	15,881	78,782	159	199
<b>AVERAGE</b>		421	<b>66,485</b>	41,272	112,373	156	261

NOTE: the average of the sums (average total cost) doesn't match with the sum of average values (average collection and transport + average disposal cost), as they are slightly affected by data not available

Table 4: Systems with Source Separation of Food Waste By Means of Road Containers

Municipality/District	Population	Average annual MSW production (kg/inh.y)	Collection + transport cost (ITL/inh.year)	Disposal cost (ITL/inh.year)	Total cost (ITL/inh.year)	Collection + transport cost (ITL/kg)	Total cost (ITL/kg)
Venezia 4 District (6 Municipalities)	n.a.	445	54,417	44,060	98,477	122	221
Verona Province (7 Municipalities)	41,167	447	66,407	47,369	113,776	149	255
<b>AVERAGE</b>		446	<b>60,367</b>	45,714	106,126	135	238

Table 5: Systems with Door to Door Separation for Food Waste

Municipality/District	Population	Average annual MSW production (kg/inh.y)	Collection + transport cost (ITL/inh. y)	Disposal cost (ITL/inh. y)	Total cost (ITL/inh. y)	Collection + transport cost (ITL/kg)	Total cost (ITL/kg)
Venezia 4 District (4 Municipalities)	n.a.	321	53,733	31,558	85,291	167	266
Verona Province (7 Municipalities)	63,697	310	61,389	25,013	86,402	198	279
Padova 1 Basin (26 Municipalities)	206,000	322	52,500	25,182	77,682	163	241
Province Bergamo (7 Municipalities)	20,013	n.a.	45,821	62,954	108,775	n.a.	n.a.
Calcio	4,765	393	31,266	61,032	92,298	80	235
Caravaggio (after implementing the scheme)	14,181	n.a.	38,079	n.a.	n.a.	n.a.	n.a.
Cinisello B..	78,000	422	55,620	n.a.	n.a.	124	n.a.
Treviglio	25,294	457	n.a.	n.a.	158,310	n.a.	346
Cameri	9,567	382	n.a.	n.a.	83,521	n.a.	219
Castiglione	4,691	234	48,658	n.a.	n.a.	208	n.a.
Cupello	3,500	275	52,000	n.a.	n.a.	189	n.a.
<b>AVERAGE</b>		346	<b>48,401</b>	41,148	98,897	161	264

NOTE: the average of the sums (average total cost) doesn't match with the sum of average values (average collection and transport + average disposal cost), as they are slightly affected by data not available.

The traditional collection systems based on separation of dry recyclables only by means of road containers shown in Table 4 shows a higher cost per inhabitant than systems with a source segregation of food waste. This is partly due to higher collection frequencies in Southern Italy (up to 6 times a week) that affect average costs, particularly since many case studies from Southern Italy are included in the Table and the climate there necessitates more frequent collection of putrescible materials.

A key outcome is that *the average collection and transport costs (per person per year) tends to be lower in schemes where source segregation of food waste uses DtD systems, than where road containers are used, and lower also than systems where no food waste is collected.*

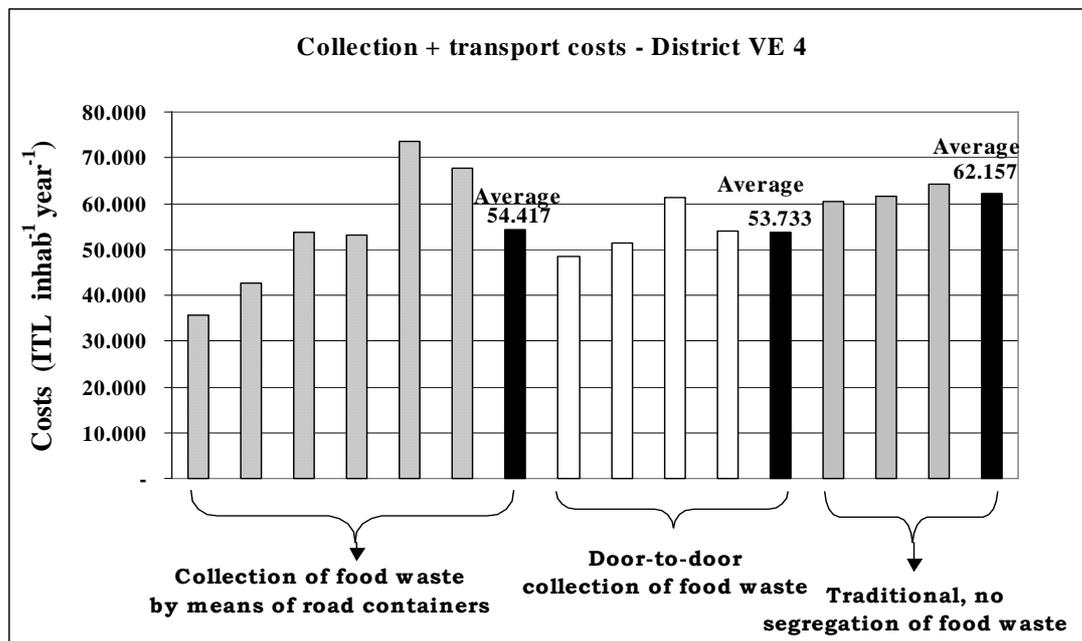
### A.2.1.3 Venezia 4

It could be argued that the above discussion is somewhat biased by the inclusion of many examples from Southern Italy, in the costs of the road container schemes, as in this instance the climate necessitates more frequent refuse collections. Scuola Agraria carried out a more focused analysis in the district Venezia 4, close to Venice (Figure 1) on the relative costs of the different systems in a relatively uniform climate.

This shows once again that source segregation of food waste with door-to-door schemes can be run with no substantial increase in overall cost, and sometimes costs are even lower than with traditional collection (no segregation of food waste) or with food waste segregation by means of road containers.

To understand the unexpected outcomes of the survey, we must underline that if source separation of food waste is *added* to that of co-mingled municipal waste, with no modification in the previous scheme for MSW collection, total costs are bound to rise. This actually tends to happen with the segregation of food waste by means of road containers. But this does not happen when food waste collection is *introduced in such a way that the overall collection system is optimised. Intensive door-to-door schemes for food waste – when made “comfortable” for householders to use - yield high captures.* This reduces the percentage of food waste in residual waste, which can then be collected less frequently with fewer complaints regarding odours.

Figure 1: cost comparison (ITL.inhab<sup>-1</sup>.year<sup>-1</sup>) for different collection schemes in a single District



## A.2.2 Modelling Carried Out in the UK Context

The prognosis for the future suggests that the background cost for dealing with residual waste, against which planning and resource allocation decisions should be made, is unlikely to be much below £60 per tonne, if indeed it is this low.

At this cost, most activities in respect of source separation, and source reduction, make financial sense since the avoided costs of collection of refuse and its disposal are significant. Elsewhere in Europe, experience suggests that once disposal costs reach €85-€100 (£60-£70) per tonne, the justification for high capture recycling systems can be established on grounds of cost.

It is interesting to note what happens to different collection systems as costs for dealing with residual waste change. The three figures below illustrate how, as disposal costs rise, high capture recycling and composting systems become cost competitive with a system which simply collects refuse and disposes of it in landfill. <sup>2</sup>

The four collection systems represented are:

1. Refuse collection only;
2. Dry Recyclables Only (weekly, on vehicle sort);
3. Weekly Dry Recyclables, *Weekly Kitchen Wastes*, Fortnightly Refuse;
4. Weekly Dry Recyclables, *Fortnightly Kitchen and Garden Wastes* and Fortnightly Refuse.

<sup>2</sup> The following is adapted from D. Hogg and J Hummel (2002) *The Legislative Driven Economic Framework Promoting MSW Recycling in the UK*, Final Report by Eunomia Research & Consulting for the National Resources and Waste Forum, [www.nrwf.org](http://www.nrwf.org).

The costs per household for each, assuming landfill disposal of residues and a gate fee (pre tax) of £16 per tonne, are shown for both low (light colour) and high (dark colour) captures. The three figures indicate that as landfill tax increases progressively from £15 per tonne to £25 per tonne to £35 per tonne, the costs of the systems become progressively more similar. Some points are worthy of note:

- First, at low disposal costs, the introduction of a system for dry recyclables increases the costs relative to the 'refuse only' system. But adding on biowaste collections makes almost no difference as long as the collection frequency for residual waste is reduced.
- As the disposal costs rise, the differentials between the source separation systems and the 'refuse only' systems become narrower, turning marginally negative in some cases. This confirms the point above concerning the fact that the costs for disposal only and those for high capture systems, are likely to converge in a period no longer than seven years, and probably much less than this since the 'shadow' disposal cost for local authorities is no longer that of landfill, but a 'LATS-adjusted' residual waste treatment cost, which is likely to be already high by 2009/10.
- As the disposal costs rise, because the biowaste treatments are lower in cost than the residual waste treatments, the higher the capture, the lower the cost. This is the reverse of the situation in which disposal costs are low. This is because although the costs of collecting more material increase the collection costs per household, the more efficient logistics reduce costs on a 'per tonne' basis. This improvement in efficiency is offset by 'avoided costs of disposal' at higher disposal costs. At low disposal costs, the 'avoided costs of disposal' are so low that the higher collection costs are not matched by the avoided costs of disposal.

The clear implications are that in future:

- All systems will cost more than they do today; and
- The increase in disposal costs makes separate collection systems more attractive from a cost perspective.

Increasingly, as disposal costs rise, high capture separate collection systems will become cost competitive.

The analysis does, however, suffer some shortcomings. In particular, there was no allowance made in the modelling above for the increase in waste arisings which is likely to occur where garden waste collections are used. This means that the costs for the kitchen and garden waste collection system are likely to be understated in the above analysis.

Figure 2: Per Household System Costs at £15 Landfill Tax

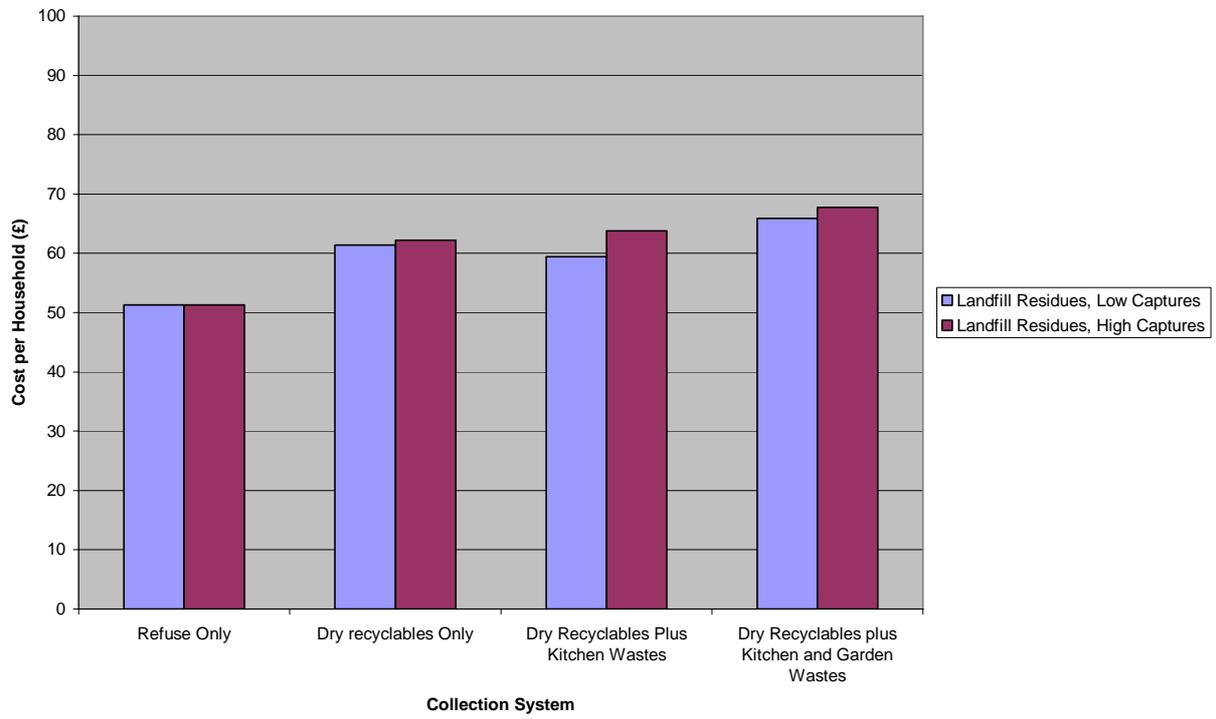


Figure 3: Per Household System Costs at £25 Landfill Tax

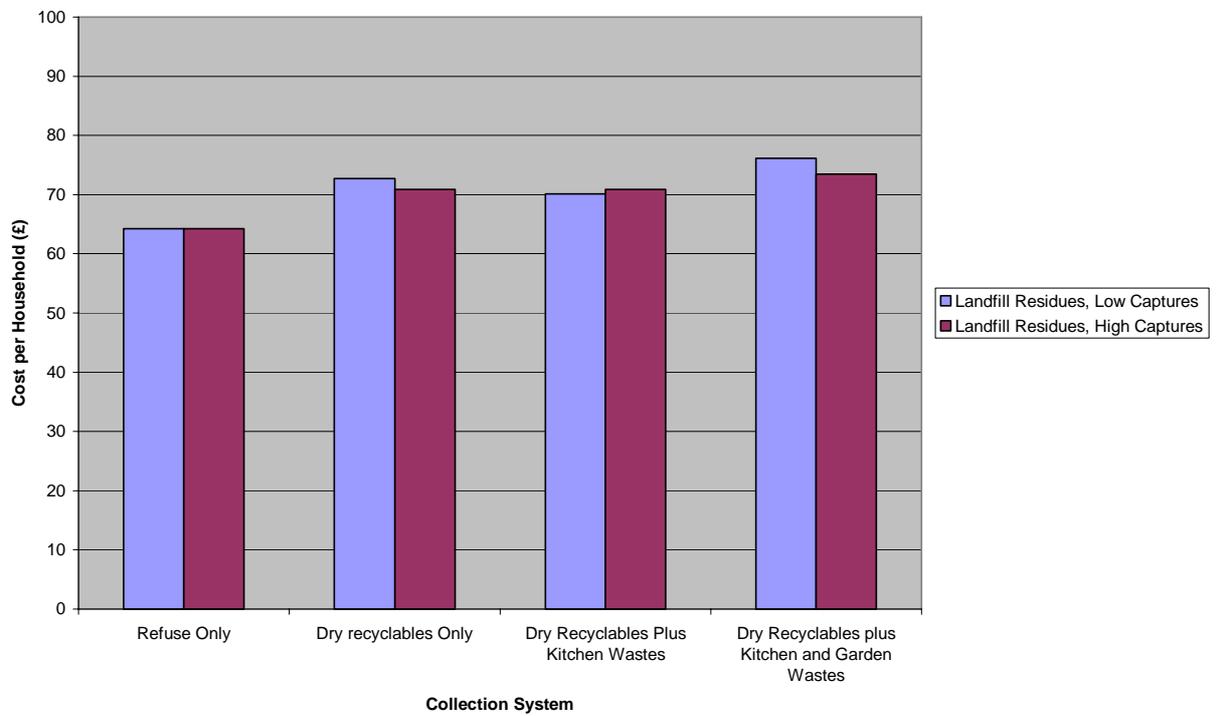
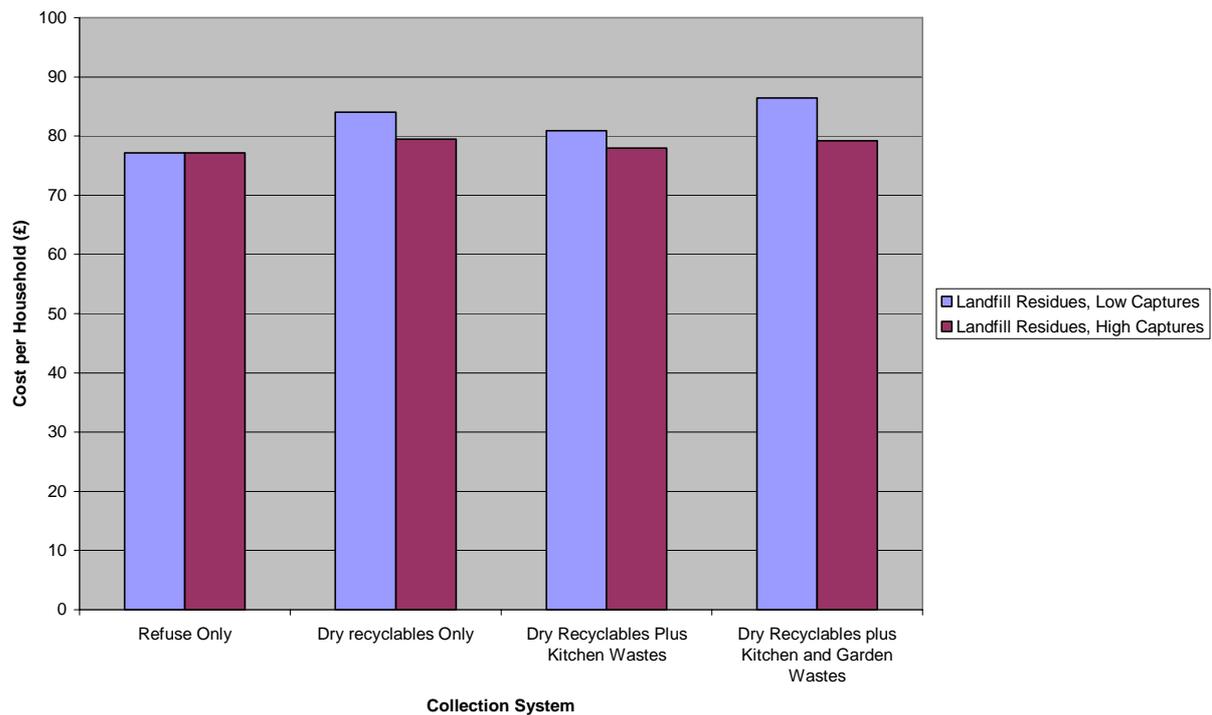


Figure 4: Per Household System Costs at £35 Landfill Tax



### A.2.3 Germany

A recent study undertaken in Germany by INFA also investigated the issues affecting the costs of biowaste collection relative to a situation in which biowaste was not separately collected.<sup>3</sup> The report effectively compares:

- Co-mingled collection of residual+biowaste with
- Separate collection of residual waste (Restabfall) and Biowaste (Biotonne)

It was assumed that:

- In Rural areas:
  - collection of all waste types is performed with compacting vehicles (side loaders or rear end loaders);
  - frequencies of collection are lowered when moving from co-mingled collection to residual+biowaste collection to constrain any additional costs;
  - the “wet/dry” scheme implies an extra-effort in the logistic structure
  - The authors explicitly cite the danger of garden waste transfer into the biowaste container. For separate collection of biowaste, typical increases are estimated at 10kg/inhab;
- In Urban areas:
  - collection of all waste types is performed with compacting vehicles (rear end loaders);

<sup>3</sup> INFA (2004) *Cost Consideration of Separate Collection and Treatment of Biowaste* Final Report for Verband der Nordrhein-westfälischen Humus- und Erdenwirtschaft e. V. 19th Nov. 2004

- frequencies of collection are halved when moving from co-mingled collection to residual+biowaste collection to constrain any additional costs;
- the “wet/dry” scheme implies an extra-effort in the logistic structure; and
- the authors claim that specific results for biowaste (i.e kg/inhab) are relatively low.

The key conclusions from the study were as follows:

1. The authors note:

*According to the investigations, besides the logistic preconditions (vehicle technology, collection frequency etc.) the difference between the disposal cost for the rest waste and the treatment costs of biowaste is decisive. The influence of the mentioned parameters are mentioned in the following figure. Contrary to urban structures with a required biowaste treatment cost difference of at least 55 - 60 €/Mg (biowaste treatment more favourable in costs than treatment of residual waste respectively household waste), in rural structures a difference of approximately 20 - 25 €/Mg is economically sensible. [see Figure 5].*

2. The authors note that another significant influence on costs is the capture rate. They argue that their example of a "very urban structure", where they posit an extremely low collection rate of 10 kg/inh/yr, implies that introduction of the biobin is only economically sensible where the treatment cost for biowaste is over 150 €/tonne less than for residual waste. They state:

*Here it must be questioned why such small amounts are collected and which political, logistical or legal requirements have to be considered here.*

Conversely, savings of approximately 4 €/inh/yr in a rural area and approximately 1 €/inh/yr in an urban area were calculated for two typical disposal areas (rural / urban) considering figures for services and costs usual in the waste sector (see Figure 6).

Figure 5: Maximum Difference Between Treatment Costs (household waste/residual waste respectively biowaste) for a Cost-neutral Introduction of the Biobin Which is Neutral in Costs (logistic and treatment costs, incl. ranges) Subject to Settlement Structure and Capture Rate

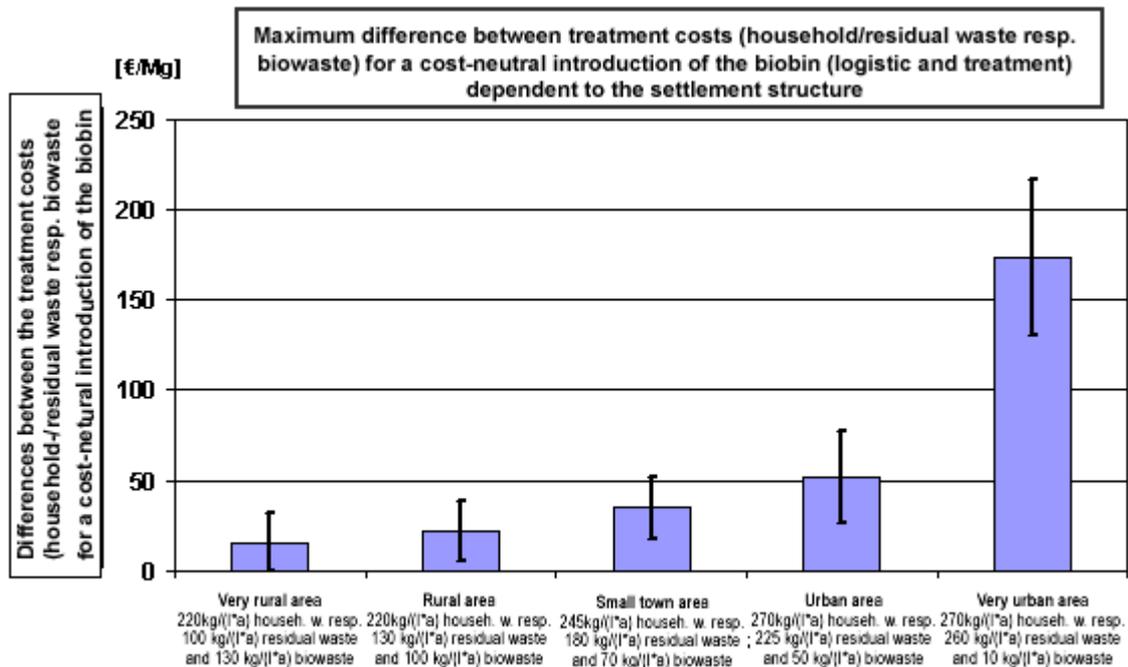


Figure 6: Costs for Logistics and Treatment of a Segregated Residual and Biowaste Collection Compared to an Exclusively Household Collection (rural / urban disposal area)

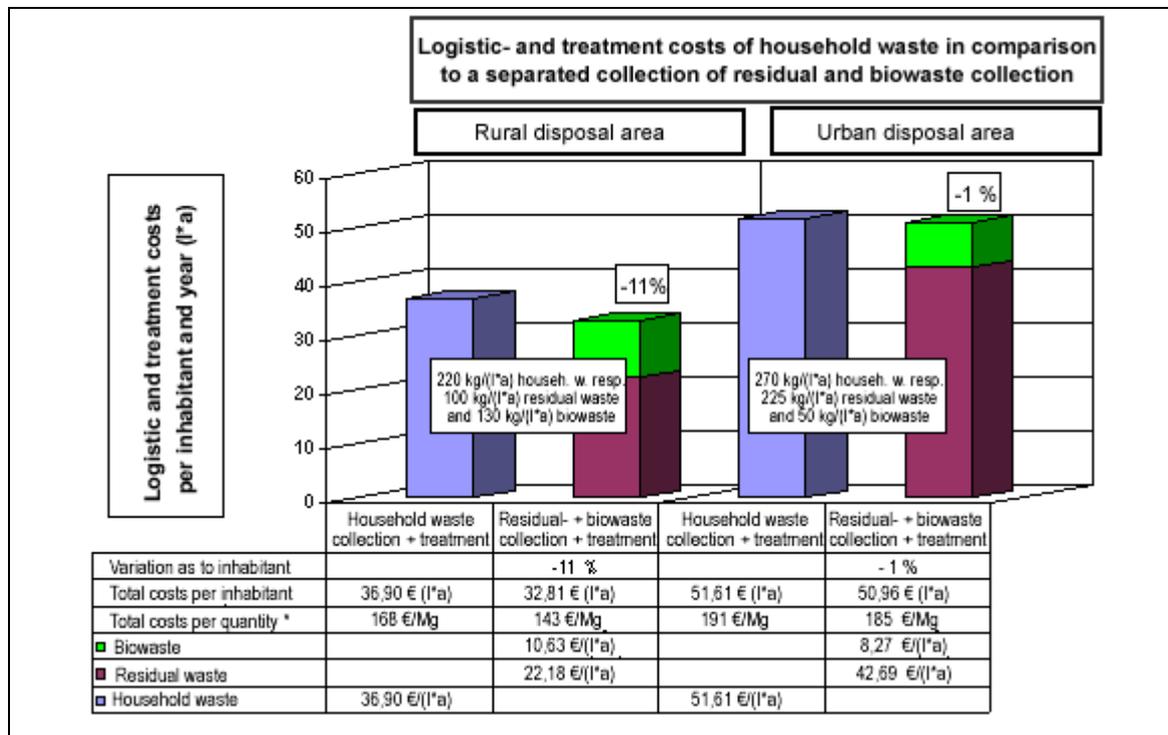


Table 6: Costs for Rural and Urban Areas for Mixed Waste Collection vs Refuse and Biowaste Collection

	Mixed MSW	Residual	Biowaste	R+B	Delta
<b>RURAL AREA</b>					
kg/inhab/y	220	130	100	230	5%
coll/week	1	0.5	0.5	1	
<b>coll/cost (€/inhab/y)</b>	<b>9.4</b>	<b>5.93</b>	<b>4.63</b>	<b>10.56</b>	<b>12%</b>
treat./cost (€/inhab/y)	27.5	16.25	6	22.25	-19%
treat./cost (€/t)	125	125	60		
<b>total cost (€/inhab/y)</b>	<b>36.9</b>	<b>22.18</b>	<b>10.63</b>	<b>32.81</b>	<b>-11%</b>
<b>URBAN AREA</b>					
kg/inhab/y	270	225	50	275	2%
coll/week	1	0.5	0.5	1	
<b>coll/cost (€/inhab/y)</b>	<b>17.88</b>	<b>14.56</b>	<b>5.27</b>	<b>19.83</b>	<b>11%</b>
treat./cost (€/inhab/y)	33.75	28.13	3	31.13	-8%
treat./cost (€/t)	125	125	60		
<b>total cost (€/inhab/y)</b>	<b>51.63</b>	<b>42.68</b>	<b>8.27</b>	<b>50.96</b>	<b>-1%</b>

3. In conclusion, the authors remark:

*As a result it can be stated that in many areas a cost-neutral introduction of the biobin can be realised, whereby (at sufficient capture rates of the compostable material) the introduction of the biobin leads to reduced costs (collection and treatment). The level of the changing cost figures depends - besides some logistic preconditions - especially on the cost difference between household waste/residual waste treatment and the biowaste treatment.*

What is not considered in the study is that in urban areas, the lower interception rates for biowaste are due to the proportionately lower quantity of garden waste and a higher proportion of food waste in the collection system. In such situations, only an intensive collection scheme for food waste could ensure relatively high captures of biowaste. Using biotonnes (wheeled bins for biowaste) is unlikely to be either sensible or cost-effective in dense urban areas given the relative absence of garden waste, and given also the constraints on storage space, and space at the kerb.

The study also indicates very small increases in collected quantities in the two situations where biowaste containers are introduced. This may reflect a scheme in which the biotonne is charged for.

What the study rightly points out, however, is that, subject to waste quantities not increasing by much, the interception rates for biowaste are important drivers for effective collection logistics. However, even where logistics are inefficient, there may be some financial rationale for the collection if the avoided costs of disposal are far in excess of the costs of treating the biowaste.

## A.3.0 DESCRIPTION OF COST MODELLING

### A.3.1 Introduction to the Model

Eunomia Research & Consulting's proprietary waste collection cost model, HERMES, is a sophisticated spreadsheet based tool that allows a wide range of variables to be accounted for, and which enables the optimisation of scenarios to accurately reflect local circumstances. It can be used to run a range of scenarios so as to select best practice with a trade off against system costs and other indicators of potential interest.

The recycling/composting performance of each collection system scenario is built up by specifying a range of performance parameters for each component of the system. Performance parameters include weight and volume of material collected by current systems, residual composition, the materials targeted by each collection service, the number of households that the service is available to, the participation rate of those households and the recognition rate achieved from participating households for the materials targeted.

Costs are accumulated by the model from cost data extracted from a database within the model. The model calculates the numbers of vehicles, containers, and crew required and multiplies these by their unit costs. Disposal costs and net cost/income from material sales are also calculated and included. Finally the model adds overheads for management and administration, depot costs, and insurances and financing. Although capital requirements are shown in the model, annual costs are based on the amortised cost of capital using depreciation periods and interest rates as are most standard in the industry.

### A.3.2 Structural Methodology to the Modelling Process

#### A.3.2.1 Inputting of Baseline Data

To set up a baseline against which the alternate scenarios can be compared, a number of standard identifiers were programmed into the model to describe a typical local authority:

- **Population / Household Numbers.**
- **Numbers of Houses by Type of Housing Stock.** Types of houses were simplified into properties with gardens (90%) and those without.
- **Waste Arisings Tonnage Data.** Tonnage data used in the model included material collected as part of standard household waste collections as identified by the mass flow calculations of household material production.
- **Composition Data.** This was based upon the standard WRAP composition<sup>4</sup> though was modified where the scenario resulted in different amounts of putrescible material passing to the waste collection systems.
- **Travel Time by Housing Type.** Travel speeds between houses were estimated based on known collection times in real authorities. The average speed with which

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<sup>4</sup> Dr Julian Parfitt (2002) *Analysis of Household Waste Composition and Factors Driving Waste Increases*, WRAP, December 2002

collection vehicles are able to progress around their rounds is primarily a function of the housing density and traffic and road conditions. Although these are subject to large variations it is assumed that there is correlation to housing type: where housing is denser it takes less time to get to each house but average speeds are lower as vehicles need to keep stopping and re-starting. The values used in this modelling were transferred from other modelling work in real local authority situations.

- **Participation, Recognition and Set-Out Rates.** The participation and set-out rates were estimated in line with typical known characteristics for particular systems, and comparative performance gains were identified when switching to the alternate systems.

### A.3.2.2 Collation of Background Data Relevant to the Modelling

- **Vehicle specifications and performance data.** Actual cost and performance data for a range of vehicle types that are either being used in current collection systems, or that are deemed probable options for future collection systems have been obtained from manufacturers and entered into the model over time and are not specific to this project. Performance parameters include working payloads by weight and volume, required or average crew sizes, fuel efficiency, capital cost, and type and number of types of materials able to be separately collected.
- **Container specifications and costs.** Actual cost and performance data for a range of container types that are either currently being used in the collection systems or that are deemed probable options for future collection systems have, over time, been obtained from manufacturers and entered into the model. Performance parameters include container volume, life expectancy, and unit cost.
- **Personnel costs.** Personnel costs are based on known industry costs used in collection contracts. Costs for a range of positions have been obtained through discussions with operators and DSO management and entered into the model including management, supervisors, administrative support, HGV drivers, Non-HGV drivers, collection crew and yard crew.
- **Depot and overhead costs.** Overhead costs including insurance, performance bonds, administrative support, rent, legal and accounting costs were estimated and entered into the model. Depot costs are calculated based on estimated rental/lease costs, building maintenance, site works, machinery operation and site personnel costs. Costs for these elements were considered to be shared across all collection systems and were apportioned relative to the tonnages handled by each system.

### A.3.2.3 Selection of Specifications for Each Scenario

- **Materials to be collected by each system.** The model allows for up to four integrated collection systems to be modelled together, comprising up to three collection systems for recyclable or compostable material plus one system for the collection of residual material. Materials specified as being collected by one of the recycling or composting collections are subtracted from the residual component based on the specified capture rates.

- **Capture rates.** Capture rates for each material and housing type were specified by estimating the participation rate for each system and a recognition rate for each material type in each system. In the projected scenarios, the baseline rates were adjusted to account for system design factors that would improve the participation or recognition rates. Factors such as increasing the frequency of a collection or container size, for example, are assumed to yield an increase in capture rate.
- **Frequency of collection.** Specific to each collection system within each scenario.
- **Container type and size.** The type of container to be used for each system was selected from the model's database. The model calculates the available volume of material per household each collection day to allow optimisation of container volumes.
- **Vehicle type and number.** The type and number of vehicles to be used for each collection system was selected from the model's database. As a default, the model automatically calculates the optimum number of vehicles required based on the time available, the number of households to be collected from, and the quantity of material to be collected.

#### A.3.2.4 Calculation and Adjustments

*As the data was inputted into the model, the potential performance of each system was calculated and a range of output data generated, which were used to evaluate each scenario. Following initial input of the above data the models were subjected to an extensive audit and review to ensure all specifications had been correctly entered and that they were an accurate reflection of what could be expected in terms of the performance of each system. Minor adjustments to performance parameters were made to the scenarios before the final results were produced.*

### A.3.3 Performance Assumption Guidelines

The assumptions used to calculate the performance of each scenario are critical to determining the outcomes, particularly when it comes to estimating future capture rates of materials targeted for recycling and composting. There is a potential for bias towards favoured systems even with the best of intentions on the part of the modellers. In order to minimise the risks of bias the following guidelines were established regarding the assumptions used.

#### A.3.3.1 General Modelling Assumptions

- In all cases the long term nature of the scenarios meant that it was assumed that waste education has long been in effect and promotional campaigns are achieving positive results. The performance data was based on experience of known system performance currently achieved by similar and well operated systems in Europe.
- Convenience to the householder is a key factor affecting capture rates. It is assumed that weekly collections are most convenient to the householder and that all other factors being equal, a weekly collection yields a higher total quantity of material than a fortnightly or monthly collection. However, if a weekly system is used for residual waste collections then this negatively impacts the recycling rates for other elements of the system.

- At each stage, checks were conducted to gauge the available space in the container. Spare capacity was considered both for the average recycling household and also higher recycling households as described by a normal distribution curve of recycling performance per individual household. Systems were optimised so that higher recycling households would not suffer a shortage of containerisation space. Similarly, restricting or making residual disposal less convenient was assumed to increase diversion through the other collection systems.
- It is assumed that sufficient processing technology infrastructure is available to enable the processing of co-mingled recyclables and in-vessel composting of kitchen waste. The performance characteristics of these recovery facilities are assumed to be similar to the best performing existing facilities. Costs of these facilities are included in the model simply through the application of a gate fee based on current figures.

### A.3.3.2 Cost Assumptions

- It was assumed that all performance characteristics of vehicles and containers including operating costs, interest rates on capital etc, are as for current systems. Although it is likely that the performance of equipment and systems will, in fact, improve over time, the extent to which this would happen is indeterminable and has therefore not been accounted for.
- A 6.5% cost of capital (interest on vehicles) was used for all systems.
- 5% profit was included on collection and overheads.
- An amortisation period of seven years was assumed for all vehicles with bins and other containers amortised over different periods specific to their expected lifespan and current practices. For wheeled bins a 4% per annum replacement rate was used (10% for boxes) for loss and breakage.
- Inflation was ignored within the costings.

It is assumed that all commodity and treatment prices remain unchanged between the base case and alternative scenario modelling. This assumption does not account for the possibility that recycle values can increase (possible as the market into which these commodities can be sold becomes more diverse and mature) or decrease (if markets weaken as a result of renewed investment in primary production in response to strengthening demand), and also does not show the sensitivity to landfill tax or LATS.

Revenues from material sales are realised and included to be offset against total gross service costs where a box/reusable sack system is used for the collection of dry recyclables. Where co-mingled collections are used, the materials are passed on to a Materials Recycling Facility (MRF) which is assumed to keep the revenues obtained from the sale of the collected materials. The gate fee used for a MRF was set at £35/tonne.

### A.3.4 Notes on Interpretation of Cost & Performance Data

- **Recycling Rate.** The recycling rate percentages quoted represent only part of the material handled by a local authority collection system. They are not equivalent to an estimated BVPI recycling rate, which would also include material from bring sites, street cleansing etc.

- **Bring Systems.** Only kerbside collection systems are included in the model. There is an interrelationship between the amounts of material collected through kerbside and bring systems which has not been modelled. Generally speaking kerbside systems are considered to be more convenient for householders and therefore they will normally divert material away from existing bring systems. However, where bring systems are available alongside existing kerbside systems they will collect some extra material not captured by the kerbside system. This may be partly due to bring systems providing a facility to recycle larger loads of material than are able to be placed in the kerbside system – such as may result from parties or cleanups etc. Bring site effects, however, are assumed to be constant across all scenarios so no impact is attributable.

The projected cost data from the modelling are indicative and ideal for comparative purposes. The system costs at this collection modelling stage of the project do not include the costs of purchasing landfill allowances, sales revenues or fines. This issue is addressed in the context of the report.

## A.4.0 COMPOSTING MODULE

The environmental impacts associated with composting consist of both positive and negative effects. The negative consequences which are most easily captured through economic analysis are the air emissions.

### A.4.1 Defra Health Effects Study

The most recent attempt to estimate air emissions from composting was undertaken in the context of work on the health effects of waste management.<sup>5</sup> The relevant report states:

*'Gaseous emissions from in-vessel systems consist predominantly of carbon dioxide, water vapour and potentially small quantities of ammonia and some volatile organic compounds and bioaerosols (fungi, bacteria, actinomycetes, endotoxin, mycotoxins, and glucans).'*

It also states:

*'Quantitative data on emissions from in-vessel composting facilities is sparse due to the relatively new expansion into this area for municipal derived organic waste processing. Therefore the existing data does not allow figures for comparison with other disposal or treatment facilities to be derived.'*

Whilst this scarcity of data may be true of the UK, it is not entirely true of other countries.

The study does not cite any figures for emissions to air from in-vessel composting systems nor does it give figures for trace substances in composted green and kitchen wastes, these are treated as 'emissions'. Since most of these trace substances are heavy metals, and since the process itself is unlikely to give rise to generation of heavy metals, then actually, these 'emissions' are simply the metals present in the input<sup>6</sup> Given that this material is to be applied to land, it makes sense to ensure quality separation and minimal contamination.

The study based estimates of leachate emissions from the composting of source separated biowastes on a very different process, which deals with fines from a system aimed at creating RDF. This is clearly an erroneous approach, since the compared process was treating mixed waste. The study itself notes:

*'The processing of RDF fines in this dataset also makes it unusual.'*

And also:

*'In practice, some or all of the leachate may be re-circulated in the process.'*

Despite these observations, estimates of emissions from in-vessel composting of source separated biowastes were based on the assumption that 75 litres of leachate would be produced per tonne of waste, and that leachate concentrations would be the same as for a process based upon the composting of mixed wastes.

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<sup>5</sup> Enviro, University of Birmingham, RPA Ltd., Open University and Maggie Thurgood (2004) *Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes*, Final Report to Defra, March 2004.

<sup>6</sup> There is no discussion as to whether these heavy metal emissions might be 'available' to plants or susceptible to leaching.

There is, therefore, very little in the Health Effects study which could reasonably be used to characterize emissions from in-vessel composting other than those emissions reported from open-air windrows which might be expected to be the same at in-vessel facilities.<sup>7</sup>

As regards open-air windrow facilities, the study suggested an enormous range of emissions of CO<sub>2</sub> (100kg-482kg per tonne input waste). The low end of these figures would clearly not withstand any interrogation since the loss of carbon implied (of the order 27kg per tonne of waste) would suggest such a low level of mass reduction and stabilization of normal feedstocks as to render the figure obviously wrong, or 'wrong' where no qualification is offered (despite which, the data pedigree is deemed to be 'moderate' in the study). The same study sites CO<sub>2</sub> emissions – of 'moderate pedigree' – for MBT processes which are said to generate 181 kg CO<sub>2</sub> per tonne of waste. It is difficult to see how, or why, mixed waste would generate so much more CO<sub>2</sub> under biodegradation than source separated organic wastes, given that in MBT processes, it is unlikely that more than 40% or so of material would be putrescible.

The study also suggests another enormous range for ammonia emissions (5-120g/tonne waste). The low figure would probably be achieved only at facilities with a biofilter in place, a feature which is not common in open-air windrow facilities, unless the input mix contained almost no nitrogen (in which case, the process would be compromised).

Figures for emissions of particulate matter and 'chloride' are also given, but their provenance is not clear. For particulate matter, the range is narrower at 163-186g/tonne, and for chloride, one figure – 2 g/tonne – is given. For the emissions other than CO<sub>2</sub>, data pedigree is categorized as 'poor'. Figures are also given for VOC emissions from an Environment Agency report.<sup>8</sup> Leachate figures given are also quoted, though again, the pedigree of the data is considered to be poor.

Generally, the study provides a limited basis for understanding the emissions from compost facilities, and as a result, the likely environmental consequences.

For this reason, we have reviewed other studies to understand emissions from composting. This informs the approach, and outlines the basis for the modelling undertaken.

## A.4.2 Air Emissions from Composting

The air emissions from composting are not straightforward to measure or to present. The gaseous emissions tend to be fugitive in nature. In addition, they can be expected to depend upon a number of inter-related factors:

1. The nature of the input wastes, in particular:
  - a. the nature of the organic carbon in the components of the waste, which determines the biodegradability of the material (and hence, the extent of biodegradation over a give period of time); and
  - b. the nature of any organic compounds in the input wastes which may be released as the mass of material heats up;

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<sup>7</sup> See Enviro and EFTEC (2004) *Valuation Of The External Costs And Benefits To Health And Environment Of Waste Management Options* Final Report for Defra, December 2004; HM Customs & Excise (2004) *Combining the Government's Two Health and Environment Studies to Calculate Estimates for the External Costs of Landfill and Incineration*, December 2004.

<sup>8</sup> Environment Agency (2000). *Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion*, R&D Project Record P1/392/4

2. The nature of the process, and the retention time in that process, as well as the maturation period;
3. The nature and effectiveness of the turning / airflow systems, and the frequency of turning;
4. The regime of management of moisture in the biomass, especially in turned windrow systems;
5. The C:N ratio of the biowaste; and
6. The nature and effectiveness of any measures to control air pollution. Implicitly, this means that gaseous emissions from windrow facilities will be higher for some gases than they will be at enclosed facilities making use of biofilters.

For the composting of source segregated materials, the key issues relate to the nature of the input materials. In particular, the nature of the organic carbon will determine the rate at which the material is broken down by microorganisms. The most rapidly biodegradable materials will be starches sugars and fats, whilst lignin and cellulose are likely to degrade more slowly.

#### A.4.2.1 Carbon Dioxide Emissions

In their study for the Danish EPA<sup>9</sup>, Baky and Eriksson assumed that the compost was well aerated during the whole process and had been allowed to fully mature. Because of the life-cycle nature of the analysis, the time profile of emissions was not accorded any significance. The study also assumed that different types of organic matter were degraded in differing proportions. Heat, CO<sub>2</sub>, and some methane (CH<sub>4</sub>) were assumed to be released, whilst the mature compost was assumed to consist of mainly humus-like substances. Different organic substances were deemed to decompose to differing degrees in accordance with an earlier study.<sup>10</sup> That study reviewed the degradation of different fractions of organic carbon as presented in three other studies. The degradation rates and the rates used are shown in **Error! Reference source not found.**

Komilis and Ham made measurements of emissions from compost plants in the context of work on the development of life-cycle inventories in the US.<sup>11</sup> As in the ORWARE model used by Baky and Eriksson, they sought to relate CO<sub>2</sub> emissions to the nature of the input materials. However, rather than looking at the constituent inputs in terms of the form of the organic carbon in the materials being composted, they looked at more intuitive categories, these being the relative proportions of mixed paper, garden waste and food waste. On the basis of their experiments, they estimated that carbon dioxide emissions, and associated percentage loss of dry matter for the organic fraction, could be calculated from knowledge of these fractions.

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<sup>9</sup> A. Baky and O. Eriksson (2003) Systems Analysis of Organic Waste Management in Denmark, *Environmental Project No. 822*, Copenhagen: Danish EPA.

<sup>10</sup> U. Sonesson (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

<sup>11</sup> Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, pp.1390-1400.

Table 7: Degradation of Organic Matter in a 'Normal Compost' (as % incoming material)

Material	Haug (1993)	Svensson (1987)	Biocycle (1991)	Chosen Value
Chsd (lignin)	0	0		30 <sup>a</sup>
Chfd (sugar, starch)	70	100		80
Chmd (cellulose)	0-90	100	90	90
Fat	50	71		60
Protein	50	81		65

<sup>a</sup> According to other studies, lignin is not decomposed at all. However, Sonesson took the view that white rot fungi could lead to some degradation of lignin during the post composting phase. He followed Wessen (1983), who measured degradation of leaves and straw in topsoil, and gave this a value of 30%.

Source: U. Sonesson (1997) *The ORWARE Simulation Model – Compost and Transport Sub-models*, AFR report 151, Swedish Environmental Protection Agency, March 1997.

It should be noted that Komilis and Ham were effectively working in situations where the intention was to derive what might be described as very mature compost. Their experiments extended until, in their own words:

*“complete” degradation was reached, as measured by the CO<sub>2</sub> flow rate and after ensuring that this was not due to moisture limitations. [...]Nitrogen was added prior to the initiation of the experiments to achieve a C/N ratio of around 30.*<sup>12</sup>

In practice, for a given material input, the emissions from compost facilities themselves are likely to be different depending upon the retention time and the nature of the facility (see below). This, in turn, is likely to relate to the end market for the material. Agricultural uses might cope better with relatively immature composts.

It is interesting to assess the consistency of the approaches of Baky and Eriksson and Komilis and Ham, and to compare these with estimates of CO<sub>2</sub> emissions from compost used in other studies. Schleiss used figures of 266kg/tonne and 281kg /tonne for enclosed automated composting and open composting in covered boxes respectively.<sup>13</sup> Eunomia et al used a figure of 400kg/tonne.<sup>14</sup> Schleiss's figure was based upon a waste composition which was 16.2% carbon, so this amounts to a 45-47% loss of carbon.

<sup>12</sup> Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, pp.1390-1400.

<sup>13</sup> Konrad Schleiss (1999) *Grüngutbewirtschaftung im Kanton Zürich aus betriebswirtschaftlicher und ökologischer Sicht: Situationsanalyse, Szenarioanalyse, ökonomische und ökologische Bewertung sowie Synthese mit MAUT*, Dissertation ETH No 13,746, 1999

<sup>14</sup> Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC Research & Consulting (2002) *Economic Analysis of Options for Managing Biodegradable Municipal Waste*, Final Report to the European Commission.

## Approach Taken in this Study

It is assumed, in this study, that the quantity of emissions of CO<sub>2</sub> is not dependent upon the nature of the facility. What can be said, however, is that the mineralization of CO<sub>2</sub> may occur more quickly or more slowly depending upon the process type and the operating parameters.

The key issue has been to relate the emissions to the input wastes. In this study we have used the approach in which different constituent elements of carbon are taken to degrade at different rates over time. Hence, the modelling of CO<sub>2</sub> emissions is based upon the carbon constituents of the input materials.

Where fresh compost materials are produced (as opposed to very mature compost), CO<sub>2</sub> emissions are deemed to be lower in the process itself. However, the less stable material is mineralized further when applied to land, so in the round, and over an extended period of time, the CO<sub>2</sub> emissions for the combined process 'compost plus land application' are very similar.

The emissions from land applied materials are modelled in time so as to facilitate the use of appropriate discounting methods in the economic evaluation.

### A.4.2.2 Methane Emissions

There is some debate as to whether methane is emitted in any significant quantities at well managed compost sites. If it is, it seems less likely that the level of emissions are related to the input materials and are more likely to be related to the quality of process management, including the size of any composting mass. For example, one study states, regarding experiments undertaken:

*The experiments were conducted under aerobic conditions and measurements of the headspace gaseous composition revealed that methanogenic conditions did not occur. This generally simulates the conditions in actual MSW composting facilities, since, even if anaerobic conditions do develop in the center of the piles, the anaerobic gases emitted would be oxidized at the surface of the piles.*<sup>15</sup>

This is a common view, but one study which has measured positive emissions of methane is that of Scheiss.<sup>16</sup> However, measurements by Schleiss were made close to the hotspot in the centre of the compost pile, where measurements could be expected to be higher than at the surface, if one accepts that some oxidation of methane occurs through the composting biomass (and this seems likely). Schleiss gives a figure of 5.4kg CH<sub>4</sub> per tonne for fully automated composting and 11.1 kg CH<sub>4</sub> per tonne of waste for composting in open air covered boxes. Eunomia et al used a figure of 0.983kg per tonne.<sup>17</sup> Grontmij and IVAM

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<sup>15</sup> Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, p.1394.

<sup>16</sup> Konrad Schleiss (1999) *Grüngutbewirtschaftung im Kanton Zürich aus betriebswirtschaftlicher und ökologischer Sicht: Situationsanalyse, Szenarioanalyse, ökonomische und ökologische Bewertung sowie Synthese mit MAUT*, Dissertation ETH No 13,746, 1999

<sup>17</sup> Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC Research & Consulting (2002) *Economic Analysis of Options for Managing Biodegradable Municipal Waste*, Final Report to the European Commission.

estimate a figure of 0.195kg per tonne for VFG facilities in the Netherlands (based upon some measurements as well as literature review), though they note some uncertainties.<sup>18</sup>

### Approach Taken in this Study

We have assumed zero emissions of methane for the 'low emissions scenario' and a value of 5.4 kg per tonne for the high case. However, we have assumed this high value applies for the case where no biofilter is in place. Where a biofilter is in place, we use a high value of 0.983kg (effectively, a biofilter efficiency of 82%). From the above discussion, these high values may well be outside what should be expected from well managed facilities.

#### A.4.2.3 Nitrogenous Emissions to Air (process emissions, raw gas)

Ammonia emissions are determined by the quantity of ammonium ions, urea, and organically bound nitrogen. The pH value, temperature, ventilation, and the C/N-relation constitute other influencing factors. High C/N relations cause NH<sub>3</sub> emissions to diminish.<sup>19</sup>

Nitrous oxide emissions are also determined by temperature, ventilation, nitrogen content, the C/N relation, and other factors.<sup>20</sup> Maximum N<sub>2</sub>O formation rates are observed if the supply of oxygen during decomposition is insufficient. This may occur, for example, if the partial pressure of oxygen in the rotting material drops to zero due to very high rates of biological activity.<sup>21</sup>

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.

Hellmann studied the emission of the gases CO<sub>2</sub>, CH<sub>4</sub>, and N<sub>2</sub>O during the composting of domestic waste.<sup>22</sup> The study's results are based on sampling with gas flux chambers and the establishment of concentration using a gas chromatograph. The total CH<sub>4</sub>-C and N<sub>2</sub>O-N emissions were shown in relation to the dry mass of the basic material or the total CO<sub>2</sub>-C emission. Depending on the composting conditions, nitrous oxide emissions range from 12

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<sup>18</sup> Grontmij and IVAM (2004) *A Life Cycle Assessment for Vegetable, Fruit and Garden Waste –Review of the LCA accompanying the 2003 Netherlands National Waste Plan*. De Bilt/Amsterdam, November 2004.

<sup>19</sup> K. Csehi, J. Beck and T. Jungbluth (1996) Emissionen bei der Mietenkompostierung tierischer Exkremente. *Landtechnik* 51, S. 218-219; K. Csehi (1977) *Ammoniakemission bei der Kompostierung tierischer Exkremente in Mieten und Kompostqualität*. Diss. Universität Hohenheim 1977, MEG-Forschungsbericht 311, 156 S; T. Maeda and J. Matsuda (1997) Ammonia Emissions from Composting Livestock Manure (Ammoniakemissionen bei der Mistkompostierung), in A. M. Voermans and G. Montney (eds.) (1997) *Ammonia and Odour Control from Animal Production Facilities* (Ammoniak- und Geruchskontrolle aus Tierproduktionsanlagen), Proceedings I, Vinkeloord, Niederlande, 6.-10. 10. 1997, S. 145-153.

<sup>20</sup> L. Hüther (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.

<sup>21</sup> H. J. Hellebrand (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; S. Zhou, H. Zaeid, and H. Van den Weghe (1999) Kompostierung tierischer Exkremente - Einfluß der Sauerstoffkonzentration auf Reaktionskinetik und Emissionsverhalten, *Agrartechnische Forschung* 5, S. 2-10

<sup>22</sup> B. Hellmann (1995) *Freisetzung Klimarelevanter Spurengase in Bereichen mit hoher Akkumulation von Biomassen*, Abschlußbericht für die Deutsche Bundesstiftung Umwelt, Osnabrück, Zeller Verlag 1995

to 114 g N<sub>2</sub>O-N per tonne of basic dry mass for a composting period of 89 days. In relation to the nitrogen content of the basic dry mass (1% to 1.7%), these emissions account for approximately 0.1 to 0.8% of the initial total nitrogen content. These N<sub>2</sub>O values span a range within which falls the value proposed by Grontmij and IVAM.<sup>23</sup> They propose a value of 101g N<sub>2</sub>O/tonne VFG waste, though this is not related back to N-content of the feedstock.

Ballestero and Douglas also used gas flux chambers and gas chromatography to measure the formation of nitrous oxide during the composting of manure and garden waste.<sup>24</sup> Within 60 days, manure composting caused 2.19% of the initial nitrogen content to be emitted as N<sub>2</sub>O. When garden waste was composted, the percentage of N<sub>2</sub>O emissions was considerably lower (1.18%).

Gronauer et al suggest that around 12% of total nitrogen escapes from the material in the form of ammonia.<sup>25</sup> This gave a figure of 0.53kg/tonne waste in raw gas, but 0.0264kg per tonne waste when the air is passed through a biofilter. Gronauer et al also assumed that 0.15kg N<sub>2</sub>O per tonne waste would be emitted.

One Swedish study<sup>26</sup> assumed the nitrogen leakage to air was 7.5% of the nitrogen content of the feedstock. Of this leakage, it was assumed 89% was emitted as NH<sub>3</sub>, 9% as N<sub>2</sub>O and 2% N<sub>2</sub>. The study for the Danish EPA assumed that of the total amount of nitrogen lost as gaseous emission, 98 % was volatilised as NH<sub>3</sub>, 0.5 % as N<sub>2</sub>O and 1.5 % as N<sub>2</sub>.<sup>27</sup> Clearly, these are figures for raw gas as opposed to gas which has been scrubbed.

Komilis and Ham took the view that depending on the C:N-ratio in the feedstock material, part of the nitrogen present would be volatilised as gas, primarily as NH<sub>3</sub>, but also as N<sub>2</sub>O and N<sub>2</sub>. As with CO<sub>2</sub>, they developed a basis for modelling the emissions of ammonia based upon the dry-matter proportions of food waste, mixed paper and garden waste in the input feedstock.<sup>28</sup> Their equation appears to give values which are much too high, especially in composts with high proportions of food waste (the figures appear too high by a factor of 10 or so).

Schleiss appears to be the only author suggesting emissions of NO<sub>x</sub>. He sets these at 0.24 kg per tonne waste, falling to 0.029 kg per tonne waste when the air is passed through a biofilter. No other study seems to indicate any NO<sub>x</sub> emissions.

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<sup>23</sup> Grontmij and IVAM (2004) *A Life Cycle Assessment for Vegetable, Fruit and Garden Waste – Review of the LCA accompanying the 2003 Netherlands National Waste Plan*. De Bilt/Amsterdam, November 2004.

<sup>24</sup> T. P. Ballestero and E. M. Douglas (1996) Comparison Between the Nitrogen Fluxes from Composting Farm Wastes and Composting Yard Wastes (Vergleich der Lachgasflüsse aus dem Kompostieren von Mist und Grünabfällen), *Trans. ASAE* 39, S. 1709- 1715.

<sup>25</sup> A. Gronauer, M. Helm, H. Schon (1997) *Verhafen und Konzepte der Bioabfallkompostierung – Vergleich – Bewertung – Empfehlungen*, Bayerische Landesanstalt für Landtechnik der TU München-Weihenstephan.

<sup>26</sup> Goran Finnveden, Jessica Johansson, Per Lind and Asa Moberg (2000) Life Cycle Assessments of Energy from Solid Waste, *Forskningsgruppen för Miljöstrategiska Studier, FMS 137*, August 2000

<sup>27</sup> B. Gunnarsdotter Beck-Friis (2001) *Emissions of Ammonia, Nitrous Oxide and Methane during Composting of Organic Household Waste*, *Agraria* 266, Doctoral Thesis, SLU, Sweden, cited in A. Baky and O. Eriksson (2003) *Systems Analysis of Organic Waste Management in Denmark*, Environmental Project No. 822, Copenhagen: Danish EPA.

<sup>28</sup> Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, p.1394.

Eunomia et al used figures for NH<sub>3</sub> and N<sub>2</sub>O of 371g/tonne waste and 11g/tonne waste respectively.<sup>29</sup> They also carried out a ‘back of the envelope’ calculation for an operating plant suggesting:

*A similar attempt at calculations concerning ammonia emissions can be attempted from plant data. One 45,000 tonne plant draws through 14,000 cubic metres of exhaust air per hour with a concentration between 20 and 40 mg per cubic metre. This equates to 545g to 1090g per tonne. This is consistent with other studies but it represents the concentration of exhaust air input to a biofilter. This would be expected to reduce ammonia emissions.*

### Approach Taken in this Study

In this study, we have modelled ammonia emissions assuming 9% of the input nitrogen is converted to ammonia, 1% is converted to N<sub>2</sub>O. These give emissions into raw gas. The effects of biofilters are discussed separately below. A more accurate approach would probably be to vary these proportions in some relation to the C:N ratio of the input material but we could find basis upon which to do this no.

With respect to emissions of nitrogenous compounds, the difference between emissions from windrow composting facilities and enclosed facilities are assumed to be related only to a) N-content and b) the removal efficiencies of the biofilter.

#### A.4.2.4 Emissions of VOCs

Relatively few studies make reference to emissions of VOCs. Komilis and Ham include them in the total inventory, but these appear to be related to energy use on site (which is dealt with separately below). In the UK, the Environment Agency did measure emissions from sites. These are shown in **Error! Reference source not found.**

**Table 8: Emissions of VOCs from Monitoring of Compost Facilities**

Compounds Detected	g/T MSW
m,p Xylene [108-38-3; 106-42-3]	0.81
Nonane [111-84-2]	0.44
o Xylene [95-47-6]	0.54
Beta.-Pinene [127-91-3]	3.7
Ocimene [13877-91-3]	3.0
D-Limonene [5989-27-5]	10.5
Undecane [1120-21-4]	2.4
Dodecane [112-40-3]	1.2
Methyl-(methylethyl)-Cyclohexane [99-82-1]	1.5
Total	24.0

Source: Environment Agency (2000) *Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion, R&D Project Record P1/392/4*

<sup>29</sup> Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC Research & Consulting (2002) *Economic Analysis of Options for Managing Biodegradable Municipal Waste*, Final Report to the European Commission.

### Approach Taken in this Study

We have assumed no emissions of VOCs in the 'low emissions' scenario and used the Environment Agency figure in the 'high emissions' scenario. The use of biofilters is estimated to reduce the emissions by 80% in the case of in-vessel composting facilities.

#### A.4.2.5 Carbon Monoxide

A study by Schleiss, gives a value for emissions of CO as 0.069kg/tonne of waste input. It is possible that this measurement was made close to 'hot spots' and that any CO could be oxidised in passing through the biomass (and biofilter).

### Approach Taken in this Study

We have assumed no emissions of carbon monoxide in the 'low emissions' scenario and used Schleiss's figure in the 'high emissions' scenario.

#### A.4.2.6 Hydrogen Sulphide

Again, only one study, that of Schleiss, gives a figure of 0.29 kg per tonne of waste. Other studies assume no H<sub>2</sub>S.

### Approach Taken in this Study

We have assumed no emissions of hydrogen sulphide.

#### A.4.2.7 Effects of Biofilters

In most life cycle studies, the presence or otherwise of a biofilter is typically seen as important in respect of the impact categories of acidification and eutrophication. For the same reason, to the extent that these impacts are deemed problematic, the assumed efficiency of biofilter removal is of some significance.<sup>30</sup> Indeed, in many processes, the likelihood of significant problems arising in respect of either acidification or eutrophication seems limited – rather, life-cycle studies highlight the *relative potential* to cause a problem, and compost processes therefore tend to fair comparatively poorly.

Typically, enclosed compost plants now operate using biofilters. Some will employ both biofilters and ammonia scrubbers. Biofilters can consist of different materials used to reduce odours and to treat exhaust gases. Typical biofilters may consist of damp wood chippings. An interesting facet of the use of biofilters is that the biofilter itself may give rise to emissions of (different types of) VOCs. This issue has acquired some significance in the regulatory context surrounding mechanical biological treatment plants in Germany. This is because the emissions from biofilters are primarily of relatively benign volatile organic compounds such as terpenes. Despite their relatively benign nature, a focus on VOCs as an undifferentiated group of compounds led to a demand for more effective gas cleaning, typically through using regenerative thermal oxidation (RTO) systems, which eliminate VOCs without generating additional (more benign) VOCs. This discussion might appear to give rise to questions concerning how VOCs should be accounted for in an analysis of the effects

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<sup>30</sup> The key phrase here is 'to the extent that these are deemed problematic'. It is not clear that compost facilities are major contributors to, for example, eutrophication. Where leachate is well managed, and re-used in the process, the potential for problems to arise is likely to be limited. This is why it is important to normalize the impacts associated with life-cycle studies, and to contextualize the nature of the impact category being examined.

given the range in the potential to cause harm that they present.<sup>31</sup> However, few studies assign significant emissions of VOCs to compost (rather, the issue is more important in the context of MBT plants, or plants seeking to ‘compost’ mixed wastes).

Biofilters vary in the efficiency with which they scrub different gases from the exhaust stream. Baky and Eriksson (2003) used the following figures for an enclosed reactor.

**Table 9: Biofilter Removal Efficiencies for Different Air Emissions**

Impact	Removal Efficiency
Reduction of NH <sub>3</sub>	99%
Reduction of N <sub>2</sub> O	90%
Reduction of CH <sub>4</sub>	50%

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

Finnvenden et al (2000)<sup>32</sup> assumed nitrogenous compounds would be removed at 60% efficiency by a bark-based biofilter. This figure is the lowest we could find reported in any literature, and is well below all other values cited (and can be disregarded). Vogt et al assumed a removal efficiency of 96% for NH<sub>3</sub>, 50% for methane and 50% for total organic carbon. Omrani et al site removal efficiencies of 97-99% for a biofilter using peat, soil and sand, whilst one of sawdust, clay and sand achieved 94% abatement.<sup>33</sup>

It should be noted that the potential for further clean up exists where scrubbing equipment is used alongside biofilters. For example, ORA report 100% removal through the combined use of biofilter and scrubbing.<sup>34</sup> Recent developments with MBT plants include the use of thermal filters. These operate so as to effectively crack the organic components of exhaust gases. It is, perhaps, unlikely that thermal filters (regenerative thermal oxidation) would be deployed at compost plants (this is more likely at MBT facilities where VOCs are more problematic). One Austrian study (Lahl et al, 2000) quotes the emissions reductions achieved through thermal processes relative to both standard biofilters, and biofilters alongside ammonia scrubbing. These are given below:<sup>35</sup>

- NMVOCs      90% reduction and 80% reduction;
- CFCs          98% reduction in both cases;
- SO<sub>x</sub>          50% reduction in both cases;
- NH<sub>3</sub>          75% reduction and 0% reduction;
- N<sub>2</sub>O          100% reduction in both cases.

<sup>31</sup> Much of the economic literature addressing the impact of atmospheric emissions of VOCs has concentrated on those VOCs associated with transport emissions, typically organic compounds related to benzene. It seems quite clear that grouping terpenes amongst such compounds would be of questionable value.

<sup>32</sup> Goran Finnvenden, Jessica Johansson, Per Lind and Asa Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, *Forskningsgruppen for Miljostrategiska Studier*, FMS 137, August 2000.

<sup>33</sup> Omrani, G., Safa, M. and Ghaghazy, L. (2004) *Utilization of Biofilter for Ammonia Elimination in Composting Plant*. *Pakistan Journal of Biological Sciences* 7. 2009-2013.

<sup>34</sup> 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.

<sup>35</sup> U. Lahl, B. Zeschmarr-Lahl and T. Angerer (2000) *Entwicklungspotentiale der Mechanisch-biologischen Abfallbehandlung: Eine Okologische Analyse*, Umweltbundesamt: Wien

In most cases, the reductions suggest equivalent performance whether using a biofilter, or a biofilter plus a scrubber. The exceptions are ammonia, and more clearly, NMVOCs. If a biofilter alone achieved a 95% abatement of NH<sub>3</sub>, this would imply approximately 99% abatement of NH<sub>3</sub> where scrubbing equipment was used. This may occur at the expense of a slight increase in emissions associated with energy used in running the plant.

### Approach Taken in this Study

In this study, only basic biofilters are assumed. The removal efficiencies are considered to be as follows:

	Biofilter only	Biofilter and scrubber
➤ NH <sub>3</sub>	95%	99%
➤ N <sub>2</sub> O	0%	0%
➤ VOCs	50%	60%
➤ Methane	50%	50%

The removal efficiency has to be understood in the context of the above discussion. What is being posited is a 50% / 60% removal of more problematic VOCs. The VOCs emitted by the biofilter itself are not attributed any external costs.

### A.4.2.8 Nature of Process, Retention Time and Maturation Period

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 biofiltering / 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 in the round.

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?

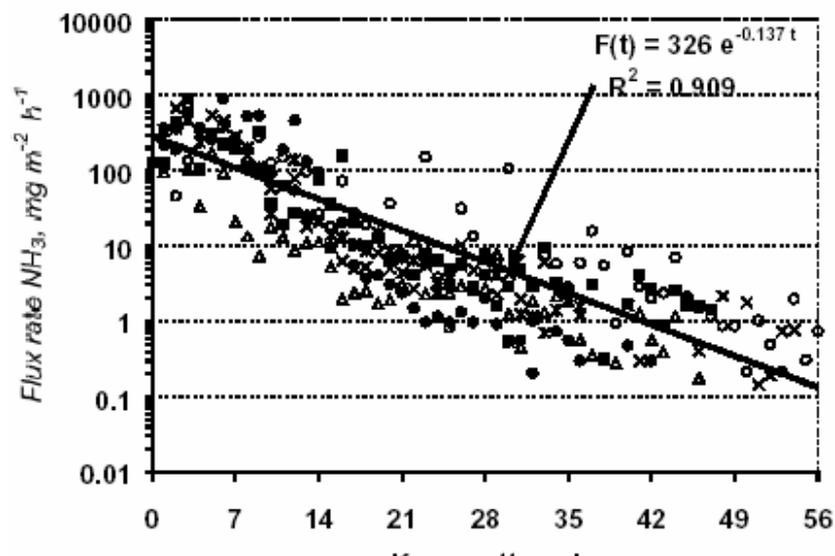
Only Vogt et al appear to have sought to distinguish, for the purpose of emissions, between fresh and mature compost. However, the effects of emissions in the process, and emissions after application to land, appear to largely cancel each other out. More important is the nature of the material which the compost substitutes when it is applied.

As regards laboratory measurements, one study (Hellebrand and Kalk, 2000) did seek to understand the evolution of the key compost-related emissions over time, but the feedstock was manure (as opposed to municipal biowastes).<sup>36</sup> An important observation of the study (to the extent that it may be transferable) was:

*Over the course of the composting period, the emissions of ammonia and methane are reduced almost completely during the first three weeks. Nitrous oxide emissions exhibit great variation over the entire composting period. As the quantity of NH<sub>3</sub> and CH<sub>4</sub> diminishes, N<sub>2</sub>O emissions show an increasing tendency. Maximum values are measured after two to six weeks. Afterwards, N<sub>2</sub>O formation slowly decreases.*

Highest ammonia emissions were shortly after the manure was added to the process, and detection limits were reached after eight weeks (see **Error! Reference source not found.**). Discussions with process managers appear to confirm that with municipal feedstocks as well, ammonia is predominantly released early in the process.

Figure 7: NH<sub>3</sub> Emissions With (Exponential Trend Function) for a 56 Day Composting Period

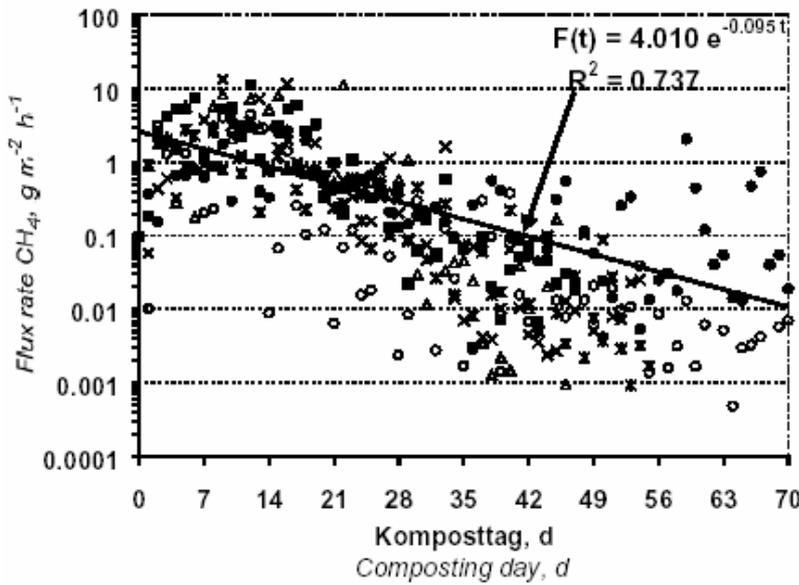


Source: Hellebrand, H. J. and W. D. Kalk (2000) Emissions Caused by Manure Composting, *Agrartechnische Forschung* 6 (2000) Heft 2, S. E 26-E 31

A similar function was used to describe methane emissions (see **Error! Reference source not found.**). However, nitrous oxide emissions were more variable, exhibiting a less stable pattern of decay over time (see **Error! Reference source not found.**).

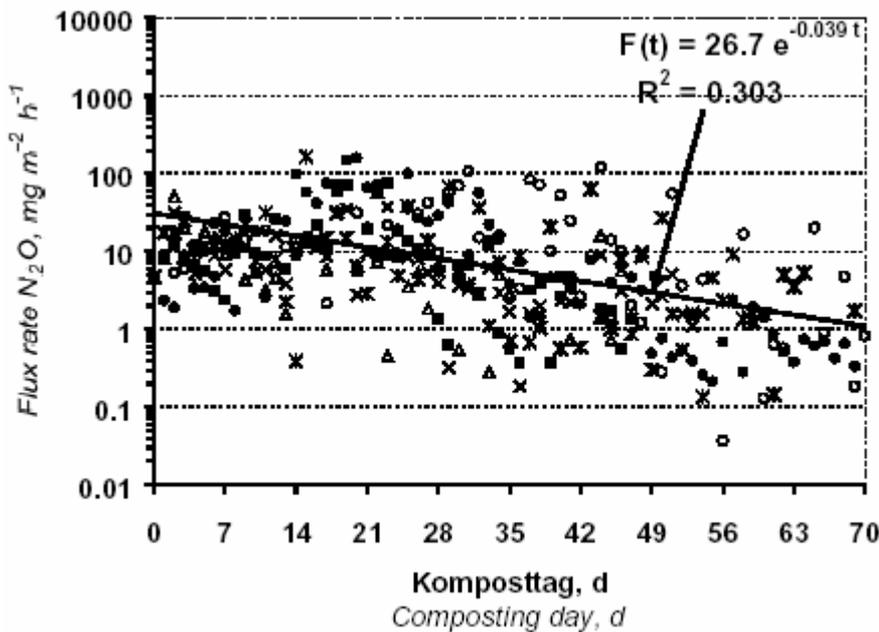
<sup>36</sup> H. J. Hellebrand and W. D. Kalk (2000) Emissions Caused by Manure Composting, *Agrartechnische Forschung* 6, Heft 2, S. E 26-E 31

Figure 8: CH<sub>4</sub> Emissions With (Exponential Trend Function) for a 70 Day Composting Period



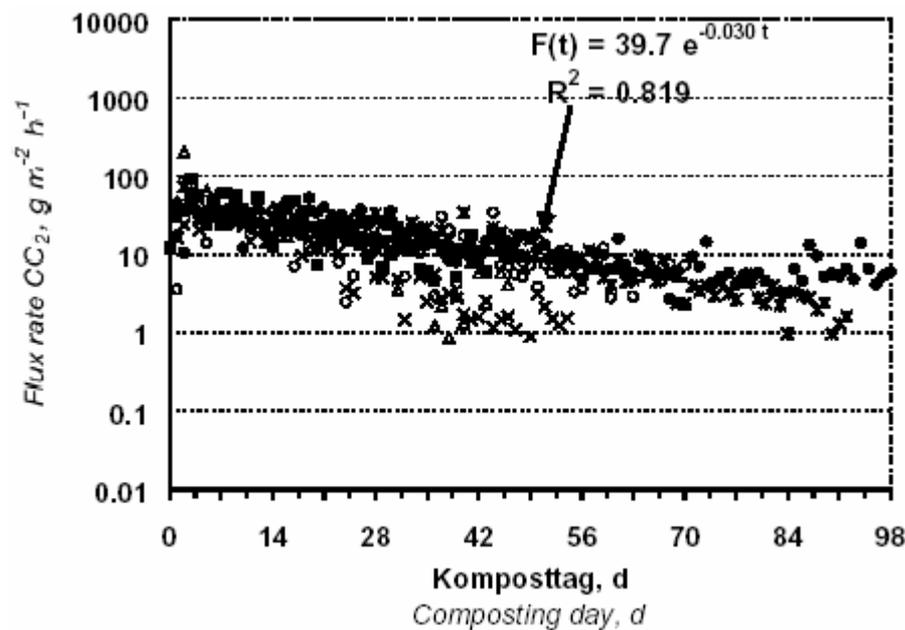
Source: Hellebrand, H. J. and W. D. Kalk (2000) Emissions Caused by Manure Composting, Agrartechnische Forschung 6 (2000) Heft 2, S. E 26-E 31

Figure 9: N<sub>2</sub>O Emissions With (Exponential Trend Function) for a 70 Day Composting Period



Source: Hellebrand, H. J. and W. D. Kalk (2000) Emissions Caused by Manure Composting, Agrartechnische Forschung 6 (2000) Heft 2, S. E 26-E 31

Figure 10: CO<sub>2</sub> Emissions With (Exponential Trend Function) for a 70 Day Composting Period



Source: Hellebrand, H. J. and W. D. Kalk (2000) *Emissions Caused by Manure Composting*, *Agrartechnische Forschung* 6 (2000) Heft 2, S. E 26-E 31

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 systems using biofiltration, the less will be the difference in emissions from the 'short duration' and 'long duration' processes.

However, this comment should be considered in the context of the discussion above which suggests that the bulk of emissions of ammonia (one of the gases likely to be dealt with effectively by biofilters) may be emitted in the first few weeks of the composting process.

#### A.4.2.9 Bioaerosols

Significant recent interest, at least within the UK, has centred on the potential consequences of emissions of bioaerosols from composting processes. The Environment Agency sponsored two studies in the early part of the decade. These studies were reviewed for the Cabinet Office in a paper by Hogg.<sup>37</sup> That review cast some doubt upon the conclusions drawn from the research undertaken, notably, the way in which it was used to influence regulatory guidance, for example, in respect of distancing from compost facilities.<sup>38</sup>

Bioaerosols are micro-organisms and other tiny biological particles that are suspended in air. They are respirable and generally invisible. Dusts are small particles that are larger

<sup>37</sup> D. Hogg (2002) *Waste Treatments Mk II: Health Effects*, Report for the Strategy Unit.

<sup>38</sup> See Environment Agency (2001) *Agency Position on Composting and Health Effects*, 2001.

than bioaerosols. They are inhalable but not respirable and are visible. It should be noted that bioaerosols from the composting process contain the same micro-organisms as ones to which citizens are routinely exposed. They are present naturally and are essential to the recycling of nutrients in our gardens, parks and countryside.

Bioaerosols and dusts can both be produced by the composting process. Surveys have drawn particular attention to a fungus called *Aspergillus fumigatus*. It is found all over the world, especially in soils and in forest litter. It is particularly associated with the composting process as it is capable of degrading cellulose (a carbohydrate found in plant material) and it is capable of surviving at temperatures of up to 65 °C. As part of its lifecycle, *Aspergillus fumigatus* produces tiny spores. If inhaled as a bioaerosol these spores may cause allergies and inflammation, which in certain individuals can cause serious health disorders such as Asthma, Alveolitis, Mucus membrane irritation, Chronic Bronchitis and coughs, Gastro-intestinal disorders and Skin disorders.

Again, effective operational management can help to control the formation of bioaerosols and dusts these include:

- Ensuring that the optimum moisture content is maintained during the composting process;
- Ensuring that the compost is turned regularly;
- Maintaining good housekeeping; and
- Erecting bunding/planting trees around the perimeter of the site.

Most surveys have led to the following general assessment:

- Activities run at a composting facility expose workers to a certain load of dust particles and aerosols, above all while turning or moving dry, dust-like materials. Fresh food waste, for instance, is too wet to release dust;
- The risk is similar to that experienced by workers at earth-moving companies;
- Health risk management should include a prevention programme for workers (as they currently do in many Member States), including:
  - ✓ Individual protection devices (dust masks should be worn during most dusty activities); and
  - ✓ Periodic health assessment;
- Nearby dwellers are not so exposed in most situations. Distances in the order of 200-300 metres are frequently enough for bioaerosols to reduce to background concentrations of airborne microorganisms. In many Member States, such distancing is frequently implemented as a means to reduce odour;
- Running operations in enclosed buildings sharply reduces the occurrence of risks in external spaces

Some plants now pass exhaust air through compounds which effectively cleanse the exhaust air of bioaerosols. A conscious attempt has been made to find appropriate cleaning agents that do not render organisms in a biofilter 'useless'.

Generally, the problem appears to be greater at open air windrow facilities. This perspective has, for the time being, been reinforced by the Environment Agency rules concerning distances from compost facilities to dwellings. If this approach achieves its goal – that effectively, no one is exposed to levels of bioaerosols above a specific level, then it seems reasonable to suggest that the external costs associated with these emissions are unlikely to be significant other than for operatives. The review by Hogg also noted that the standard set by the Environment Agency would have implied – even in cases which were examined in the case studies – that the compost plants had to act as sinks for bioaerosols.<sup>39</sup>

Measurements concerning bioaerosols were made at a silo-style in-vessel plant. Because the plant operates on natural convection processes (rather than turning, or forced aeration) there is no forcing of air through the compost mass. The upper layers of the column effectively act as a biofilter. Because there is so little turning and forcing of air, this would be expected to reduce bioaerosols, although shredding and bagging processes might still lead to elevated measurements. This plant, it should be noted, is close to a chicken farm.

The results (see Table 10 11) suggest low levels of emission. Although far more critical appraisal of these results is required, there is at least a suggestion that facilities designed to reduce the degree of turning / forced aeration can actually lead to reduced bioaerosol emissions.

The message seems to be that more work is required on the measurement across different types of facilities using standardised approaches (to ensure comparability). This should be supported by measurements at other waste treatment facilities to understand the degree to which the bioaerosols are a problem only at compost plants, or whether the issue arises at other facilities as well (though this is only of concern if the issue is regarded as being of significance).

**Table 10: Concentrations of Micro-organisms**

		Moulds count per m3	TVC count per m3
<b>Process Not in Operation</b>			
Inside Building	by mixer	367	33
Inside Building	on high gantry	33	17
Inside Building	by bagging unit	117	200
Inside Building	by auger extractor	83	267
Outside Building	10m downwind	17	350
Outside Building	10m downwind	133	67
Outside Building	10m downwind	117	633
Outside Building	10m upwind	33	33
Outside Building	10m upwind	50	33
Outside Building	100m downwind	33	50
Outside Building	100m downwind	0	150
Outside Building	100m downwind	17	267
<b>Process in Operation</b>			
Inside Building	adjacent to bagging	133	233

<sup>39</sup> D. Hogg (2002) *Waste Treatments Mk II: Health Effects*, Report for the Strategy Unit.

		Moulds count per m3	TVC count per m3
Outside Building	10m downwind during bagging	150	2200
Inside Building	adjacent to mixer	1267	2133
Inside Building	on gantry during top-up	467	600
Inside Building	during compost extraction	100	467
Inside Building	during compost extraction	500	3067
Inside Building	by Auger top end (RHS)	533	2533
Inside Building	by Auger bottom end (RHS)	867	1233
Inside Building	by Auger top end (IHS)	367	2750
Inside Building	by Auger bottom end (IHS)	133	833
Inside Building	by site office	333	1350
Inside Building	by bagging line (not operating)	267	300
Outside Building	in delivery bay during drop off of material	400	967
Inside Building	moving raw material to storage area	2200	2000
Outside Building	10m downwind	67	167
Outside Building	10m downwind	233	400
Outside Building	10m downwind	283	433
Outside Building	10m downwind	367	367
Outside Building	10m downwind	367	300
Outside Building	10m downwind	50	400
Outside Building	10m upwind	150	133
Outside Building	10m upwind	133	600
Outside Building	10m upwind	100	267

### Approach Taken in this Study

A considerable amount of interest in bioaerosols has arisen on the back of the Environment Agency studies, and in the wake of concerns regarding the potential for the emissions to give rise to health problems. Even data already existing, however, suggests that the fall off in concentration of most organisms of concern is fairly swift. Hence, where the Environment Agency enforces distancing requirements to reduce exposure to any elevated concentrations of bioaerosols, it seems reasonable to argue that the associated external costs will be negligible. The same could not necessarily be said for operators. Here, however, we do not have the necessary dose response functions which would allow for some estimate of impact related to specific health end-points. Consequently, this is an omission from the study, but one which is unlikely to affect results significantly. Clearly, the inference for operators is that they should seek to ensure minimal exposure of staff to the organisms concerned, both in routine work, but especially where events likely to lead to release of bioaerosols are being undertaken. Good practice should be able to significantly reduce, even if it might not eliminate completely, the effects of bioaerosols.

### A.4.3 Leachate

The potential for leachate to be produced in significant quantities is dependent upon the nature of the process, most notably, whether it is enclosed or in-vessel. In enclosed facilities, it is perhaps more common for problems to arise in keeping the material moist rather than it producing excessive, and potentially problematic, leachate. Hence, leachate is commonly recirculated.

Komilis and Ham note, in their review, that varying amounts of leachate have reportedly been produced in MSW and garden waste composting facilities starting from 0 to approximately 490 litres / tonne.<sup>40</sup> They took the view that:

*Given the limitations in the available data, this aspect of leaching emissions from MSW and yard wastes during and after composting was not included in this LC (Life Cycle Inventory). The LCI includes, however, all waterborne emissions associated with diesel and electricity precombustion and combustion processes.<sup>41</sup>*

Where biofilters are operated, some processes generate a liquid associated with the deployment of a heat exchanger on the exhaust air prior to its entering the biofilter. The rationale for this is to condense some of the compounds responsible for the generation of odours, so that the liquid contains some of the odorous emissions from the process.

### Approach Taken in this Study

Assigning an external cost – an impact given a monetary value – is extremely difficult where leachate is concerned. Many studies, even those covering landfills, where controls on leachate are, potentially, more problematic, have assigned no external cost to leachate.<sup>42</sup> In the case of compost facilities, given the potential problems associated with maintaining the biomass in a moist condition, the recirculation of leachate may be beneficial. In addition, the recirculation can reduce any issues associated with nitrogen in the leachate (though it might increase the rate of volatilisation of ammonia), and may make this available to the microbes giving rise to the degradation of material.

For these reasons, and because also assigning a value to leachate externalities is problematic, we have assigned a zero value to leachate generation. Furthermore, the process modelling, especially of nitrogen, effectively assumes that recirculation occurs.

## A.4.4 Input Energy

The energy use at compost plants obviously depends upon the nature of the plant in question. It is difficult to be precise about energy use at plants because there is considerable variation in the type of equipment used.

In the study of Finnvenden et al, it was assumed that 54.4 MJ electricity per tonne of food waste would be consumed. It was also assumed that diesel would be consumed in the wheel loader, different types of mills and a screen, this amounting to 555.5 MJ/tonne food

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<sup>40</sup> J. P. Metcalfe, D. Wood, and S. Harrad (1997). *An assessment of the behaviour of organic micropollutants in waste composting processes*. Draft Technical Rep. Prepared for United Kingdom's Environment Agency; U. K. Woodthorne, U. Krogmann and H. Woyczehowski (2000). Selected characteristics of leachate, condensate and runoff released during composting of biogenic waste, *Waste Manage. Res.*, 18, 235–248.

<sup>41</sup> Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, p.1394.

<sup>42</sup> Typically, the argument is made that the leachate is collected and tankered off for treatment, so that the treatment cost is already internalised in the operation of the landfill. This is not strictly true since a) the treatment process itself has some environmental impacts; b) the capture of leachate is not necessarily 100%, as measured instantaneously in the period of operation of the landfill; and c) where landfills are concerned, the instantaneous rate of capture of leachate in the short-term may mask much lower rates later in the life of the landfill.

waste.<sup>43</sup> This latter figure – equivalent to more than 10 litres of diesel per tonne of waste - is enormous and cannot be considered seriously.

Baky and Eriksson used different quantities of energy depending upon whether the composting process was enclosed or in-vessel. The figures used are shown in **Error! Reference source not found.** It can be seen that the figures vary significantly across the processes, and that they differ considerably from those of Finnvenden et al. The diesel use for the open air facility is extremely low.

Eunomia et al assumed that both diesel and electricity would be used. It was assumed that 50kWh (180 MJ) per tonne of waste would be used and 1l (around 42 MJ) of diesel.<sup>44</sup>

**Table 11: Energy Use at Open Air Windrows and Reactor Composts**

Type of compost	Value
Open windrow	
<i>Electricity consumption</i>	0 MJ/tonne
<i>Diesel consumption</i>	1.5 MJ/tonne
Reactor compost	
<i>Electricity consumption</i>	180.1 MJ/tonne
<i>Diesel consumption</i>	75.5 MJ/tonne

A US study by Diaz et al. suggests average energy requirements in a MSW composting facility are 34.4 kW h/ t of MSW. <sup>45</sup> The measured values include energy consumed directly within the facility, including extensive preprocessing prior to composting (e.g., size reduction, screening). However, no odour control system was included. These figures are close to those given by Grontmij and IVAM in recent work for the Netherlands Association of Waste Management Companies. An inventory of VFG- composting facilities revealed an average value of 29kWh per tonne of waste.<sup>46</sup>

A more recent US study suggested values of 97kWh/t and 167 kWh/t for high quality (HQCF) and low-quality facilities (LQCF), respectively, with only 29kWh/t for open-air windrow facilities.<sup>47</sup> These higher values are partly a reflection of the fact that precombustion and combustion energy requirements were included in the more recent study. It is worth reporting some of the findings in more detail:

*Based on the HQCF, 29, 56, 3, 7, and 1% of the total energy requirements are due to hammermill, odor control, front end loaders, windrow turner, and trommel screen*

<sup>43</sup> Goran Finnvenden, Jessica Johansson, Per Lind and Asa Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, Forskningsgruppen for Miljostrategiska Studier, FMS 137, August 2000.

<sup>44</sup> Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC Research & Consulting (2002) *Economic Analysis of Options for Managing Biodegradable Municipal Waste*, Final Report to the European Commission.

<sup>45</sup> L. F. Diaz, C. G. Golueke and G. M. Savage, G. M. (1986). Energetics of compost production and utilization *BioCycle*, 27(8), 49–54.

<sup>46</sup> Grontmij and IVAM (2004) *A Life Cycle Assessment for Vegetable, Fruit and Garden Waste –Review of the LCA accompanying the 2003 Netherlands National Waste Plan*. De Bilt/Amsterdam, November 2004.

<sup>47</sup> Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, p.1394.

operation, respectively, while 4% is due to building operation. Relatively similar values apply to the LQCF. In the case of the YWCF [garden waste composting facility], 55, 16, 8, and 21% of total energy are due to the tub grinder, the front-end loader, screens and building operation, respectively.

In the LQCF and HQCF, electricity related energy accounts for more than 90% of the total energy with the rest being diesel derived energy. In the YWCF, 60% of the total energy is diesel combustion energy with 29% being electricity and the rest being diesel precombustion energy. This is because only diesel-powered equipment is used in the YWCF and electricity is limited to the building operation.

[...] Due to the extensive usage of electricity in the LQCF and HQCF diesel combustion is responsible for less than 50% of [the energy-related emissions within the boundary of the facility] with the exception of CO. [...] Diesel combustion is generally responsible for production of a relatively large percentage of the NO<sub>x</sub> and CO emissions from both the MSW composting facilities. SO<sub>x</sub> emissions are primarily produced due to electricity consumption and are therefore produced outside the boundaries of the facility itself. Because of the limited use of electricity in the YWCF, the majority of the atmospheric emissions are due to diesel combustion.

The study also notes that for the HQCF:

Approximately 90% of the overall fossil CO<sub>2</sub> emitted from the aforementioned facility—i.e., approximately 33 kg CO<sub>2</sub>/ t MSW—is a result of electrical usage. However, 92, 91, and 98% of the overall CO<sub>2</sub> emissions from the LQCF, HQCF, and YWCF, respectively, are due to the decomposition of the organic substrate. Essentially all of the ammonia emitted from all facilities is due to decomposition.

The sensitivity of energy usage to specific variables was also tested in the study (see Table 12). The significant variation with some of the parameters makes clear why generalization is so difficult in this area. Importantly, since the HQCF was a plant treating mixed waste, the hammermill design was found to have a marked effect on total energy and emissions, since it accounts for a large usage of electrical energy. The study found that presorting would affect the shredding coefficient by changing it from 1 to 0.67, resulting in reductions in total energy usage and associated gaseous environmental pollutants.

**Table 12: Sensitivity of Energy Use to Variation in Parameters**

Parameter	Base case	Adjusted value	Change in total energy (%)
Turning frequency (No./week)	1	1.5	0.7
Retention time (days)	30	45	16.1
Odour control air change rate (min)	120	80	18.8
No odour control system	Present	Absent	-37.7
Compost windrow height (m)	1.8	2.7	-10.6
Change of hammermill shredding coefficient (raw and presorted waste)	1.0 (Raw)	0.67 (Presorted)	-17.4

Source: Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, p.1394.

## Approach Taken in this Study

In this study, we have assumed that:

- For open-air windrow facilities:
  - Diesel use is 1l per tonne input
  - Electricity use is zero.
- For enclosed facilities with a biofilter:
  - Diesel use is 0.8l per tonne input
  - Electricity use is 75kWh per tonne

These are believed to be quite generous in terms of energy consumption.

Clearly, it would be desirable to link the market for the output, and hence the retention time, to energy use. However, we feel these values are a fair reflection of what might be expected at facilities today.

## A.4.5 Quantity and Quality of Compost Output

The quantity of compost resulting from any process is likely to relate to the nature of the process, the retention time in the active phase, and the time for curing (or maturation) of the compost. The modelling assumes a relatively mature compost. Such a mature compost would not be necessary for most agricultural applications, though it would be essential in higher quality and value uses such as in potting mixes (see Section 7 of the Main Report).

The quantity of material is derived through use of an equation for dry matter loss (from Komilis and Ham).<sup>48</sup> The remaining mass of material has not been calculated but is taken from empirical data concerning the quantity of compost produced from different feedstocks.

### A.4.5.1 Heavy Metals and Persistent Organic Pollutants

In the life-cycle context, the quality of compost is typically used to estimate emissions of heavy metals and persistent organic pollutants to water and land from compost.

Where composts are derived from source separated feedstocks, the level of contamination with potentially toxic elements is likely to be low.<sup>49</sup> Recent investigations have, indeed, sought to ensure that standards for quality composts are set to ensure no net accumulation of these contaminants in soil. Evidently, the issue is of greater significance – though it is no easier to analyse – where composts derived from mixed wastes are concerned.<sup>50</sup>

In most studies, heavy metals are considered such that the total amount fed into the compost equals the amount in matured compost. Because of the degradation of organic

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<sup>48</sup> Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, Vol. 130, No. 11, November 1, 2004, p.1394.

<sup>49</sup> See D. Hogg, J. Barth, E. Favoino, M. Centemero, V. Caimi, F. Amlinger, W. Devliegher, W. Brinton, and S. Antler (2002). *Comparison of Compost Standards Within the EU, North America and Australia*. Main Report. The Waste and Resources Action Programme (WRAP), Banbury, UK.

<sup>50</sup> See F. Amlinger, M. Pollak and E. Favoino (2004) *Heavy Metals and Organic Compounds from Wastes Used as Organic Fertilisers*, Final Report to the European Commission, ENV.A.2/ETU/2001/2004, July 2004.

matter, the concentration of metals on a dry matter basis increases because of the mass loss in the process. The more mature composts will, other things being equal, show higher contaminant levels by this measure.

Where persistent organic pollutants (POPs) are concerned, different processes occur. Persistent organic pollutants appear to be affected by the compost process in different ways. Furthermore, different feedstocks appear more or less likely to carry high concentrations of POPs.<sup>51</sup> It is difficult to make generalisations concerning the concentrations of POPs in compost, still less to understand what their effects might be (with a view to subsequent monetisation of those impacts). Clearly, rates of application are also important.

### Approach Taken in this Study

No positive or negative impacts associated with heavy metals or POPs are attributed in this study. It would appear that any negative effects associated with compost applications would be associated with heavier PAHs and PCBs, and possibly, with PCDD/Fs. Positive effects may also arise in location-specific contexts due to the application of what are pejoratively termed 'heavy metals', some of which (e.g. zinc and copper) are essential for plant nutrition, and depleted in some soils.

### A.4.5.2 Nutrients

Of more significance for the analysis is the nutrient content of the output compost, and the quantity of material derived per tonne of input waste. In principle, it would be possible to calculate this from a substance flow analysis. However, this was felt to be unnecessary given that more accurate empirical data exists from actual measurements of nutrient content. Readily available, real data was used in this context rather than adopting the substance flow route, with potential to give rise to errors.

The nutrient values we have used for composts are shown in **Error! Reference source not found.**

Table 13: Nutrient Content of Composts with Different Biowaste Components

Mix	N	P	K
Garden Only	1.07%	0.47%	0.42%
Mainly Garden	1.31%	0.77%	0.70%
Kitchen and Garden	1.79%	1.38%	1.26%

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<sup>51</sup> See Rahel C. Brandli, Thomas D. Bucheli, Thomas Kupper, Reinhard Furrer, Franz X. Stadelmann, and Joseph Tarradellas (2005) Persistent Organic Pollutants in Source-Separated Compost and Its Feedstock Materials—A Review of Field Studies, *Journal of Environmental Quality*, Vol.34, No.3, May-June 2005, pp.735-60.

## A.5.0 Anaerobic Digestion Module

### A.5.1 Defra Health Effects Study

As with composting, we start with the review of emissions provided in the Defra health effects study. This study provided emissions from the plant in Munster, Ireland. The study describes the system as a wet one-stage process. It is a process treating an organic fraction separated from mixed wastes. Consequently, though this is not clearly acknowledged by the authors, certain gases are likely to be present which might not be present in emissions from plants treating source separated biowastes.

The emissions reported are shown in **Error! Reference source not found.** Although the data pedigree is given as moderate, as the study itself says the data is from one study, and the data is for digestion of *mixed* municipal waste. It goes on to note that additional emissions may be related to aerobic maturation:

*There is also commonly an aerobic maturation stage of the digestate which may release minor quantities of carbon dioxide and oxides of nitrogen as well as water vapour.*

Table 14: Air Emissions from Defra-funded Health Effects Study

Substance	Measured concentrations in Biogas (mg/Nm <sup>3</sup> )	Estimated concentrations from energy utilisation plant (mg/Nm <sup>3</sup> )	Mass released per tonne of waste processed (g/T)
Nitrogen oxides	29	432 †	188
Particulates	No data	No data	No data
Sulphur dioxide	44	6.9 †*	3.0
Hydrogen chloride	< 0.3	< 0.047 †*	< 0.02
Hydrogen fluoride	< 0.1	< 0.017 *	< 0.007
Total VOCs	No data	No data	No data
1,1-Dichloroethane	No data	No data	No data
Chloroethane	No data	No data	No data
Chloroethene	< 2	0	0
Chlorobenzene	No data	No data	No data
Tetrachlorethene	0.081	0.00081#	0.0004
Methane	No data	No data	No data
Cadmium	< 0.0018	< 2.81 x 10 <sup>-4</sup> §*	< 0.0001
Nickel	< 0.0039	< 6.09 x 10 <sup>-4</sup> §*	< 0.0003
Arsenic	< 0.0079	< 1.23 x 10 <sup>-3</sup> §*	< 0.0005
Mercury	< 0.0083	< 1.30 x 10 <sup>-3</sup> §*	< 0.0006
Dioxins and furans ngTEQ/Nm <sup>3</sup>	< 0.0001	No data	No data
Dioxin-like PCBs	< 0.000042	No data	No data
Carbon dioxide	No data	No data	No data

Source: *Enviros, University of Birmingham, RPA Ltd., Open University and Maggie Thurgood (2004) Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes, Final Report to Defra, March 2004.*

It is somewhat strange that reference is made to nitrogen oxides (given that the study itself expresses some surprise that these are reported in the raw gas data), and that no mention is made of, for example, the potential for methane emissions in the transition from an anaerobic process to an aerobic one. This *may* be an important aspect of management of the digestion process to minimize air emissions overall (see below).

For the purposes of this study, it is reasonable to state that there is little meaningful data regarding emissions from plants digesting source separated biowaste, based upon the Defra study. It should be noted that the Defra study did not report any emissions for CO<sub>2</sub> even though this is a key component of the raw biogas (and is widely reported in a number of different studies). Indeed, CO<sub>2</sub> is *the* major component of emissions once the biogas is combusted, and the magnitude of emissions is readily estimable from known parameters.

We have therefore looked at a number of other sources to develop an understanding of the likely emissions from anaerobic digestion processes.

## A.5.2 Air Emissions

As with composting, 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). The following provides a review of some of the data from different studies.

### A.5.2.1 Biogas Generation

Frequently, biogas generation is quoted in terms of production per unit of volatile solids input to the digester. Volatile Solids (VS) are the components (largely carbon, oxygen, and nitrogen) which burn off of a dry sample in a laboratory furnace at 500-600 °C, leaving only the ash (largely calcium, magnesium, phosphorus, potassium, and other mineral elements that do not oxidize). Volatile solids can further be categorised according to whether they are biological volatile solids or refractory volatile solids.

For biological materials, the carbon content and the volatile solids content of the biowaste are related by the following equation:<sup>52</sup>

$$\% \text{ Carbon} = (\% \text{ VS}) / 1.754$$

Reactor performance is often related to the rate of volatile solids destruction. For digesters, the figure usually lies between 50-70%. This figure can vary in accordance with the type of reactor, the retention time and the nature of the biodegradable wastes entering the reactor. Performance is often measured in terms of methane production per unit of volatile solids input to the digester. For example, Vandevieuvre et al report variation from 90 Nm<sup>3</sup> CH<sub>4</sub> per tonne fresh garden waste to 150 Nm<sup>3</sup> CH<sub>4</sub> per tonne fresh food waste (210-300 Nm<sup>3</sup> CH<sub>4</sub>/tonne VS, equivalent to a 50%-70% reduction in VS). Knowledge of one or other of the carbon content or the volatile solids content allows an estimation of the methane generated. Since the methane content of biogas is usually 55%-65% of the total biogas, and since the remainder is principally carbon dioxide, basic estimates of CO<sub>2</sub> and methane generation in raw biogas can be made.

Biogas generation varies considerably across different facilities. One study from Canada (Chavez-Vasquez and Bagley, 2002) sought to relate the production of biogas from different processes to the different fractions of the waste stream. The results are shown in **Error! Reference source not found..** From this, it seems quite clear that performance varies

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<sup>52</sup> R. C. Adams, F. S. MacLean, J. K. Dixon, F. M. Bennett, G. I. Martin, and R. C. Lough (1951) The Utilization of Organic Wastes in N.Z: Second Interim Report of the Inter-departmental Committee, *New Zealand Engineering* November 15, 1951, pp.396-424.

significantly across digestion facilities depending upon their design and operational characteristics.

**Table 15: Digester Performance from Different Reactor Types (m<sup>3</sup> biogas per tonne volatile solids)**

	One-stage High solids			One-stage Low Solids	One-stage High Solids (Batch)	Two-stage High Solids (with Steam Disruption)
	Valorga	Kompogas	Dranco	BTA	Biocel	SUBBOR
Kitchen waste	634	460	671	695	498	880
Garden waste	211	152	426	230	166	293
Paper and Cardboard	353	256	485	385	277	490

Source: Mariana Chavez-Vasquez and David M Bagley (2002) *Evaluation of the Performance of Different Anaerobic Digestion Technologies for Solid Waste Treatment*, Paper Presented to CSCE / EWRI of ASCE Environmental Engineering Conference, Niagara (Canada) 2002

The ORWARE model also tries to predict the biogas generation from the composition of input wastes. It is based upon a plant in Uppsala described by Dalemo.<sup>53</sup> This was a one-stage wet CSTR plant. In ORWARE, the amount of biogas from anaerobic digestion depends of the incoming materials composition of organic carbon. The model assumes degradation of organic substances according to first order kinetics:

$$D = D_0 / (1 + (1/k * HRT))$$

where

D = Degradation ratio

D<sub>0</sub> = Maximum degradation ratio

k = first order rate constant

HRT = Hydraulic retention time

Different organic substances are decomposed at different rates depending upon the constituent organic compounds. The parameters used are shown in **Error! Reference source not found.**

**Table 16: Performance Parameters for the Anaerobic Digestion Process.**

Organic substances	D <sub>0</sub>	K (days <sup>-1</sup> )	% CH <sub>4</sub>
C <sub>-chsd</sub> (lignin)	0	0.001	50
C <sub>-chmd</sub> (cellulose)	1-1.77* C <sub>-chsd</sub>	0.18	50
C <sub>-chfd</sub> (starch. sugars)	1.0	0.23	50
C <sub>-fat</sub>	0.95	0.13	69
C <sub>-protein</sub>	0.8	0.13	78

<sup>53</sup> M. Dalemo (1997). *The ORWARE Simulation Model - Anaerobic Digestion and Sewage Plant Sub-models*. Licentiate thesis. Swedish University of Agricultural Sciences, SLU, Uppsala. M. Dalemo. (1999). *Environmental Systems Analysis of Organic Waste Management. The ORWARE Model and the Sewage Plant and Anaerobic Digestion Submodels*. Ph D Thesis. Swedish University of Agricultural Sciences, Uppsala.

It should be noted that the work upon which the study was based – outlined in more detail by Dalemo – made reference to the validity of the data. It noted that the data was applicable to CSTR systems and that the rates of degradation were applicable to the mesophilic range. It is not clear how these data were adapted (if at all) by Baky and Eriksson. Dalemo also includes some modelled figures for digestion of source separated household waste and other materials using data on organic composition from Sonesson and Jonsson.<sup>54</sup> These estimates of biogas production are shown in **Error! Reference source not found.**

Even though the ORWARE model is capable of relating biogas generation to input materials, Baky and Eriksson essentially restricted the generation of biogas to 125 Nm<sup>3</sup> per tonne of waste input.<sup>55</sup> They assumed 50% thermophilic and 50% mesophilic operation. This is a low rate of biogas generation.

Finnvenden et al<sup>56</sup> based their inventory data on a study by Nilsson.<sup>57</sup> The inventory data was collected at the digestion plant in Kristianstad, Sweden. The plant was a wet one-stage mesophilic anaerobic digestion process, and was receiving organic waste mainly from households, food industry, restaurants and agriculture. In their study, the original inventory data for anaerobic digestion was supplemented with data on metals according to the food waste composition. The metals were assumed to be retained within the digestion residue.

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<sup>54</sup> U. Sonesson and H. Jonsson (1996) Urban Biodegradable Waste Amount and Composition – Case Study Uppsala. Dept. of Agricultural Engineering, Swedish University of Agricultural Sciences, Uppsala.

<sup>55</sup> A. Baky and O. Eriksson (2003) Systems Analysis of Organic Waste Management in Denmark, *Environmental Project No. 822*, Copenhagen: Danish EPA.

<sup>56</sup> Goran Finnvenden, Jessica Johansson, Per Lind and Asa Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, Forskningsgruppen for Miljostrategiska Studier, FMS 137, August 2000.

<sup>57</sup> B. Nilsson (1997). *Kompostering eller rötning? En jämförande studie med LCA-metodik*. Master Degree Thesis. Chalmers Tekniska Högskola, Göteborg.

Table 18: Modelled Performance of Reactors

		Source separated household waste	Source separated waste from restaurants	Manure <sup>1</sup>	Slaughterhouse Waste
Content					
Protein	% DM	13	13	8.8	50
Fat	% DM	18	24	0.7	37
Fast degrading carbohydrates	% DM	22	19	0	2.7
Medium degrading carbohydrates	% DM	23	20	57	0
Slow degrading carbohydrates	% DM	4.4	4	8	0
TS	%	35	30	1.6	30
Gas Production					
HRT=10	l CH <sub>4</sub> /kg VS	210	190	155	160
HRT=15	l CH <sub>4</sub> /kg VS	330	360	210	480
HRT=20	l CH <sub>4</sub> /kg VS	360	395	230	520
HRT=30	l CH <sub>4</sub> /kg VS	390	430	250	570
Gas Composition	% CH <sub>4</sub>	61	63	55	72

<sup>1</sup> circa 50% manure from pigs and 50% from cattle

Source: M. Dalemo (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.*

A weight loss of 40% of the total solids (TS) in food waste (assumed 30% dry matter) was assumed due to degradation through release of CO<sub>2</sub> and CH<sub>4</sub> during the digestion process. Some of this data is discussed further below.

### Approach Taken in This Study

In this study, we have examined the biogas generated using three approaches:

- The ORWARE approach, assuming a hydraulic retention time of 18 days;
- An approach based on the work of Chavez-Vasquez and Bagley discussed above; and
- An approach based on VS content and using assumptions concerning the rate of VS destruction and the rate of conversion of VS into methane from Cecchi et al.<sup>58</sup>

We then compared these results with those from suppliers which we gathered in the context of recent work in Northern Ireland (see **Error! Reference source not found.**).

<sup>58</sup> F. Cecchi, P. Traverso, P. Pavan, D. Bolzonella and L. Innocenti (2003) Characteristics of the OFMSW and Behaviour of the Anaerobic Digestion Process, in J. Mata-Alvarez (ed) (2003) *Biomethanization of the Organic Fraction of Municipal Solid Wastes*, London: IWA Publishing, pp.141-179.

Table 19: Biogas Production in Plants

Feedstock	Kitchen and Garden		Kitchen and Garden		Kitchen and Garden		Kitchen	Kitchen and Garden
Retention Time (days)	21	16	21	14	16	25	25	
Gas production per kilogram wet waste (m <sup>3</sup> biogas/tonne input waste)	120	110	150	122	138	140	170	
Methane production per tonne wet waste (m <sup>3</sup> methane / tonne input waste)	55%	56%	62%	60%	65%	61%	59%	
Methane production per tonne wet waste (m <sup>3</sup> methane / tonne input waste)	66	75		76	86	85	100	

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

The first two approaches give, in our view, values which are at the higher end of the likely range (though they are plausible). The last of the above methods – in some ways, the most simple – gives results which most closely resemble what is quoted by technology suppliers.

We have carried forward high, low and average values for carbon dioxide and methane emissions. These are then used as the basis for the calculations concerning energy and fuel recovery (see below).

### A.5.2.2 Trace Gases in Raw Biogas Generation

It is generally accepted that trace elements of gases such as hydrogen sulphide will be present in the gas generated by anaerobic digesters.

Schleiss notes:

*For emissions of hydrogen sulphide (H<sub>2</sub>S) from the rotting goods, the data situation is rather poor. As one of the few, but relatively high data, Gronauer et al (1997) cite up to 700g per t of input dry substance. That produces 2,852kg H<sub>2</sub>S per 10,000t with 40% dry substance.<sup>59</sup>*

The same value is used as for compost. This is surprising given the differences in the process, as well as the fact that H<sub>2</sub>S, when combusted, would give rise to SO<sub>2</sub> emissions. Ammonia and N<sub>2</sub>O emissions are also treated as for compost. For methane, Schleiss gives figures varying from 8.4-14.5kg of waste per tonne of waste input. Schleiss also gives figures of carbon monoxide varying from 0.041-0.166 kg per tonne input.

The ORWARE model notes that the emissions of hydrogen sulphide are dependent upon the presence or otherwise of precipitation chemicals which may remove the sulphur arising from degradation of proteins as iron sulphide. The model also held that any PAHs, phenols and PCBs would not be emitted as gases in the process but would be degraded in the digestion process by 12%, 13% and 45% respectively.<sup>60</sup>

<sup>59</sup> Konrad Schleiss (1999) Grüngutbewirtschaftung im Kanton Zürich aus betriebswirtschaftlicher und ökologischer Sicht: Situationsanalyse, Szenarioanalyse, ökonomische und ökologische Bewertung sowie Synthese mit MAUT, Dissertation ETH No 13,746, 1999.

<sup>60</sup> M. Dalemo (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.

**Error! Reference source not found.** gives manufacturers' data for the DRANCO plant in Salzburg. These appear to be raw gas data (i.e. before any combustion), as evidenced by the methane content. The combustion process and air pollution control technology will, together, determine the ultimate emissions to air.

**Table 17 Gaseous Emissions from DRANCO Plant in Salzburg**

Component	Unit	sample 1	sample 2
Water content	Vol %	6.5	6
methane	Vol %	57.2	54
CO <sub>2</sub>	Vol %	31	35.2
O <sub>2</sub>	Vol %	1.1	0.9
N <sub>2</sub>	Vol %	4.1	3.9
CO	µg/m <sup>3</sup>	nd	Nd
H <sub>2</sub> S	mg/m <sup>3</sup>	284	289
1,1,1,-Trichloroethane	µg/m <sup>3</sup>	nd	Nd
Trichloroethene	µg/m <sup>3</sup>	nd	nd
Tetrachloroethene	µg/m <sup>3</sup>	nd	nd
vinyl chloride	µg/m <sup>3</sup>	nd	nd
1,1-dichloroethene	µg/m <sup>3</sup>	nd	nd
dichloromethane	µg/m <sup>3</sup>	nd	nd
chloroform	µg/m <sup>3</sup>	2	nd
1,1-dichloroethane	µg/m <sup>3</sup>	nd	nd
1,2-dichloroethane	µg/m <sup>3</sup>	nd	nd
benzene a	µg/m <sup>3</sup>	70	50
toluene a	µg/m <sup>3</sup>	220	250
Ethylbenzene	µg/m <sup>3</sup>	610	630
M+p+o xylene	µg/m <sup>3</sup>	360	290
total Chlorine	µg/m <sup>3</sup>	1.5	nd
total Fluorine	µg/m <sup>3</sup>	nd	nd

Source: Environment Agency (2000) *Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion, R&D Project Record P1/392/4*

Neither the study of Baky and Eriksson, nor that of Finnvenden et al, estimate the trace gas composition in raw gas. Instead, the studies use conversion factors relating emissions to the energy content of the biogas used. Eunomia et al also did not generate figures for raw gas composition, but looked at the emissions from the whole process.<sup>61</sup>

<sup>61</sup> Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC Research & Consulting (2002) *Economic Analysis of Options for Managing Biodegradable Municipal Waste*, Final Report to the European Commission.

## Approach Taken in This Study

In this work, we have not sought to estimate raw gas composition. Below, we estimate other emissions dependent upon the utilisation of the biogas.

### A.5.3 Atmospheric Emissions from Complete Processes

From the perspective of the process as a whole, what matters are the emissions which are emitted to the atmosphere. These emissions depend critically upon the way in which the biogas is utilised.

There are two main options in use today:

- The first, and probably the most common, is to use the biogas to generate energy as either electricity, heat, or a combination of the two. In this case, the gas is usually used on-site;
- The second, becoming more common, is the use of biogas to power vehicles. In this case, the gas is usually cleaned prior to its utilisation off site as compressed natural gas.

Consequently, the ultimate emissions to atmosphere, and indeed, the emissions associated with the compensatory system, are different depending upon the utilisation route. Future applications include the use of biogas in fuel cell technology.

#### A.5.3.1 Biogas Used to Generate Energy

Where biogas is used to generate energy, the methane is largely converted to carbon dioxide. Other emissions will, however, be associated with the combustion process.

Eunomia et al used the data from White et al<sup>62</sup> for the gaseous emissions from anaerobic digestion (**Error! Reference source not found.**). These are the same data as were chosen for use by Aumonier on the basis of an earlier literature review.<sup>63</sup> There are a number of gaseous emissions that would be expected from the combustion process. These include heavy metals, dioxins, NO<sub>x</sub> (partly related to NH<sub>3</sub> in raw gas) and SO<sub>x</sub> (partly related to input H<sub>2</sub>S) but the emission levels of some of these species are very low.

White et al do not give emissions of CH<sub>4</sub> or N<sub>2</sub>O. One might expect near-zero emissions of methane if the combustion process was highly efficient and if capture of methane was complete. However data from other studies (such as ORWARE examined below) indicate non-zero emissions of these greenhouse gases and one would expect this to be the case in reality as complete capture of all methane in the whole process is unlikely. Furthermore, combustion of the gas seems likely to lead to some emissions of N<sub>2</sub>O. Therefore it is likely that White et al. underestimate the emissions of certain greenhouse gases.

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<sup>62</sup> P. R. White, M. Franke and P. Hindle (1995) *Integrated Solid Waste Management: A Lifecycle Inventory*, Blackie Academic & Professional, Chapman and Hall.

<sup>63</sup> S. Aumonier (1999) *Life-cycle Assessment of Anaerobic Digestion: A Literature Review*, Report for IEA Task 24.

Table 18: Emissions Data for Anaerobic Digestion (per tonne of waste) from White et al

Parameter	Value
Compost Production (kg)	300
Energy Consumption (electrical)	50 kWh
Energy Production	160 kWh
Air Emissions (g / tonne biowaste)	
CO <sub>2</sub>	440,000
CH <sub>4</sub>	0
NO <sub>x</sub>	10
N <sub>2</sub> O	0
SO <sub>x</sub>	2.5
HCl	0.011
HF	0.0021
H <sub>2</sub> S	0.033
HC	0.0023
Halogenated HC and PCBs	0.00073
Dioxins/furans (TEQ)	1E-08
Ammonia	ND
Cadmium	9.4E-07
Chromium	1.1E-07
Lead	8.5E-07
Mercury	6.9E-07
Zinc	1.3E-05

Source: White, P. R., M. Franke and P. Hindle (1995) *Integrated Solid Waste Management: A Lifecycle Inventory*, Blackie Academic & Professional, Chapman and Hall.

Both the approach of Finnveden et al and that of Baky and Eriksson were based upon modelling of emissions other than methane through the energy content of the biogas. The emissions factors are shown in **Error! Reference source not found.** below. Close examination shows, however, that the data used for the calculation are from a study which looked at the performance of gas engines from the late 1980s. It seems unlikely that this data would remain relevant in the present day since the performance of gas engines (in efficiency and in terms of emissions of, for example, NO<sub>x</sub>) is better than it was almost twenty years ago. The ORWARE figures for NO<sub>x</sub> are much higher than those from White et al (factor of 70 or so) or from Schleiss (factor of 18 or so). The figures for SO<sub>x</sub> are greater by a factor of 200 or so than those from White et al. This suggests that although the approach might be interesting, the values used are outdated.

Table 19: ORWARE Data Concerning Gaseous Emissions from Anaerobic Digestion

Gas	g per MJ of methane
CO <sub>2</sub>	85
CH <sub>4</sub>	0.1
NMVOCS	0.06
CO	0.25
NO <sub>x</sub>	0.20
N <sub>2</sub> O	0.03
SO <sub>x</sub>	0.15

Source: M. Dalemo (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.*

SO<sub>x</sub> emissions are generally a matter for process management. To the extent that they stem from the H<sub>2</sub>S in raw gas, the emissions are related to the use of, for example, precipitation salts which seek to precipitate out the sulphur emitted from the degradation of proteins as iron sulphide.

#### Approach Taken in This Study

In this study we have used an approach in which the emissions are related to the energetic content of the biogas. However, noting the considerable discrepancy between the sources, we have used a data set where emissions of NO<sub>x</sub>, N<sub>2</sub>O and SO<sub>x</sub> more closely reflect other studies (and more modern engines and emissions abatement).<sup>64</sup> These also reflect those figures quoted in the Draft BREF note for Waste Treatment in terms of grams per tonne of waste.<sup>65</sup>

#### A.5.3.2 Biogas Used as 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<sup>66</sup>. However, the 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<sub>4</sub> and CO<sub>2</sub>, smaller amounts of H<sub>2</sub>S and NH<sub>3</sub>, and trace amounts of H<sub>2</sub>, N<sub>2</sub>, CO and O<sub>2</sub>. Across different countries the minimum methane content specification is between 95% and 97%, the permissible remainder being mostly CO<sub>2</sub>. Typically also, the vapour content must be lower than 15 mg/Nm<sup>3</sup>, the H<sub>2</sub>S content should

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<sup>64</sup> It is interesting to note, for example, that whilst Baky and Eriksson use the same emissions factors as Finnvenden et al, relating to engines from the 1980s, Baky and Eriksson's later study assumed a much higher efficiency of conversion to electricity (at 38% as opposed to the 30% used by Finnvenden et al). Presumably, this reflects some evolution in the engine efficiency over time. It is strange, therefore, that the source of emissions data was not also reviewed.

<sup>65</sup> European Commission (2005) Integrated Pollution Prevention and Control, Draft Reference Document on Best Available Technologies for the Waste Treatment Industries, Final Draft, May 2005.

<sup>66</sup> IEA Bioenergy (u.d.) *Biogas Upgrading and Utilisation*, Task 24: Energy from Biological Conversion Of Organic Waste.

not exceed 100mg/Nm<sup>3</sup> and the particle size is limited at 40 microns. The typical sequence for gas preparation<sup>67</sup> is:

- compression at 15-20 bars;
- desulphurization and decarbonation by water scrubbing (loss of 8% of methane);
- dehydration by the process Pressure System Adsorption;
- removal of halogens through contact with activated carbon; and
- compression up to 250-350 bars.

One study suggests that these steps require an input of electricity amounting to around 6% of the energy produced.<sup>68</sup> Another study suggests that 1% of methane will be emitted during the gas purification process.<sup>69</sup>

Due to the lack of direct data concerning gas powered vehicles running on biomass derived fuel it is assumed that the emissions are comparable to vehicles running on compressed natural gas (CNG). This is reasonable since the typical chemical composition of natural gas is mostly CH<sub>4</sub>, (between 87% and 96%, typically 95%) with some ethane (C<sub>2</sub>H<sub>6</sub>) and smaller amounts of N<sub>2</sub> and CO<sub>2</sub>.<sup>70</sup>

A literature review of the emissions from gas operated vehicles compared to their conventional fossil fuel counterparts revealed considerable variation in the data. A major problem is that there is considerable variation in the size and use of vehicles identified. A bus under constant acceleration and braking travelling around a city has a greatly different emission profile to a car used for business. As such it can be difficult to draw comparisons between different data sources. Furthermore, both liquid fuel and gas operated vehicles have seen considerable improvements in emissions over the past few years, and yet there remains considerable variation in performance between currently available vehicles.

It should be noted that all heavy duty European vehicles are equipped with a catalyser which is not always the case with US vehicles. It is important not to allow comparisons between data sources of old against recent vehicles or between vehicles of different type or operation. As such the literature contains many data sources which can lead to confusion or invalid conclusions.

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.

A report by VTT Processes, Nylund et. al.<sup>71</sup> provides some general observations concerning the operation of vehicles of different fuel types. These are shown in **Error! Reference source not found.**

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<sup>67</sup> Trendsetter (2003) *Biogas as Vehicle Fuel. A European Overview*, Report No 2003:3. Stockholm, Sweden.

<sup>68</sup> A. Baky and O. Eriksson (2003) *Systems Analysis of Organic Waste Management in Denmark*, *Environmental Project No. 822*, Copenhagen: Danish EPA.

<sup>69</sup> Goran Finnveden, Jessica Johansson, Per Lind and Asa Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, Forskningsgruppen for Miljostrategiska Studier, FMS 137, August 2000.

<sup>70</sup> Uniongas website, May 2005, [www.uniongas.com/aboutus/aboutng/composition.asp](http://www.uniongas.com/aboutus/aboutng/composition.asp)

<sup>71</sup> N. Nylund, K. Erkkilä, M. Lappi, and M. Ikonen (2004) *Transit Bus Emission Study: Comparison of Emissions from Diesel and Natural Gas Buses*, VTT Processes, October 2004

Table 20: Emission Overview Comparison between Diesel and CNG Vehicles

Emission component	Significance of component*	Diesel (continuously regenerating trap particle filter)	CNG (stoichiometric)
CO	low	lower	
THC	low	lower	
NMHC	high/moderate	similar	similar
NOx	moderate		much lower
NO2	high		much lower
PM mass	high		lower
CO <sub>2</sub>	moderate	similar	similar**
Engine efficiency	moderate	better	
Total particle numbers	high	similar	similar***
Nanoparticle numbers	high	similar	similar***
Total PAH	high/moderate		much lower
Carcinogenic PAH	high		lower
Mutagenicity	high		much lower
Aldehydes	high		much lower

\* For urban buses, giving priority to toxic emissions.

\*\* Lower for stoichiometric, higher for lean-mix (combustion).

\*\*\* Similar for the lean-mix vehicle, higher for the stoichiometric vehicle with roof mounted catalyst.

The VTT report also lists tailpipe emissions for buses operating on different fuels in different engine and post combustion emission cleaning configurations, and includes comparison to references from other literature. This is reproduced in **Error! Reference source not found..**

This data shows some valuable recent emission levels for buses operating with the different available gas cleaning mechanisms. The potentially significant variation between different vehicle emission studies is also highlighted. Unfortunately this does not give a complete picture of all the emissions of interest. What is needed is a complete dataset which compares a full range of emissions for equivalent vehicles using CNG and conventional fuel. The US study by Wang et al<sup>72</sup> provides this level of analysis and allows for desired comparisons to be made.

The study calculates emissions (using their GREET model) through a full fuel cycle for cars operating with different fuels - summary data is reproduced in **Error! Reference source not found..** The feedstock stage includes all activities from resource extraction to delivery at fuel production plants. The fuel preparation stage accounts for emissions at the fuel production plant up until delivery at the refuelling station. The vehicle operation stage includes activities at the refuelling station though to completion of onboard fuel combustion.

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<sup>72</sup> M. Q. Wang and H. S. Huang (1999) *A Full Fuel-cycle Analysis of Energy and Emissions Impacts of Transportation Fuels Produced from Natural Gas*, Centre for Transportation Research, Energy Systems Division, Argonne National Laboratory, Illinois, December 1999

Table 21: Emissions from Gasoline and CNG Vehicles

Study/ Component	NOx (g/km)	NO <sub>2</sub> (g/km)	PM (g/km)	Formaldehyde (mg/km)	1,3- butadiene (mg/km)	Benzene (g/km)	PAHs (µg/km)
<b>VTT Processes</b>							
Diesel	8	0.1	0.17	37	8	3	613
Diesel, oxidation catalyst	8.5	0.3	0.12	16	0	1	427
Diesel (continuously regenerating trap particle filter)	9	0.8	0.02	5	0	1	94
Lean burn CNG oxidation catalyst	7	0.3	0.01	6	0	0	8
Stoichiometric CNG	2	0.05	0.005	0.1	0	0	7
<b>ARB 2002<sup>73</sup></b>							
Diesel, oxidation catalyst	19	-	0.08	?	0	2	250
Diesel, diesel particle filter	20	10	0.01	3	0	0.4	90
Lean burn CNG	11	-	0.02	500	1	2	110
Lean burn CNG, oxidation catalyst	8	-	0.01	30	0	0.6	30
<b>International 2003<sup>74</sup></b>							
Diesel	9	1	0.12	25	0	3	-
Diesel, diesel particle filter	6	3	0.01	5	1	0	-
Lean burn CNG	10	1	0.03	300	3	3	-

### Approach Taken in This Study

Compiling these statistics and using a fuel efficiency of 22.4mpg for the petrol vehicle<sup>75</sup> and calculating the equivalent efficiency for the CNG vehicle gives results for emissions in terms of g/MJ of energy. These figures can be used to generate offsets relative to the energy input of the fuel. We have assumed that vehicles are equally efficient in the use of fuel energy irrespective of the fuel source (conversion efficiency is about 20%). Some justification for this assumption is provided from the first rows of **Error! Reference source not found.**

We have assumed that electricity equivalent to 3% of the energy delivered is used to compress and clean the gas. Emissions for this are as with the energy generation module.

<sup>73</sup> Air Resources Board (ARB) (2002) *Study of CNG and Diesel Transit Bus Emissions*, Air Resources Board, California Environmental Protection Agency, April 2002. <http://www.arb.ca.gov/research/cng-diesel/cng-diesel.htm>

<sup>74</sup> International (2003) *School Bus Emission Study*, Presentation by Warren Slodowske, International Truck and Engine Company, from Proceedings of DEER 2003. 9<sup>th</sup> Diesel Engine Emission Reduction Conference. August 24-28, Newport, Rhode Island, USA.

<sup>75</sup> M. Q. Wang and H. S. Huang (1999) *A Full Fuel-cycle Analysis of Energy and Emissions Impacts of Transportation Fuels Produced from Natural Gas*, Centre for Transportation Research, Energy Systems Division, Argonne National Laboratory, Illinois, December 1999.

Table 22: Emissions from Gasoline and CNG Vehicles

		Conventional Gasoline vehicle			Compressed natural gas vehicle		
		Sourcing feedstock	Fuel preparation	Vehicle operation	Sourcing feedstock	Fuel preparation	Vehicle operation
Total energy	Btu/km	118	708	3204	344	446	3445
Fossil fuels	Btu/km	104	679	3204	326	342	3445
Petroleum	Btu/km	30	330	3152	16	7	0
VOC: Total	g/km	0.011	0.045	0.129	0.017	0.011	0.038
VOC: Urban	g/km	0	0.016	0.129	0.001	0.009	0.038
CO: Total	g/km	0.050	0.030	3.428	0.070	0.026	2.743
CO: Urban	g/km	0	0.004	3.428	0.002	0.019	2.743
NOx: Total	g/km	0.032	0.084	0.171	0.107	0.101	0.171
NOx: Urban	g/km	0	0.007	0.171	0.007	0.052	0.171
PM10: Total	g/km	0.002	0.009	0.205	0.003	0.004	0.014
PM10: Urban	g/km	0	0.001	0.205	0	0.001	0.014
SOx: Total	g/km	0.009	0.062	0.031	0.012	0.058	0.001
SOx: Urban	g/km	0	0.001	0.031	0	0	0.001
CH4	g/km	0.290	0.070	0.052	0.680	0.066	0.522
N2O	g/km	0	0.001	0.017	0.001	0.001	0.017
CO2	g/km	11	41	242	23	28	205
GHGs	g/km	17	43	249	38	29	221

#### A.5.4 Process Energy Use and Generation

Finnvenden et al assumed that electricity consumed in the anaerobic digestion process was 31 MJ/ton organic household waste (or around 9kWh per tonne).<sup>76</sup> They used a value for the energy content of the methane gas collected from anaerobic digestion of 3,743 MJ/tonne food waste. 30% of the heat was assumed to be converted to electricity whilst it was assumed that 495 MJ of the remainder would be used internally at the plant (see above).

Baky and Eriksson assumed that energy for pumping and mixing of material etc. required approximately 5% of produced energy.<sup>77</sup> Heat requirement in the digester was calculated from a mechanistic approach. Heat consumption was assumed to depend upon the temperature of the digested material and losses through the digester reactor during digestion. It was also assumed that 50% of the heat in the material leaving the digester could be reused through use of a heat exchanger thereby reducing the temperature from digester temperature to ambient temperature. This would reduce the need for additional heat to maintain temperature in the reactor. They assumed an efficiency of electricity

<sup>76</sup> Goran Finnvenden, Jessica Johansson, Per Lind and Asa Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, Forskningsgruppen for Miljostrategiska Studier, FMS 137, August 2000.

<sup>77</sup> A. Baky and O. Eriksson (2003) *Systems Analysis of Organic Waste Management in Denmark*, Environmental Project No. 822, Copenhagen: Danish EPA.

generation of 38%. They assumed a further 52% of the energy would be recovered as heat with the remaining 10% being losses.

Where biogas was assumed to be used as vehicle fuel, they assumed that electricity would be required equivalent to 3% of the energy produced.

Eunomia used ranges, based upon a review of literature, to model the energy generated and exported (in both electrical and heat forms).<sup>78</sup> The study assumed that electricity generation would occur at 30% efficiency and that heat production, under CHP operation would be 70% of the energy content of the biogas, but that only 80% would be available for export. Electricity generation was assumed to be 81 to 176 kWh per tonne of waste and heat export was assumed to be 151 to 329 kWh.

### **Approach Taken in This Study**

In this study, we have assumed that electricity is recovered at 35% efficiency. We have assumed that 7% of the heat content of the biogas is used by the plant itself. Where biogas is used as vehicle fuel, we have assumed the same fraction is used to heat the plant but electricity equivalent to 3% of the energy content of the gas is used to compress and clean it (as discussed above).

## **A.5.5 Post-Digestion Treatment**

The literature concerning the emissions of various gases in the post-treatment stages of anaerobic digestion is not especially rich. There are, of course, various options for dealing with the material which remains post-digestion.

In essence, 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 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. One of the reasons for stabilising the solids from digestion is to reduce the potential for nitrogen to leach following application – the stabilisation process makes for an amendment with high organic matter content, but with reduced availability of nitrogen. The level of nutrients such as phosphorous is much lower than in, say, sludge based materials, making it possible to apply more organic material without creating problems of groundwater pollution / surface water run-off.

### **Approach Taken in This Study**

In this study, we have assumed that the digestion process is followed by an aerobic treatment phase, and that the early stages of the aerobic treatment employ a biofilter. This adds slightly to the costs of treatment, and some equipment suppliers prefer to make the slurry available without such a treatment step.

Additional emissions of CO<sub>2</sub> and NH<sub>3</sub> are assumed to occur in the aerobic phase, and some additional CH<sub>4</sub> is assumed to be emitted in the transition from the anaerobic to the aerobic phase. The use of the biofilter necessitates the use of some additional electricity.

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<sup>78</sup> Eunomia (2004) *Feasibility Study Concerning Anaerobic Digestion in Northern Ireland, Final Report for Bryson House, ARENA Network and NI2000.*

The assumptions made are, essentially, guesstimates. However, they mark an improvement on the situation where no assumptions regarding the post-anaerobic treatment phase are concerned.

Also implicit is that the aqueous residues are recycled either in the process or into agriculture where surplus is available. No benefits or costs have been assigned to this.

## A.5.6 Physical Characteristics

In recent work in the Northern Ireland context, suppliers offered information on the characteristics of their output, and these are shown below in Table 26.

**Table 23: Nutrient and Physical Characteristics of Compost Derived from Digestate**

	Dry Matter	Organic Matter	Total N	N-NO3	N-NH4	K	P	pH
	% TM	%TS	% TM	% TM	% TM	% TM	% TM	
Supplier 1	35	72	1.71		0.03	0.33	0.45	7.4
	% TM	% TM	% DM	% TM	% TM	g/kg DM	g/kg DM	
Supplier 2	>50	<50	>1	>0.04	<0.3	10	6	8.2
	% TM		% TM			g/kg DM	g/kg DM	
Supplier 3	56		1.12			12.4	5.1	
	% TM		% TM	mg/l compost	mg/l compost	% TM	% TM	
Supplier 4	57.8		0.82	53	215	0.34	0.58	7.9

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

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 300kg per tonne of waste input. The nutrient content is deemed to be the same as in the case of composting of kitchen and garden waste.

As regards the quantity of material produced, it is often stated that the mass loss through digestion processes will be much greater than is the case with composting. This may or may not be true depending upon a variety of (both) process variables. In this case, we have assumed that the mass is slightly lower than for the equivalent aerobic composting process. It should be noted, however, that concentrations of the various components of the digestate, as with compost, are likely to vary from time to time and from one process to another.

### A.5.6.1 Contaminants

The quality of solid residues from 4 different suppliers is shown in **Error! Reference source not found..** The one company who preferred to spread slurry on land did not give details of the content of the slurry.

The figures are shown alongside standards under the Biowaste Directive and the PAS100. It shows that all suppliers are compliant with Class 1 of the 2<sup>nd</sup> Draft of the Biowaste Directive with the exception of two suppliers who appear to fail in respect of cadmium limit values. All are well below the WRAP PAS 100 standard.

Table 24: Levels of Heavy Metal Contamination in Compost from Digestion Residual (mg/kg dry matter)

	Zn	Cu	Ni	Cd	Pb	Hg	Cr
Class 1	200	100	50	0.7	100	0.5	100
Class 2	400	150	75	1.5	150	1	150
Stabilised Biowaste	1500	600	150	5	500	5	600
PAS 100	400	200	50	1.5	200	1	100
Supplier 1	132	56	26	1.2	72	0.15	43
Supplier 2	<200	<50	<15	<.5	<60		
Supplier 3	194	27	8	0.5	67	0.10	23
Supplier 4	180	32	8	1	97	0.15	23

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

## A.6.0 EFFECTS OF COMPOST / DIGESTATE APPLICATION TO LAND

It is important to understand that the process we are trying to place a value upon – the application of organic matter to the soil – is one that has extraordinarily complex ramifications for the soil. This is true even if the application occurs in isolation, but the interactions between processes which may occur simultaneously with the aim of improving soil quality increases the complexity of the analysis.

Sequestration of carbon in the soil depends not only upon what is applied to the soil (both organic carbon and nitrogen), but also upon the way in which the soil is tilled. Many of the most vociferous proponents of soil management as a means of sequestering carbon are ‘no-till’ farmers from the United States. Indeed, one Canadian energy and utility company, TransAlta, established the Saskatchewan Soil Enhancement Project, which promotes ‘low-disturbance direct seeding’. The aim is to offset increased greenhouse gas emissions from the company over time. Equally, no-till practices may, at least in the short-term, require changes in management of weeds, and in some areas, this has increased use of glyphosate herbicides.

The point to be made is that the soil is a living ecosystem. However, soils are relatively poorly researched so inevitably, there are interactions occurring which are only relatively poorly understood.

The application of stabilised compost to soil can affect soil fertility by modifying soil chemical, physical and biological properties. Compost can result in the storage of inorganic plant nutrients, affect the soil’s ion exchange capacity, chelating ability and buffering capacity. Physical changes include changes to the soil’s bulk density, structure, strength, and water management. The addition of compost can also increase the nutritional base for soil micro-organisms. Once incorporated, composts become part of the soil humus and therefore have a long-term effect on soil properties.

Under natural conditions a balance is established whereby the soil supports as much plant growth as it can nourish. In modern agricultural systems the soil is often artificially altered by using mineral fertilisers to increase its capacity to support plant growth.

There are many factors that affect soil fertility including soil pH, supply of mineral nutrient elements, moisture content, temperature, composition of the soil, strength and biotic factors. A mature compost will affect all of these factors and will therefore greatly alter the fertility of the soil. When compost is applied to the soil, micro organisms will continue to degrade the humified compost releasing mineral nutrients through a process called mineralisation. This process takes place slowly in temperate climates and at an increased rate under warm conditions. In temperate climates, a proportion of the organic matter becomes stabilised as soil humus, which can result in one application of compost having a benefit lasting several years.

Unlike mineral fertilisers, the use of compost does not provide a specific amount of N, P or K that will be immediately available to the growing plant. Compared to mineral fertilisers, composts provide low levels of N, P and K. However, the addition of compost can provide essential trace minerals to the soil (calcium, sulphur, iron, boron, molybdenum and zinc) that are not supplied when mineral fertilisers are added.

Although compost does not immediately provide minerals to plants, the application of compost can enhance nutrient uptake by reducing leaching of minerals. Losses of nutrients by leaching can be reduced by increasing the soil organic matter content by the addition of compost. Some nutrients in the water soluble form required by plants are readily leached from mineral soil particles whereas they are effectively held on the surface of humified organic matter.

**Error! Reference source not found.** provides a summary of some of the advantages and disadvantages of using compost compared to mineral fertilisers in agriculture. The composting industry needs to overcome the disadvantages and promote the advantages of compost if it is to convince farmers that compost is a viable alternative or can be used in conjunction with conventional fertilisers. This will effectively ‘internalise’, in market prices, some of the external benefits which we are about to explore.

**Table 28: Advantages And Disadvantages Of Using Compost Compared To Mineral Fertilisers.**

Material	Advantages	Disadvantages
Compost	Improves soil structure	Dilute source of nutrients
	Controls erosion	Even application can be difficult
	Supplies wide range of nutrients	High C:N ratios may rob soil of N
	Method of waste disposal	
	Increases activity (and presence) of soil micro-organisms	
Mineral fertilisers	Convenient	Easily leached
	Lower transport and handling costs	Overuse may lead to breakdown of soil structure
	Quick crop response	Supply only major nutrients

The addition of compost to soil results in a reduction in bulk density, an increase in soil porosity and increased water retention. All these factors have a positive effect on plant growth and subsequent crop yields. They may also act to reduce the potential and / or frequency of flooding in periods of high rainfall, and for soil erosion.

The issue of soil erosion is more relevant to some parts of the UK than others. **Error! Reference source not found.** below gives some indication of declining soil organic matter status, but the data is somewhat dated now. Defra has several research projects in progress investigating soil organic matter status.

The use of compost in agriculture is particularly suited to organic farming methods as it offers several advantages over the use of mineral fertilisers. Positive yield responses have been observed after applications of compost. However, applications of immature composts can have a negative effect on crop yield, as soil nitrogen is utilised by micro organisms degrading the compost instead of being available to the crops. This can be avoided by adding either mature compost or by applying immature compost well ahead of planting to allow for additional decomposition. Organic farming certification bodies have shown concern regarding using composts derived from municipal waste since there is believed to be potential for contamination with genetically modified materials, specifically, seeds which may still be capable of germination after the composting process.

Table 25: Change in Organic Matter Status of Soils in UK (% of samples in category), 1979-1995

Category	1979-1981	1995
Less than 3.6% organic matter	32%	41%
3.6%-7% organic matter	46%	47%
more than 7% organic matter	22%	12%

Note: Soil organic matter is expressed as organic carbon percentage of non-humose mineral arable and short-term grassland topsoils

Source: National Soil Resources Institute

### A.6.1 Displacement of Alternative Nutrient Sources

When compost is applied to the soil, it may displace nutrients which are otherwise applied through other means. These other means may be animal manures, but more typically, soil nutrients are supplied through use of synthetic fertilisers. Exceptionally, it may supply nutrients where none would otherwise be applied, whilst in organic systems, the concept of 'displacement' is likely to have a different meaning owing to the limitations on use of synthetic fertilisers. Typically, however, it seems reasonable to assume that the product being displaced is synthetic fertiliser. This will not always be true.

We start by assuming in the analysis that nutrients are displaced on a one-for-one basis from the perspective of plant uptake. Taking the view from the perspective of the plant is important since the rate at which nutrients are leached from humus is lower than the rate at which they might be leached from synthetic fertiliser. More of the nutrient in synthetic form would be required to be applied to have the equivalent mineral fertilisation effect, although how much more depends upon rainfall after application and the stage of growth of the crop. Note that composts from biowastes (as opposed to green waste only) are likely to have lower C:N ratios so the likelihood that nitrogen becomes locked in the soil due to stimulation of microbial activity is much reduced.

The assumption of 'one-for-one' nutrient displacement is, however, an unrealistic one to the extent that one is implying a perfect optimisation of the replacement process. Such a situation is only likely to be achieved where farmers are well informed about the nutrient content of the matter being applied. We have therefore used a range of 40% to 100% displacement.

Note that the degree to which nutrient replacement is achieved (which is likely to be a function of the available information concerning composts, as well as climatic variables) may well affect the degree to which one attributes any private benefits derived from avoidance of fertiliser use as an 'externality'. Arguably, where the information available allows greater optimisation with respect to replacement, then the benefits from compost use ought to be internalised in market prices. Where the replacement occurs in a more haphazard manner, one might argue that the benefit is an 'external' one arising through improved soil fertility.

We have assumed that 10 tonnes of dry matter is applied per hectare. We assume nutrient contents as in Section **Error! Reference source not found.**

The mineralisation rate of the nutrients is assumed to be 30%. This determines the time profile of the displacement effect (which in turn affects the external benefits associated with displacement via the discounting mechanism).

For synthetic fertilisers, a loss rate of 23% is assumed for nitrogenous fertilisers.<sup>79</sup> The nitrogen in compost is assumed to be 100% available to plants over time, with the mineralisation rate determining the rate at which the nutrient is made available. This means that more nutrient has to be applied in a given year in the synthetic form than would be available in mineralised form from the composted materials. For an application of 10 tonnes dry matter per annum in one year, the N displacement would follow the evolution set out in **Error! Reference source not found.** and **Error! Reference source not found.** below. Equivalent projections for P and K displacement have been calculated using mineralization rates of 70% for both.

**Table 26: Evolution In N Displacement Associated With 10 Tonnes Dry Matter Of Compost Applied To Farmland, Southern Member State Case**

Year	Displacement of N (kg)	Cumulative Displacement
1	50.9	50.9
2	35.7	86.6
3	25.0	111.6
4	17.5	129.0
5	12.2	141.3
6	8.6	149.8
7	6.0	155.8
8	4.2	160.0
9	2.9	163.0
10	2.1	165.0

## A.6.2 Avoided External Costs of Fertiliser Manufacture

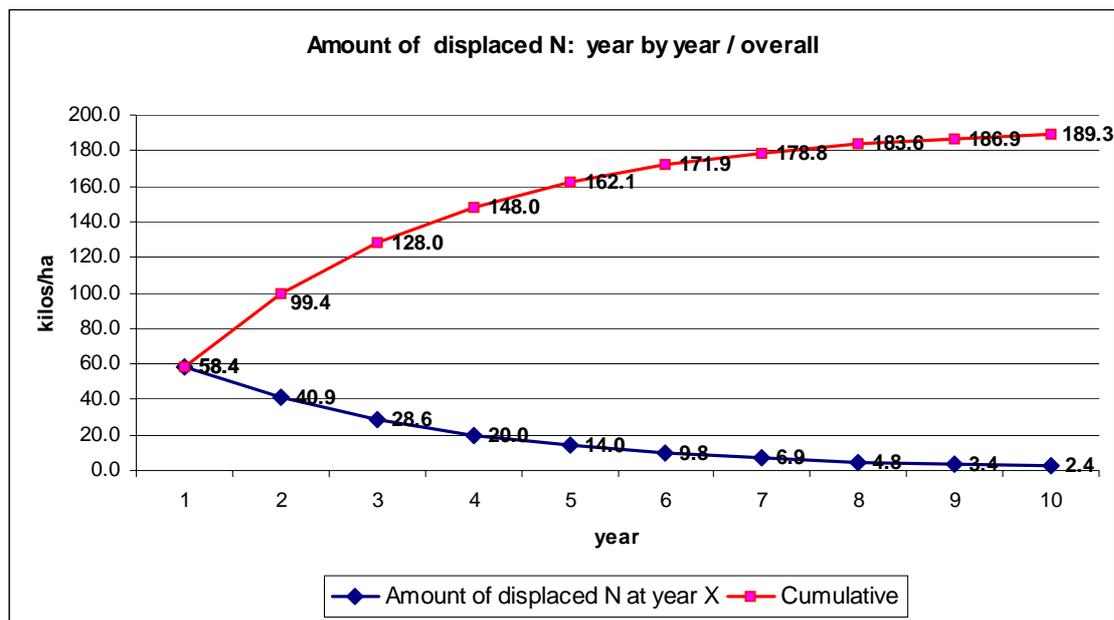
The use of compost and matured digestate as a replacement for fertiliser will displace the pollution and other impacts associated with fertiliser production. These are discussed in this section to derive an externality cost per tonne of fertiliser produced and therefore a subsequent benefit per tonne of compost/digestate applied.

There are a number of different routes to NPK fertiliser production. The most widespread are the mixed acid and nitrophosphate routes. This study used two sources of data, the first on Best Available Technologies from EFMA (European Fertilisers Manufacturing Association), the second from Davies and Haglund.<sup>80</sup> The first of these sources refers to production via the mixed acid route. There are three further distinctions within this route (granulation with a pipe reactor system, drum granulation with ammoniation and digestion). The data concerning emissions have been taken from those for the 'digestion' process because it is the only process for which this study has obtained data associated with all the required raw materials. The EFMA booklets suggest that these three processes cover the majority of NPK fertiliser production in Europe. The data from Davies and Haglund are calculated from, separately, production of ammonium nitrate, Triple Superphosphate and (for potassium), PK fertiliser (22% P<sub>2</sub>O<sub>5</sub>, 22% K<sub>2</sub>O).

<sup>79</sup> This is the loss rate from Hydro Agri Europe (1995) *Important Questions on Fertilizer and the Environment*, Brussels: Hydro Agri Europe.

<sup>80</sup> J. Davies and C. Haglund (1999) *Life Cycle Inventory (LCI) of Fertiliser Production – Fertiliser Products Used in Sweden and Western Europe*, SIKreport, No 654 1999, SIK The Swedish Institute for food and biotechnology, Gothenburg, Sweden.

Figure 11: Evolution of Nitrogen Displacement Over Time from Single Compost Application



The EFMA booklets provide gaseous emissions and energy consumption data associated with sulphuric acid, nitric acid, phosphoric acid and ammonia production, the base acids used in mixed acid production of NPK (15:15:15). The data also quotes the raw material requirement for the mixed acid route, in terms of sulphuric acid, nitric acid, phosphoric acid, ammonia and phosphate rock. Having derived the emissions and energy requirement associated with the production of each of these materials (e.g. extraction in terms of phosphate rock), these were factored according to the relative proportions used in the NPK (15:15:15) fertiliser. It was then assumed that for each of the nutrient components, that one-third of the processing requirement was attributable to the manufacture of 150 kg N, one-third to the manufacture of 150kg K<sub>2</sub>O portion and 150kg P<sub>2</sub>O<sub>5</sub>. Hence, through this attribution process, the levels from the mixed acid route itself are apportioned to the different nutrients (for the purpose of displacement calculations).

Using the two data sources described, we have calculated (for each) emissions associated with N, P and K production. These figures are multiplied by unit damage costs to estimate the external costs of production. These are, in turn, multiplied by avoided N, P and K quantities to arrive at a benefit per tonne of compost. The low externality value is the minimum value derived from the two datasets, the medium is the average of the medium values derived from the two datasets, and the maximum figure is the maximum of the two values derived from the datasets.

Mining phosphate rock is an energy intensive activity and approximately 3.3 tonnes of phosphate rock are required to produce one tonne of phosphorous pentoxide (P<sub>2</sub>O<sub>5</sub>) (100%).<sup>81</sup> Energy use for producing phosphate rock has been estimated at 73.5 kWh/tonne.<sup>82</sup> In a previous report, Eunomia simply attributed additional energy

<sup>81</sup> B. Bocoum and W. C. Labys (1993) Modelling the Economic Impacts of Further Mineral Processing: the Case of Zambia and Morocco. *Resources Policy*, **19**, (4), pp.247-63.

<sup>82</sup> UNEP and UNIDO (United Nations Environment Programme And United Nations Industrial Development Organisation) (1998). *Mineral Fertilizer Production and the Environment – Part 1: The Fertilizer Industry's*

consumption for phosphate fertiliser on this basis. However, in this report, we have used the estimated damages associated with phosphate rock extraction from Nolan-ITU.<sup>83</sup> The figure quoted there is converted to UK sterling and adjusted to current sterling using UK GDP deflators.

### A.6.3 Phosphate Fertiliser - Phosphogypsum and Process Wastewater Disposal

Soulsby et al site a major review, undertaken by the US Environmental Protection Agency (EPA), of the production and environmental impacts of phosphoric acid production at 21 locations in the USA.<sup>84</sup> Significant impacts of process wastewater and phosphogypsum disposal have been identified. The study reports that no economic valuation studies are available in the literature that can provide support in estimating the monetary value of the externalities associated with the processes. However, based on US data, the same study looked at the costs to industry of the environmental regulation involved.

The US EPA (1990) developed various possible regulatory scenarios for the control of phosphogypsum and process wastewater disposal. The costs of the compliance scenarios at the 21 locations range from \$380 million (£233 million) p.a. to nearly \$1 billion (£613 billion) p.a. Taking the lower value, the study attributed a unit cost for compliance of £27.56/tonne of phosphate fertiliser. The study assumed a concentration of P<sub>2</sub>O<sub>5</sub> in phosphoric acid of 32% and a concentration of 46% of P<sub>2</sub>O<sub>5</sub> in the product. This would be equivalent to £59.91 per tonne P<sub>2</sub>O<sub>5</sub>. We have used this as an admittedly crude estimate of the externalities associated with P<sub>2</sub>O<sub>5</sub> production (which may be avoided when compost is applied).

### A.6.4 Greenhouse Gases Avoided from Mineral Fertiliser Applications

Nitrous oxide emissions from soil are complex since the gas is simultaneously produced and consumed in soils through processes of denitrification, nitrification, nitrate dissimilation and nitrate assimilation. The rates at which these processes occur are affected by temperature, moisture, the presence of plants and the soil composition, as well as the (related) activity of bacteria in the soil column.

It is generally accepted that nitrogenous fertilisers increase fluxes of N<sub>2</sub>O. Different fertilisers appear to be more or less susceptible to the loss of nitrogen as nitrous oxide. Ammonia products appear most susceptible, with anhydrous ammonia and aqua ammonia losing between 1% and 5% of nitrogen as nitrous oxide. Other products such as sodium

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Manufacturing Processes and Environmental Issues. In collaboration with the International Fertilizer Industry Association. Technical Report Number 26 – Part 1. UNEP and UNIDO, Paris and Vienna. 1998.

<sup>83</sup> Nolan-ITU (2004) *TBL Assessment of Garden Organics Management*, Final Report to the NSW Dept of Environment and Conservation, Sustainability Programs Division, May 2004.

<sup>84</sup> P. G. Soulsby, G.A.W. Hickman and P.L. McMahon (2000) *The Environmental and Agricultural Economics of Recovering Value from Recycling Treated Biosolids to Land*, Paper prepared for presentation at the Chartered Institution of Water and Environmental Management / Aqua Enviro Technology Transfer conference 'Wastewater Treatment: Standards and Technologies to Meet the Challenges of the 21st Century', Leeds, 4 – 7 April, 2000.

nitrate appear to lose much less nitrogen in this way.<sup>85</sup> The emissions depend upon temperature, soil moisture, fertiliser type, fertiliser amount, the timing and mode of application, and the type of soil (including its pH) and crop cultivated.<sup>86</sup> A Dutch study cites figures for N<sub>2</sub>O losses as between 1 and 3% of mineral N applied.<sup>87</sup>

The following nitrogen volatilisation rates have been derived using the MANNER Model<sup>88</sup>. The application of slurry (6% dry matter) onto grassland in the spring at a rate to supply 250kg of fresh nitrogen, results in 46kg of nitrogen per hectare being volatilised. If fresh farm-yard-manure were applied and incorporated within 3-5 days, and applied in the spring to supply 250kg of nitrogen, the nitrogen volatilisation is approximately 32kg/ha. By comparison, compost has a very stable nitrogen content, on average only 0.7% of the nitrogen in compost is of a mineral from (NO<sub>3</sub>-N and NH<sub>4</sub>-N).<sup>89</sup>

### External Costs

The valuation of reduced N<sub>2</sub>O emissions from fertiliser applications clearly depends upon:

- The rate at which one assumes compost replaces nitrate fertilisers and / or other products (which, as discussed above, is not known with certainty, and is likely to vary with knowledge of the product and familiarity with its use); and
- The relative rates of N<sub>2</sub>O emissions from compost and from the displaced products.

Both of these are the subject of considerable uncertainty so the attempt being made here should be understood as a first attempt to understand what the magnitude of these external cost savings might be.

In the analysis, we assume that in the 'Low' case, relative to compost, 0.5% of nitrogen applied as fertiliser is lost as N<sub>2</sub>O. In the 'High' case, we assume a figure of 2% is lost. The low case is also subject to the '40% displacement' issue discussed above, whilst in the high case it is assumed that nitrogen in compost replaces for fertiliser on a one-to-one basis. These are combined with the time profile of N replacement for the compost as outlined in earlier sections. Hence, the externality also depends upon the mineralisation rate.

## A.6.5 Disease Suppression

It has been shown that compost can help to control plant diseases and subsequently reduce crop losses in both agriculture and horticulture. Disease control in compost has been attributed to four main mechanisms these are;

1. Successful competition for nutrients by beneficial micro-organisms.

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<sup>85</sup> D. Lashof and D. Tirpak (1990) (eds.) *Policy Options for Stabilising Global Climate*, London: Hemisphere; Ehrlich, A. (1990) *Agricultural Contributions to Global Warming*, in J. Leggett (1990) *Global Warming: The Greenpeace Report*, Oxford: Oxford University Press.

<sup>86</sup> I. P. McTaggart, B. C. Ball, and C. A. Watson (1998) *Influence on Land Use in Scotland on Soil N<sub>2</sub>O Emissions and Atmospheric CH<sub>4</sub> Uptake*, British Society of Soil Science Annual Meeting, Queen's University, Belfast, September 1998.

<sup>87</sup> A. R. Mosier (1993) *Nitrous Oxide Emissions from Agricultural Soils*, Proceedings of Methane and Nitrous Oxide - Methods of National Emissions Inventories and Options for Control, RIVM, Netherlands.

<sup>88</sup> The MANNER Model (MANure Nitrogen Evaluation Routine) is a computer model developed by ADAS, UK to help improve efficiency of animal manure use on farms.

<sup>89</sup> Figure calculated using data from HDRA Consultants Compost Analysis Database which contains data from over 80 different composts.

2. Antibiotic production by beneficial micro-organisms.
3. Successful predation against pathogens by beneficial micro-organisms.
4. Activation of disease resistant genes in plants by composts.

Compost quality is critical to the disease suppressive characteristics of compost. Composts that are allowed to mature have greater suppressive qualities than immature composts. The compost production method also appears to have an impact on suppressive qualities, with compost produced by an open-air method of composting have more suppressive properties than those produced by in-vessel composting methods and of those being matured undercover.

Maintaining consistent compost quality is vital. Suppressiveness is not a consistent quality, therefore to accommodate this, the compost feedstock should be as homogenous and as consistent as possible. Homogeneity is critical when supplying, for example nurseries as variations from pot to pot can have significant production implications.

The beneficial effects of compost use can help growers to save money and reduce their reliance on pesticides, subsequently conserving natural resources. Disease suppressive soils are a well known phenomenon, especially in organic agriculture. Suppressiveness in soils is related to changes in microbial populations which the addition of compost enhances. There is also evidence to suggest that the application of compost can help to control numbers of parasitic nematodes by providing nutrients to the soil that encourage the growth of fungi and bacteria which compete with, or destroy the nematodes.

Overall disease suppressive properties can add significant value to compost and have the potential to stimulate the needed agricultural and horticultural markets for compost. The addition of compost has also been shown to have a remedial effect on soils that have had over applications of pesticides. Organic matter can effectively bind pesticides therefore reducing their concentrations in the soil. In several studies, the addition of compost removed the toxic effect of soil applied pesticides.

There is a growing body of evidence to show that composted materials can suppress a number of soil borne diseases, with obvious benefits for reducing the reliance on conventional fungicides.<sup>90</sup> In addition, several studies have shown soil microbial activity to be higher and disease incidence lower, after the application of compost. **Error! Reference source not found.** gives a summary of specific work carried out on specific plant pathogens. In these studies, various degrees of control were achieved by using compost.

In the United States the nursery industry used disease suppressive compost widely and as a consequence significantly reduced the need to use fungicide drenches, in some instances replacing the need to use methyl bromide completely (used primarily as a soil fumigant). This is particularly relevant as methyl bromide is implicated in ozone depletion (and consequently the use of methyl bromide has been phased out).

Communications with Professor Hoitink of Ohio State University on savings related to suppressive power in pot cultivation suggest that the reduction in the use of fungicides resulting from use of a potting mix suppressive to Pythium and Rhizoctonia diseases (i.e. including composted materials) leads to savings of approximately \$20 in fungicide use per drench per cubic metre of potting mix placed into 1,200 ml pots for the ornamentals

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<sup>90</sup> For a review, see H. A. J. Hoitink and M. J. Boehm (1999). Biocontrol within the context of soil microbial communities: a substrate-dependant phenomenon. *Annual Review of Phytopathology*, **37**, 427-436.

industry. Apparently, this is usually of the order 50% of the purchase cost of the potting mix in the United States.

As with the agricultural applications, it is difficult to quantify the pesticide use savings and their relationship to compost use. We assume here that the savings are the same as made in the agricultural sector.

**Table 27 Research into Control of Plant Pathogens by the Addition of Compost**

Pathogen	Authors reporting control	Compost used and rate applied
<i>Fusarium oxysporum</i>	Hoitink & Fahy (1985)	30t/ha composted larch bark
	Widmer et al (1996)	20% by volume composted municipal waste
<i>Phytophthora capsici</i>	Kim et al (1997)	0.2% w/v chitosan
<i>Phytophthora cinnamoni</i>	Hoitink & Fahy (1985)	10% by weight composted municipal sludge
<i>Phytophthora nicotianae</i>	Widmer et al (1996)	20% by volume composted municipal waste
<i>Pythium ultimum</i>	Widmer et al (1996),	20% by volume composted municipal waste
	Ringer et al (1997)	30% by volume composted dairy manure and leaves
<i>Rhizoctonia solani</i>	Hoitink & Fahy (1985)	10% by weight composted municipal sludge
<i>Sclerotinia homoeocarpa</i>	Hoitink & Fahy (1985)	7-10t/ha composted municipal sludge
<i>Venturia inaequalis</i>	Yohalem et al (1994)	2:1 spent mushroom substrate: potting compost

Sources: Hoitink, H.A.J. and Fahy, P.C. (1986) *Basis for the Control of Soil Borne Plant Pathogens with Composts*, *Annual Review of Phytopathology* **24**, 93. Ringer, C. E., Millner, P.D., Teerlinck, L.M. and Lyman, B.W. (1997). *Suppression of Seedling Damping-off Disease in Potting Mix Containing Animal Manure Composts*. *Compost Science & Utilization*, **5**, 6-14. Yohalem, D. S., R. F. Harris, and J. H. Andrews (1994). *Aqueous Extracts of Spent Mushroom Substrate for Foliar Disease Control*. *Compost Science & Utilization*, **2**, 67-74.

## A.6.6 External Costs of Pesticides

A number of attempts have been made to estimate in monetary terms the environmental costs of using pesticides.<sup>91</sup> One study estimated the total external costs of pesticide use in US agriculture at just under \$8 billion against private expenditure on pesticides of some \$4 billion.<sup>92</sup> Of the \$8 billion external costs, the authors recognise that \$3 billion is paid,

<sup>91</sup> These include D. Pimentel, C. Harvey, P. Resosudarmo, K. Sinclair, D. Kurz, M. McNair, S. Crist, L. Shpritz, L. Fitton, R. Saffouri and R. Blair (1995) *Environmental and Economic Costs of Soil Erosion and Conservation Benefits*, *Science* **267** (5201): pp.1117-1123.; and R. Steiner, L. McLaughlin, P. Faeth and R. Janke (1995), *Incorporating Externality Costs into productivity Measures: A Case Study Using US Agriculture*, in V. Barbett, R. Payne and R. Steiner (eds) (1995) *Agricultural Sustainability: Environmental and Statistical Considerations*, New York John Wiley and Sons.. A study by Fleischer and Waibel has also undertaken a similar exercise in the German context (see Barbara Dinham (1998), *The Costs of Pesticides*, *Pesticides News*, No.39, March 1998, p.4.), whilst another study (Paul C. James (1995), *Internalising Externalities: Granular Carbofuran Used in Rapeseed in Canada*, *Ecological Economics*, **13** (3), pp.181-4.) has sought to establish what one might term a 'break-even kill rate' for birds above which the net economic gain from using granular carbofuran on rapeseed in Canada turns negative (this kill rate is marginally above the median of several estimates obtained).

<sup>92</sup> D. Pimentel, H. Acquay, M. Biltonen, P. Rice, M. Silva, J. Nelson, V. Lipner, S. Giordani, A. Horowitz and M. D'Amore (1993), *Assessment of Environmental and Economic Impacts of Pesticide Use.*, in D. Pimentel and H. Lehmann (eds) (1993), *The Pesticide Question: Environment, Economics and Ethics*, London: Chapman and Hall, pp.47-84

effectively, by farmers (pollination losses etc.) leaving a \$5 billion bill to be absorbed by society.<sup>93</sup> They estimate benefits associated with reduced crop losses from using pesticides of some \$16 billion.

In another US study, the external costs are estimated at the much lower level of between \$1.3 and \$3.6 billion. This results from omitting the estimates for pesticide resistance and crop losses. It should be noted, however, that the range of estimates associated with loss of biodiversity range from \$0.3 to \$20 billion.<sup>94</sup>

A study carried out in Germany against the backdrop of the 1986 Pesticides Act estimated that farmers spend, in total, some DM 1.69 billion per year, generating an estimated net return of DM 1.15 billion. The external costs were estimated at between DM 0.25 and 0.31 billion.<sup>95</sup>

Taken together, these three studies suggest that the external costs associated with pesticide use are not trivial. They lie between 25% and 125% of the total private costs of pesticide use.

One review of external costs from pesticide use also attempted to assess the external costs of health damage through estimates of the value of a statistical life.<sup>96</sup> Globally, this study suggests a figure of \$4 billion, apparently on the basis of WHO estimates of 40,000 unintentional deaths per year resulting from pesticides. They also point out that biodiversity losses are potentially the least well understood of the external costs. Not untypically, at the higher end of the range, they can swamp the estimated total external costs of pesticides.<sup>97</sup>

A more recent study by Pretty et al, which the authors believe give estimates of pesticide externalities on a conservative basis, estimates these to be (in the UK situation) of the order £8.6 per kg active ingredient used.<sup>98</sup> Another study used a contingent valuation survey to estimate an externality per kg of active ingredient of £12 per kg on average.<sup>99</sup> The study went on to look at pesticide ranking methodologies to investigate the potential for tax differentiation according to the environmental impact of specific products (as now happens in Norway and has been proposed in the UK).

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<sup>93</sup> Loss of natural enemies and the build up of resistance might not be considered as externalities in the true sense. This is to some extent recognised by the authors when they conclude that of the \$8 billion 'external costs', \$3 billion is already being paid for by farmers (i.e. to the extent that farmers are aware of these, the costs are already internalised).

<sup>94</sup> R. Steiner, L. McLaughlin, P. Faeth and R. Janke (1995), Incorporating Externality Costs into productivity Measures: A Case Study Using US Agriculture, in V. Barbett, R Payne and R. Steiner (eds) (1995) *Agricultural Sustainability: Environmental and Statistical Considerations*, New York John Wiley and Sons.

<sup>95</sup> See Barbara Dinham (1998), The Costs of Pesticides, *Pesticides News*, No.39, March 1998, p.4.).

<sup>96</sup> David Pearce and Robert Tinch (1998), The True Price of Pesticides, in William Vorley and Dennis Keeney (eds) (1998), *Bugs in the System: Redesigning the Pesticide Industry for Sustainable Agriculture*, London: Earthscan, pp.50-93.

<sup>97</sup> Professor Waibel has commented that in the German study, the estimates for biodiversity losses are the 'lower limit', these including losses of plant species due to herbicide use only (personal comm.).

<sup>98</sup> J. N. Pretty, C. Brett, D. Gee, R. E. Hine, C. F. Mason, J. I. L. Morrison, H. Raven, M. D. Rayment and G. van der Bijl (2000) An Assessment of the Total External Costs of UK Agriculture, *Agricultural Systems* (65) pp.113-136.

<sup>99</sup> V. Foster, S. Mourato, R. Tinch, E. Ozdemiroglu and D. Pearce(1998). Incorporating External Impacts in Pest Management Choices, in Vorley, W. and D. Keeney, (eds.) *Bugs in the System- Redesigning the Pesticide Industry for Sustainable Agriculture*, London: Earthscan Publications Ltd.

### A.6.6.1 Assumptions in this Study

In this study, we assume that where compost is applied at 10 tonnes dry matter per hectare, the use of pesticides falls by 20%. This may be a conservative assumption. A US study gives an example of a farm where 50% reductions in synthetic pesticides and fertilisers were achieved.<sup>100</sup> Work undertaken in Europe in this area assumed that a 50% reduction was possible as composting was integrated into conventional, low-input and organic systems.<sup>101</sup> In both studies, the reductions occur as a consequence of more widespread changes in farm practice, of which applying compost is only one. Hence, it is difficult to attribute such large reductions to the act of applying compost alone. On the other hand, there is a point to be made (from the wider perspective of agricultural policy) that the use of composted materials may well have a role to play in pesticide reduction strategies. The sorts of reduction in use achievable may be comparable with those attained through instruments such as pesticide taxes (although the effect of these is difficult to discern).

From the two studies considered above that cite externalities per kilogram of active ingredient, we assume that external costs of pesticide use lie between £8 and £12 per kg of active ingredient used. Figures from the Pesticide Usage Survey indicate that the UK used around 31 million kg of active ingredient in 2004.<sup>102</sup> This is applied over a land area of around 5 million hectares, an average of just over 6kg active ingredient per hectare.

Assuming, therefore, that pesticide use could be reduced by 20% (and clearly this would vary according to the country concerned, the crops being grown and, therefore, the pesticides currently in use) through annual applications of 10 tonnes dry matter per hectare, we have calculated high and low levels of the external benefits that might be derived from compost use through the avoided external costs of pesticides which might otherwise be used. Note that to the extent that the 20% figure is considered as reasonable, we may be underestimating the reduction in externalities since the application of pesticides per unit of land will be heavily weighted towards arable agriculture which is a relatively low intensity user of pesticides. Compost applications may be weighted towards sectors of agriculture which make greater use of pesticides, such as horticulture and fruit and vegetable growing.

Clearly the assumption concerning the percentage reduction achieved due to the application of compost is the subject of considerable uncertainty. Not only is the percentage itself subject to some doubt, but most likely, it would vary across countries in accordance with the distance that different countries have already moved in this direction. The level of pesticide reduction achieved is likely to be predicated quite strongly on the concomitant adoption of low-external input methods of cultivation. Also subject to uncertainty are the external costs of pesticide use which one is assuming are being avoided. Pesticides are an extremely heterogeneous group of compounds with differing impacts on different media. A more sophisticated analysis could try to elicit avoided external costs more accurately. However, such efforts would be fraught with difficulty because the impacts associated with pesticide use are location specific. Hence, the estimates above are subject to considerable uncertainty. However, they represent an attempt to capture the potential external benefits associated with reducing pesticide use

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<sup>100</sup> Edward C. Jaenicke (1998) *From the Ground Up: Exploring Soil Quality's Contribution to Environmental Health*, mimeo from the University of Tennessee, October 1998.

<sup>101</sup> A. Meier-Ploeger, H. Vogtmann and H Zehrm (1996) *Eco Balance Compost Versus NPK Fertiliser*, INAC for ORCA, Brussels: ORCA.

<sup>102</sup> Pesticide Usage Statistics <http://pusstats.csl.gov.uk/mygraphresults.cfm>.

through the application of compost. Of course, to the extent that other soil management practices change, these will have implications for pesticide use also.

It should be noted that the external benefits from the displacement, whilst they are uncertain, are based on estimates of pesticide externalities that are generally thought to be conservative. No account has been taken of the avoided costs of the production of pesticides which might otherwise have been used. This is the exact opposite of the situation in respect of fertilisers where the attempt was made to capture the external benefits from avoiding production, but the external costs of, for example, nitrate leaching, have not been estimated.

### A.6.7 Greenhouse Gas Emissions from Compost After Application to Soil

The use of compost in agriculture can have a positive effect on soil carbon levels and subsequently act as a carbon reservoir. In this study, instead of assuming some 'reference point' from which it is assumed certain processes are 'net sequesters of carbon', all sources and sinks are treated equally. Effectively, the reference point shifts from one of whether something does or does not act as a 'net sequester of carbon', but instead, to one of how much carbon is emitted over time.

Compost does not result in the permanent and irreversible locking up of all carbon in compost. What compost can do is reverse the decline in soil organic matter which has occurred in relatively recent decades through contributing to the stable organic fraction in soils (effectively locking-up carbon). Historically agricultural practices have probably been responsible for much of the increase in atmospheric carbon dioxide. It is also important to realise that whilst the debate concerning 'sequestration' has emerged as a topical one in the wake of the debate on climate change, the role played by soil organic carbon is far more complex, and potentially far more important, than the single role played in terms of carbon sequestration.

It is clear that the effects of soil organic matter on soil biota are at the heart of the disease suppressing effects of compost. The interrelationship between carbon and nitrogen largely determines the magnitude of soil microbial populations. Utilisation of carbon and nitrogen by microbes is also responsible for the turnover between organic and mineral forms of nitrogen. Hence, the biomass production potential of soil is largely dependent upon the ability of a soil to support microbes such as bacteria and fungi.

Three pools of organic carbon are available for microbial utilisation:

1. The active soil fraction (turnover of around two years, and representing short-term sequestration of carbon – provides source of energy for microbes, and soil carbon and nitrogen supply necessary components for amino acid synthesis);
2. The slow or decomposable soil fraction (turnover time two to three years – of great importance to developing good soil structure – disturbed by cultivation and other disturbances – provides a source of carbon for biological digestion by microbes, so linking to the active pool. – can be viewed as mature compost); and
3. The passive soil organic fraction. (turnover time of the order 1,000 years - resistant to oxidation processes – acts as a 'cement' that binds particles).

Only the first two of these pools contain carbon in readily available forms for microbial utilisation. The last pool contains carbon in a highly stable form. Some microbes can utilise

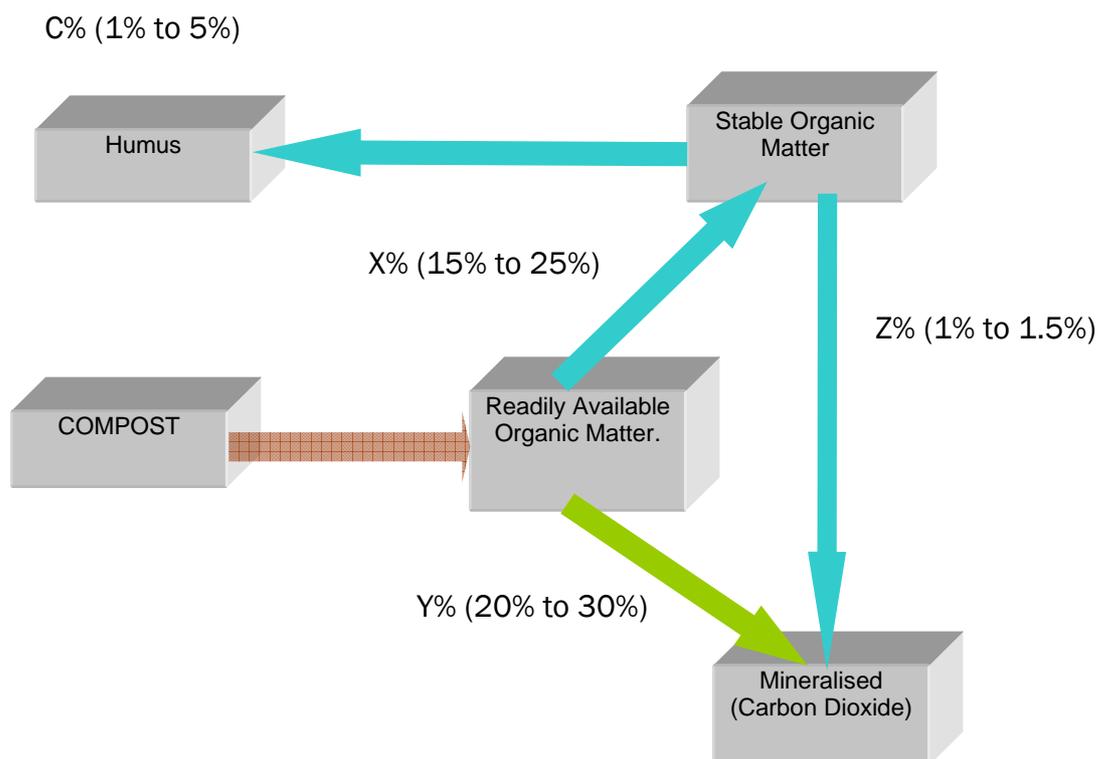
this pool so depletion does occur. It can also be replenished from active and slowly decomposable fractions. It is the fact that this passive pool of carbon can be maintained or increased that leads to the idea that the passive pool can act to ‘sequester’ carbon in the soil. Clearly, this long turnover time does appear to imply that, for all intents and purposes (certainly for any economic analysis deploying a non-zero discount rate), this carbon is not released into the atmosphere.

We have sought to model the dynamics of soil organic carbon where it is applied in composted form. The pathways modelled are outlined in below.

The application of compost is assumed to lead to the readily available carbon being mineralised at  $y\%$  whilst  $x\%$  of the readily available organic carbon is converted to stable organic matter. Of this stable organic matter, some carbon is mineralised, but at a much lower rate than that at which the readily available matter is converted to stable organic matter.

Consequently, application of organic matter to soil can act to increase soil organic carbon levels (though as we shall see, the degree to which this occurs varies according to the choice of the different parameters chosen, the rate of application of compost and the baseline level of organic matter in the soil).

Figure 12: Basic Description Of Modelling Of Fate Of Carbon In Compost / Soil

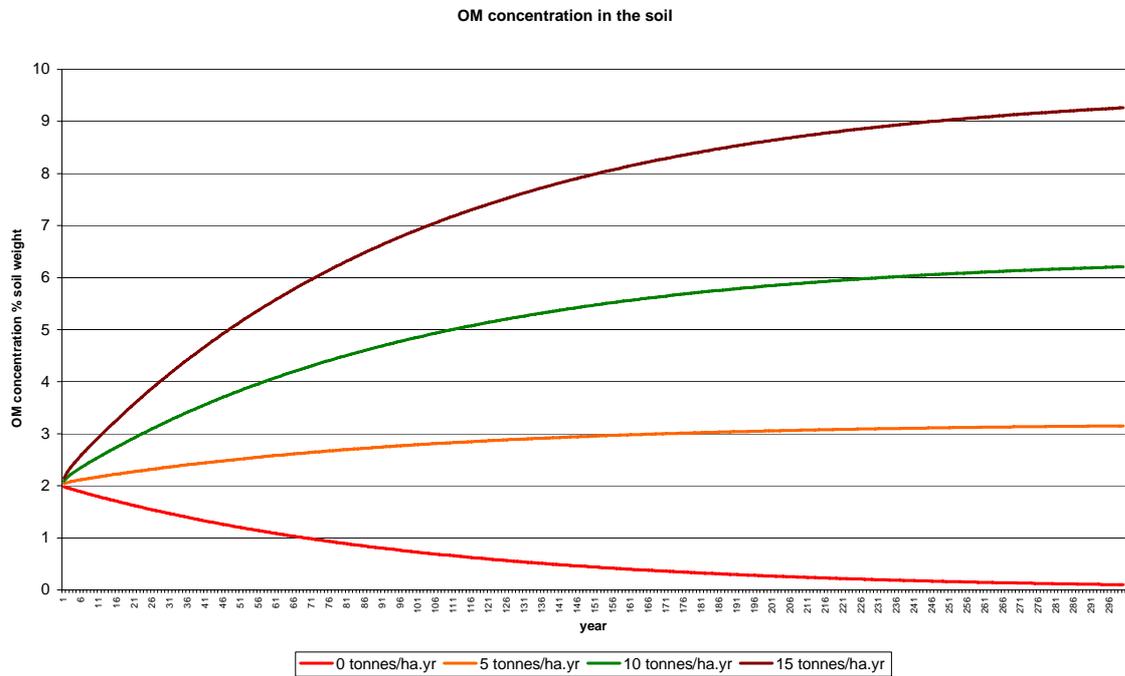


Using the following figures:

$X = 25\%$ ,  $Y = 20\%$ ,  $Z = 1\%$ , and with an initial organic matter concentration of 2%, one can understand the effects of different rates of compost application. This is shown in **Error! Reference source not found.** The production of compost and incorporation in topsoil has the potential to act as a significant reservoir for carbon. When combined with responsible

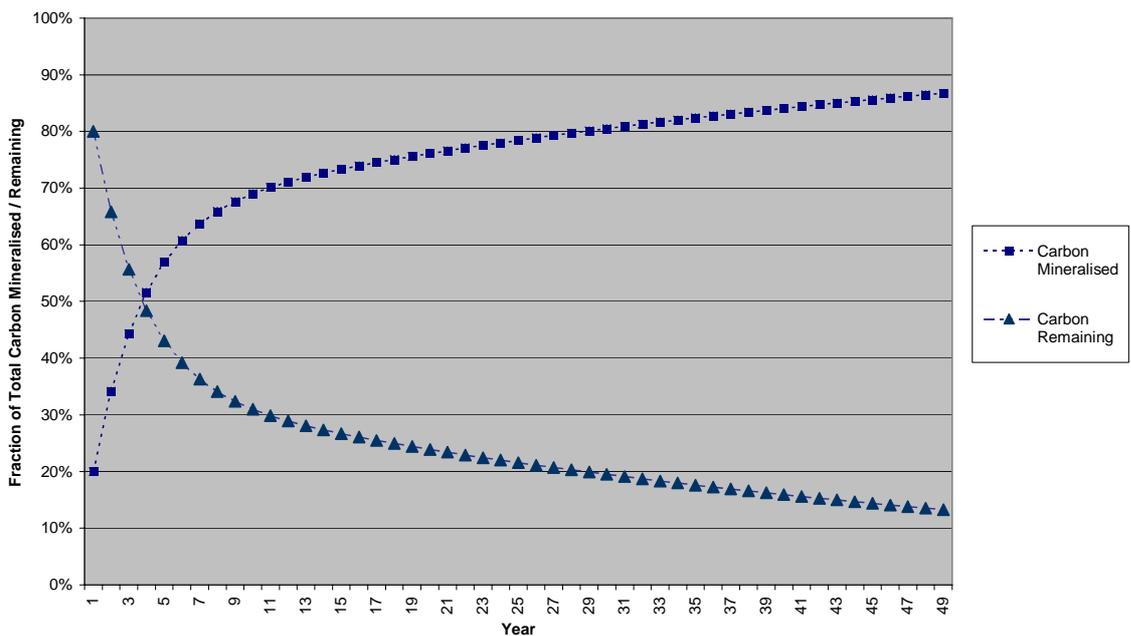
agricultural practices it could have a positive impact on reducing the rate of global warming.

Figure 13: Effect Of Different Rates Of Compost Application On Soil Organic Matter Levels



However, after compost is applied, the processes of mineralisation do lead to releases of carbon dioxide. The pattern of the release through mineralisation processes is shown in **Error! Reference source not found.** for a 50 year period (after which some 13% of the original soil organic carbon remains in the soil).

Figure 14: Carbon Remaining in Soil / Mineralised Over Time



We have used the following figures in the analysis: X = 25%, Y = 20%, Z = 1%. These figures are estimates only. Relatively little is known about the different rates of transformation, and there is some evidence that the rates may be endogenous with respect to the prevailing level of soil organic matter in the soil.<sup>103</sup> These figures generate profiles for carbon dioxide emissions. The external costs from these emissions, discounted over time, are then assessed and included in the modelling.

## A.6.8 Compost as an Alternative to Peat

The area of peatlands represents approximately 5%-8% of the world's surface. As peat formation is linked to climate, the majority of the world's resources lie in northern temperate zones. Intense industrialisation has resulted in the steady destruction of Europe's peatlands, indeed in some European countries such as the Netherlands, and Poland there are no natural peatlands or bogs remaining.

As extraction of peat in western European countries begins to decline, extraction in eastern European countries is increasing. In 1998 a 400% increase in the amount of peat exported to the UK from Estonia was reported and there has also been an increase in peat from other Baltic states, such as Lithuania.

In the following sections the primary uses of peat and the environmental impacts of peat extraction are briefly reviewed.

### A.6.8.1 Primary Uses of Peat

The main uses of peat are as a fuel (predominantly in Ireland, Finland and Baltic states) and in professional and amateur horticulture as a growing media (much more widely – there is a thriving export market). Due to the increase in gardening as a leisure time activity, the demand for peat has increased dramatically. In the UK alone, use of peat by amateur gardeners accounts for 70% of all the peat used, the remainder being used in professional horticulture and landscaping.

The main use of peat in amateur gardening is as a soil improver. Compost can be used as a substitute for peat as a soil improver. It performs well and is likely to be accepted by the public as an alternative to peat. However there are several barriers to overcome before this is the case.

Retailers and consumer must be assured that compost is of an acceptable quality (fit for purpose) and that there is uniformity between batches. Although, many consumers are aware of the environmental implications of peat use, many are unaware that products purchased contain peat. The labelling of products need to be clearer so that potential consumers can make an informed choice related to environmental factors when purchasing. Critically, compost and alternatives to peat must be priced competitively, as price is the predominant factor influencing purchases for most consumers.

The other main use of peat is as the main constituent in growing media. Peat as opposed to compost is the preferred growing media as it has no nutritional content and has superior water holding capacity. Some research has been carried out into the use of compost as a constituent of growing media and successful results have been achieved. However, growers are reticent to use compost growing media alternatives as it is believed that they

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<sup>103</sup> Personal comm.. F. Tittarelli.

do not perform as well as peat. There is also the issue of quality, especially with regard to the presence of weed seeds and plant pathogens. Growers need to be assured that compost is free from contaminants and that it will not have any phyto-toxic effects. These assurances need to be given based on substantiated research and adequate quality and user guidelines.

Currently, it is unlikely that compost will be accepted as a blanket alternative to peat in growing media. However, there is considerable scope for blending compost with peat. In the UK alone, if 10% of all peat usage were to be replaced with an alternative, peat use would fall to 3.1 million m<sup>3</sup> a year as opposed to 3.4 million m<sup>3</sup>.<sup>104</sup> As consumer confidence increases in the peat/compost mix, so could the proportion of compost in the blend, subject to the blend being fit for purpose.

### A.6.8.2 Environmental Costs of Peat Extraction

The environmental costs of peat extraction are difficult to quantify in monetary terms. The primary losses to the environment through peat extraction are:

- Loss of biodiversity;
- Loss of landscape and recreational value;
- Loss of palaeoecological and archaeological value; and
- Increased carbon emissions and loss of carbon reservoirs.

### A.6.8.3 Loss of Biodiversity

Many rare and protected species thrive in Europe's peatlands and bogs. The bog moss *Sphagnum imbricatum* is entirely restricted to bogs and is the principal peat forming species in oceanic peatlands. It is becoming increasingly rare as more sites are being developed. There is also the loss of rare and unique plants which have potential medicinal properties. These benefits are difficult to value although studies concerning biodiversity loss have reported high values reflecting the willingness of citizens to pay for conservation.

### A.6.8.4 Loss of Landscape and Recreational Value

In Europe areas of peatlands and bogs have a cultural importance as some of the last true remaining wilderness areas. They attract visitors for this reason. Travel-cost and contingent valuation studies capture consumer surplus associated with, and preferences for, respectively, the continued existence of these landscapes. One study translates values for the Somerset Levels into a value of £7,245 per hectare.<sup>105</sup> Another study estimated a preservation value of £68.4 million, or £4.1 million per annum using a 6% discount rate.<sup>106</sup>

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<sup>104</sup> ODPM (2002) *Monitoring of Peat and Alternative Products for Growing Media and Soil Improvers in the UK*, 2002.

<sup>105</sup> K. Willis, G. Garrod and C. Saunders (1993) *Valuation of the South Downs and Somerset Levels and Moors Environmentally Sensitive Landscapes by the General Public*, Report to MAFF, Newcastle-upon-Tyne: Centre for Rural Economy. This is the interpretation of the original study from Alan Ingham (1996) *The Use of Economic Instruments to Protect Raised Lowland Peatbogs*, Report to Royal Society for Protection of Birds, Department of Economics, University of Southampton..

<sup>106</sup> N. Hanley and S. Craig (1991) *The Economic Value of Wilderness Areas*, in F. Dietz, F. Van der Ploeg and J. van Straaten (1991) *Environmental Policy and the Economy*, Amsterdam: North Holland.

Other valuation studies looking at wetland areas are shown in **Error! Reference source not found..**

Table 28 Valuation Studies On Wetlands

Study	Location / technique	Effect valued	Euro (2000)	Data needed for aggregation
Brower et al (1997)	30 studies from the USA, UK, other European countries meta-analysis	mean WTP including indirect use and non-use values: <ul style="list-style-type: none"> <li>• average for all types of wetlands</li> <li>• average for flood control</li> <li>• average for biodiversity</li> <li>• average for USA</li> <li>• average for the UK</li> <li>• average for the rest of Europe</li> </ul>	42.9 / household / year 66.0 / household / year 54.5 / household / year 70.2 / household / year 25.2 / household / year 22.0 / household / year	Total loss of a wetland would mean total loss of this value. No estimate for the relationship of percentage lost and WTP changes. Assess the threshold level of loss. Assess the population affected.
Bateman et al (1992) and (1997)	UK, Norfolk Broads: CVM	average WTP to preserve present landscape: <ul style="list-style-type: none"> <li>• use values</li> <li>• non-use values of local population</li> <li>• non-use values of the rest of the UK</li> </ul>	117 - 158 / household / year 22 / household / year 7.3 / household / year	no. of visitors no. of local population (non-visiting) no. of the rest of the UK (non-local non-visiting)
Bateman et al (1995)	UK, Norfolk Broads: CVM	WTP to preserve Broadland from the effects of increased flooding	36.7 / holiday visit / year 33.8 / day trip / year 16.2 / non user / year	no. of staying visitors no. of day trippers no. of non users in UK
Kosz (1996)	Austria, Donau-Auen riverside wetlands: CVM	mean WTP to preserve the wetlands	29.3 / person / year	no. of people affected
Willis (1990)	UK, Derwent Ings, Yorkshire:	WTP for the preservation of the current state of the wetlands: <ul style="list-style-type: none"> <li>• total use value</li> </ul>	66 / ha	the estimates are additive. No adjustment factor is available. no. of ha affected

Study	Location / technique	Effect valued	Euro (2000)	Data needed for aggregation
	CVM	<ul style="list-style-type: none"> <li>total non-use value</li> </ul>	1223 / ha	no. of ha affected.
Hanemann et al (1991)	USA, San Joachin Valley, California	WTP for <ul style="list-style-type: none"> <li>maintenance</li> <li>improvement</li> </ul>	188.6 / household / year 310.2 / household / year	choose which change applies. no. of affected households no. of affected households
Whitehead (1990)	USA, Clear Creek, Kentucky	aggregate benefits	189 - 126.8 / ha	no. of ha affected
Stone (1992)	Australia, Barmah wetlands	mean annual WTP for wetlands protection	127.8 - 164.6 / ha	no. of ha affected
Tapsell et al (1992)	UK	WTP for recreational values <ul style="list-style-type: none"> <li>present condition</li> <li>some improvement towards natural conditions</li> <li>recovery to full river condition</li> </ul>	2.8 / user / visit 2.2 / resident / visit 4.0 / user / visit 3.4 / resident / visit 5.0 / user / visit 4.8 / resident / visit	<b>METHOD NOT RECOMMENDED</b> choose which change applies no. of visits by non-residents no. of visits by residents

### A.6.8.5 Greenhouse Gas Emissions (Loss of Carbon Reservoirs)

The greenhouse gas emissions associated with peatbogs are extremely complex, and they change once the process of development (for extraction) occurs. In northern peatlands, the anaerobic conditions and cold temperature result in increased sequestration of carbon (relative to other wetlands).<sup>107</sup> Unperturbed peatbogs, whilst they may act as a sink for carbon, may also emit methane. However, as long as they are unperturbed, they most likely retain a balance between methane emissions and carbon sequestration.

Drainage and degradation of peatlands increases carbon dioxide emissions. It also increases nitrous oxide emissions significantly.<sup>108</sup> It has been estimated that peatlands contain between 329 and 528 billion tonnes of carbon (equivalent to 1,200-1,900 billion tonnes of carbon dioxide). Unless the bogs are disturbed by extraction, drainage or other human intervention, much of the carbon will remain in-situ for near geological timescales.

Drainage of peatlands and other wetlands acting as carbon reservoirs will result in the oxidation of the organic matter, releasing it to the atmosphere as carbon dioxide, methane and other greenhouse gases. Conversely, restoration or creation of new wetlands may provide additional carbon sinks.<sup>109</sup>

### A.6.8.6 Loss of Palaeoecological and Archaeological Value

Peatlands and bogs contain a rich archive of information about our history. Examination of peatlands provides an insight into past climates, culture and economy. These non-use benefits of peat are lost once they are developed for exploitation.

### A.6.8.7 Summary

The use of peat would seem to incur considerable environmental costs. Although these can be captured in various ways, it is very difficult to impute an environmental cost per tonne of peat extracted.

The development of peat bogs is a more or less irreversible process. The environmental impacts associated with peat extraction therefore exhibit strong discontinuities at low levels of extraction. For this reason, the concept of

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<sup>107</sup> Wetlands store carbon in short- and long-term reservoirs. Storage occurs when primary production is high and exceeds the rate of decomposition, or when the rate of decomposition is slowed by a process known as anoxia, and cold temperatures (leading to accumulation of undecomposed organic matter).

<sup>108</sup> N. Roulet, R. Ash, W. Quinton and T. Moore (1993) Methane Flux from Drained Northern Peatlands: Effect of a Persistent Water Table Lowering on Flux, *Global Biogeochemical Cycles* 7 (4) pp.749-69; K. Regina, H. Nykaken, M. Maljanen, J. Silvola and P. Martikainen (1998) Emissions of N<sub>2</sub>O and NO and Net Nitrogen Mineralization in a Boreal Forested Peatland Treated With Different Nitrogen Compounds, *Canadian Journal of Forest Research*, 28, pp. 132-40; C. Freeman, M. A. Lock and B. Reynolds (1992) Fluxes of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O from a Welsh Peatland Following Simulation of Water table Draw-down: Potential Feedback to Climate Change, *Biogeochemistry*, Vol.19 No.1, pp.51-60.

<sup>109</sup> See, for example, Environment Canada (1998) *Wetland Conservation: An Excerpt from the Sinks Table Options Paper*, November 1998.

environmental damages associated with marginal extraction of peat is an awkward one. Once extraction occurs, restoration is difficult, if not impossible. Hence, the Irish Country Implementation study in the ExternE programme sought some justification for omitting the effects of peatland preparation through appeal to the fact that the power station examined was using peat from already drained bogs.<sup>110</sup>

Alternatively, as with disamenity associated with landfill and incineration, one tries to impute average external costs of development and extraction of peat. This is not straightforward. Emissions and absorption of different greenhouse gases occurs at different rates at different stages of peat bog development.

We have taken the data from the Finnish life cycle study used in the National Implementation study.<sup>111</sup> We have adapted this since that study, which includes emissions from various aspects of the peat fuel life cycle, includes emissions from power production and from restoration. The former are subtracted from the data for obvious reasons. The latter are subtracted because at non-zero discount rates, the effects of the restoration ‘savings’ are likely to be relatively small. The emissions per tonne of peat extracted are shown in **Error! Reference source not found.** below.

**Table 29 Gaseous Emissions From Peat Draining And Extraction (tonnes per tonne peat extracted)**

Pollutant	SO <sub>2</sub>	NO <sub>x</sub>	TSP	CO <sub>2</sub>	CH <sub>4</sub>	N <sub>2</sub> O
t/t peat	5.76E-05	0.0002	2.88E-05	0.1844	-0.0006	6.92E-05

These gaseous emissions are used as the basis for the external cost savings from compost use where it displaces peat. Peat is replaced by compost more on the basis of volume than on weight. The density of peat is low, estimated here at 200kg/m<sup>3</sup>. The density of compost, on the other hand, is of the order of 500kg/m<sup>3</sup> for compost with dry matter content 60%. This implies that to replace one tonne of peat would require compost resulting from 7.14 tonnes of waste material.

The external cost savings would appear to be a significant under-estimate of the avoided external costs to the extent that pressure to develop new peat bogs is reduced. This is due to the fact that the non-use values of peat bogs appear to be significant.

### A.6.9 Uses of Compost and Market Shares

In order to understand external benefits (through avoided burdens and changes in environmental quality) associated with using compost, it is necessary to understand where compost is likely to be used and for what purpose. In addition, one would seek to understand which products one is displacing in specific markets where compost is being used. This is not a straightforward task (so that the individual benefit categories hypothesised above can be attributed in appropriate ratios). A wide range of soil improvers and conditioners now exist, some synthetic and some natural, and each

<sup>110</sup> Chapter 12, Ireland, in European Commission (1999a) *ExternE: Externalities of Energy, Vol.10 – National Implementation*, Luxembourg: Office for Official Publications of the European Communities.

<sup>111</sup> Chapter 9, Finland, in European Commission (1999a) *ExternE: Externalities of Energy, Vol.10 – National Implementation*, Luxembourg: Office for Official Publications of the European Communities.

with its own specific characteristics that make it more or less suitable for application in a specific context.

The UK composting market is currently dominated by the composting of garden wastes. The Composting Association reports that in 2003/4, 1.97 Mt of waste was composted at 325 facilities. 73% of this was household waste, and 4% was non-household municipal waste. During 2003/04, garden waste accounted for 95% of municipal wastes composted, and virtually all household wastes composted. The Association notes:

*Despite considerable and sustained growth in particular through the on-farm composting sector, only approximately one-fifth of the estimated 7 Mt of household garden waste arisings in the UK in 2003/04 were composted by the industry, whilst the estimated 6 Mt of kitchen wastes remained a largely untapped resource.*

The survey recorded the manufacture of approximately 1.2 Mt of composted products, of which, the largest fraction was soil conditioner (61 %), followed by mulches (16 %). Other fractions including growing media constituent and ingredients in manufactured topsoil, whilst turf dressings accounted for the remainder.

Compost products were distributed to several markets and outlets. Agriculture was both the largest and the fastest growing outlet. Around 40 % of composted product went to agriculture in 2003/04 reflecting the increase in on-farm composting facilities highlighted by the report. 36 % was utilised in markets with a sales value including horticulture, landscaping and domestic gardening.

Quality assurance of composted products continues to grow as an important element of compost production with over 850,000 tonnes of material complying with at least one independently certified standard. The utilisation of composted products in landfill engineering fell by 18% from 2001/2 and when combined with product used in land restoration combined accounted for the remaining 24 %.

We have assumed that the material is used in agriculture. It would be possible to alter this assumption and assume displacement of peat, or to assume ratios thereof. The study is, however, hampered by the fact that the nature of product value and product displacement are simply less well understood in other markets than in agriculture.

## A.6.10 Water Use

Water availability has not, traditionally, been an issue of great concern to UK agricultural producers. However, climate change may be changing this situation. Research by Cranfield University suggests that demand for irrigation water is increasing:<sup>112</sup>

*Despite 2001 being a relatively “wet” year, the area irrigated was almost as high as in the dry year of 1995, when the previous survey was undertaken. Analysis suggests that demand for water for a dry year is growing at 2 to 3% per annum*

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<sup>112</sup> E. K Weatherhead and K. Danert (2002) *Survey Of Irrigation Of Outdoor Crops in 2001 – England*, Research for Defra’s Climate Change and Demand for Water (CCDeW) project, October 2002.

This is reflected in the figures in **Error! Reference source not found..**

**Table 30: Volumes of Water Applied ('000m<sup>3</sup>) by source, 1982-2001**

	1982	1984	1987	1990	1992	1995	2001
Surface water	34,390	57,210	19,250	74,070	41,820	90,860	75,760
Ground water	16,680	32,420	11,800	50,540	28,470	61,620	47,810
Public mains	2,040	3,840	1,100	3,860	2,620	4,390	4,300
Rain collected	included in other						2,050
Re-used water	included in other						670
Other	1,830	3,540	1,470	5,330	2,160	4,880	710
Total	54,940	97,730	33,630	133,790	75,070	146,960	131,300

Source: E. K Weatherhead and K. Danert (2002) Survey Of Irrigation Of Outdoor Crops in 2001 – England, Research for Defra’s Climate Change and Demand for Water (CCDeW) project, October 2002.

Notes:

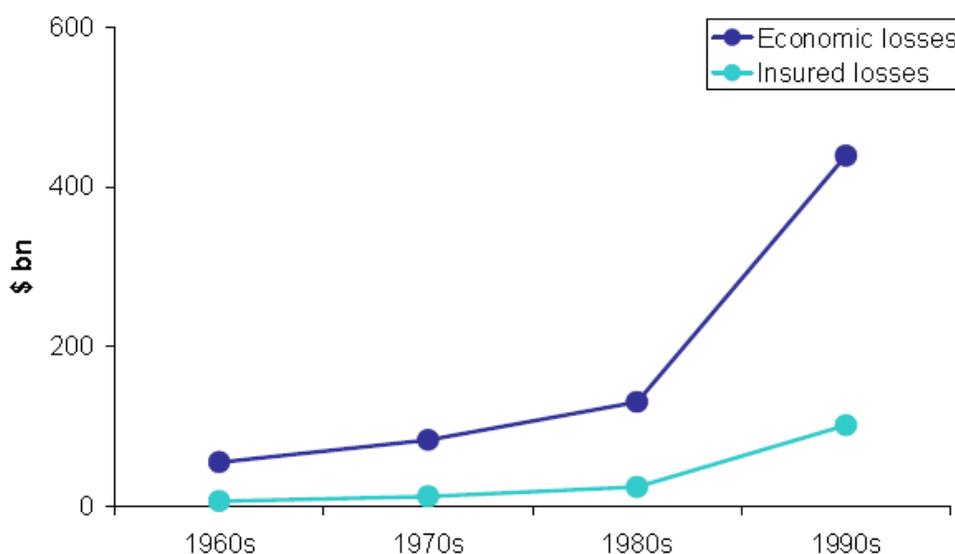
Surface water includes ponds, lakes, gravel or clay workings, rivers, streams or other water courses.

Ground water includes wells, bore holes and springs rising on the holding.

Data up to 1992 for England and Wales, data for 1995 and 2001 for England only

At the same time, weather-related insurance damages are increasing. This is shown in **Error! Reference source not found..**

**Figure 15: Costs of Weather-related Disasters (total economic losses and insured losses, \$ billion) over Last Four Decades, Adjusted to 2003 Prices.**



Source: Annual review of natural catastrophes, Munich Re, 2004.

In the UK, the risks of flood damage appear to be increasing in some areas. A report produced by the then Ministry of Agriculture, Fisheries and Food in 2001 estimated that approximately 10% of the population of England, in around 1.8 million homes and 130,000 commercial properties, are within areas at risk from river and coastal

flooding.<sup>113</sup> Together with agricultural land that is at risk, the value of the assets 'at risk' was estimated at around £220 billion.

The Association of British Insurers (ABI) states that:<sup>114</sup>

*Weather risks are already increasing by 2 - 4 % per year on the household and property accounts due to changing weather. Claims for storm and flood damages in the UK have doubled to over £6 billion over the period 1998 - 2003, compared to the previous five years, with the prospect of a further tripling by 2050.*

The DTI's Foresight Programme has a far more serious prognosis suggesting that flood damages may increase twentyfold over 100 years.<sup>115</sup> It is quite clear that the insurance industry believes that changing risks associated with climatic events are driving insurance claims upwards, and flood damage is playing a significant part in this. A recent report for the ABI puts it thus:

*Flood and subsidence cost the UK insurance industry a significant amount today. Average annual claims for flood-related property damage are around \$1.28 bn (£0.7 bn) while subsidence damage equates to \$0.55 bn (£0.3 bn).*

*But weather claims are much more volatile than many other insured risks, e.g. fire or crime. Typically, weather damage accounts for one quarter of total property claims in the UK, but this may rise to between one third and one half of total claims in event years such as 1990 and 2000. The severe storms in 1990 in the UK led to property claims of more than \$4 bn (£2.4 bn, 2004 prices), while the floods in autumn 2000 resulted in insured costs of \$1.8 bn (£1 bn, 2004 prices).<sup>116</sup>*

In scenario analysis, the same report anticipates things becoming much worse:

*Flooding losses represent by far the greater source of climate change impacts on property insurance, with increases of almost 15-fold by the end of the century under the high emissions scenario, leading to potential total losses from river, coastal and urban flooding of more than \$40 bn (£22 bn).<sup>117</sup>*

This is reflected in **Error! Reference source not found.**, depicting insured losses associated with flood events in Europe, Japan and the US.

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<sup>113</sup> Defra (2001) *National Appraisal of Assets at Risk from Flooding and Coastal Erosion, Including the Potential Impact of Climate Change*, Final Report, July 2001

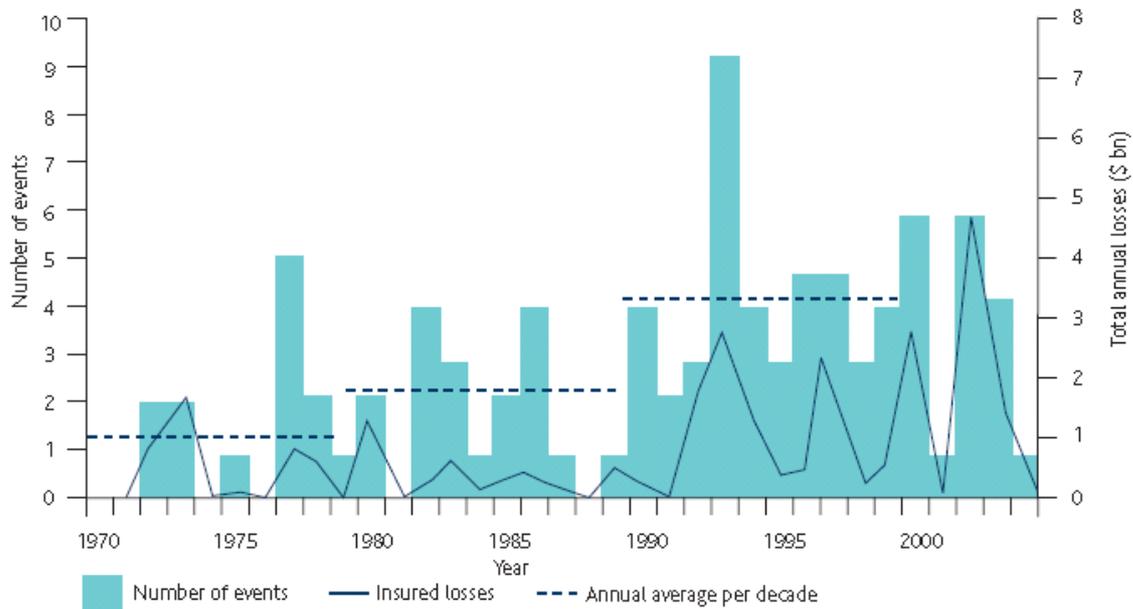
<sup>114</sup> A. Dlugolecki (2004) *A Changing Climate for Insurance: A Summary Report for Chief Executives and Policymakers*, Report for the Association of British Insurers, June 2004.

<sup>115</sup> See Norwich Union Risk Services (2004) *The Cost of Floods Could Escalate*, [http://www.nu-riskservices.co.uk/news/articles/cms/10841105336264214118114\\_1.htm](http://www.nu-riskservices.co.uk/news/articles/cms/10841105336264214118114_1.htm), 23/04/2004.

<sup>116</sup> Climate Risk Management and Metroeconomica (2005) *Financial Risks of Climate Change*, Report to the ABI, June 2005.

<sup>117</sup> Climate Risk Management and Metroeconomica (2005) *Financial Risks of Climate Change*, Report to the ABI, June 2005.

Figure 16: Number of Significant Flood Events and Insured Losses (2004 prices) in the US, Europe and Japan 1970-2004



Source: Sigma Database, Swiss Re

These two effects – increased flooding, and increased demand for irrigation water – are consistent with the out-turns of climate change models, suggesting the likelihood of drier summers and wetter winters, with increasingly ‘intense but infrequent’ rainfall events. Evidently, the use of compost may play a part, albeit a small one, in reducing emissions of greenhouse gases. But as far as both the use of irrigation water, and the increasing risk of flooding are concerned, the use of compost may have a role to play in reducing both. This is due to the fact that compost improves soil quality and increases the capacity to hold moisture.

#### A.6.10.1 Changing Irrigation Demand

Studies have indicated that the application of composted products can enhance the water use efficiency by improving infiltration and storage in the root zone and reducing deep drainage, run-off, and evaporation, and water use by weeds. The beneficial effects of compost application arise from improvements in soil physical and chemical properties.<sup>118</sup>

<sup>118</sup> A. Shiralipour, D. B. McConnel and W. H. Smith (1992) Physical and Chemical Properties of Soils as Affected by Municipal Solid Waste Compost Application, *Biomass and Bioenergy* 3(3-4): 261-266; S.A.R. Movahedi Naeini and H. F. Cook (2000) Influence of Municipal Compost on Temperature, Water, Nutrient Status and the Yield of Maize in a Temperate Soil. *Soil Use and Management* 16:215-221; L. M Bresson, C. Koch, Y. Le Bissonnais, E. Barriuso and V. Lecomte (2001) Soil Surface Structure Stabilization by Municipal Waste Compost Application. *Soil Sci. Soc. Am. J.* 65:1804-1811; J. Albaladejo, V. Castillo and E. Diaz (2000) Soil Loss and Runoff on Semiarid Land as Amended with Urban Solid Refuse, *Land Degradation & Development* 11: 363-373; M. Agassi, A. Hadas, Y. Benyamini, G. J. Levy, L. Kautsky, L. Avrahamov and H. Zhevelev (1998) Mulching Effect of Composted MSW on Water Percolation and Compost Degradation Rate. *Comp. Sci. Util.* 6(3): 34-41.

When used in sufficient quantities, the addition of composted soil conditioner has both an immediate and long-term impact on soil structure. An increasing proportion of UK soils are characterized by declining organic matter and poor structure. As organic matter decays to humus, the humus molecules bind mineral components of the soil (such as particles of sand, silt, and clay) and organic matter into water stable aggregates and improve soil porosity and soil structure. Due to the aggregate stability and improvements in soil structure, the application of composted soil conditioner reduces surface sealing, improves infiltration and the water holding capacity thus reducing runoff generation.

These aggregates are also effective in holding moisture for use by plants. In addition, humus molecules can absorb and hold large quantities of water. Therefore, the addition of composted soil conditioner may provide greater drought resistance and more efficient water use. Thereby, the frequency and intensity of irrigation may be significantly reduced in irrigated agriculture. Increased porosity and decreased soil compaction may also result in increased root penetration, resulting in deeper and more elaborate root systems to explore a larger soil mass for moisture and nutrients. Increased root exploration and water holding capacity can also reduce deep drainage below the root zone, resulting in reduced nitrate leaching. Low nitrate leaching can reduce eutrophication of water resources.

Use of composted mulch in cropping lands can also significantly increase the water use efficiency by lowering the evaporation losses from soil surface. Mulching reduces radiation and wind speed at the surface and hence, reduces the water evaporation from soil surface. Researchers have reported that surface application of mulch has resulted in reduction of between 30–70% of irrigation water required by crops due to the reduction of water evaporation from soil surface.<sup>119</sup> Buckerfield and Webster showed that, in South Australia, the surface application of organic mulches to vineyard soils resulted in a 34% increase in soil moisture content and an increase in grape yield.<sup>120</sup> This reduced evaporation offers obvious benefits for irrigated agriculture.

To the extent that compost application increases water holding capacity, it results in a reduction in leaching and irrigation water requirements. As a consequence, water extractions from natural waterways will be reduced leading to increased water flows and improved water quality in natural river systems and reduced impacts on biodiversity. Moreover, reduced leaching will result in reduction in eutrophication and induced salinity. In addition, increased water holding capacity of soil can reduce stresses due to soil moisture deficits leading to higher crop yields.

On the basis of a literature review, one study has sought to plot relationships between compost applications as soil improvers (**Error! Reference source not found.**) and mulches (**Error! Reference source not found.**) on soil moisture content in the 0-15cm

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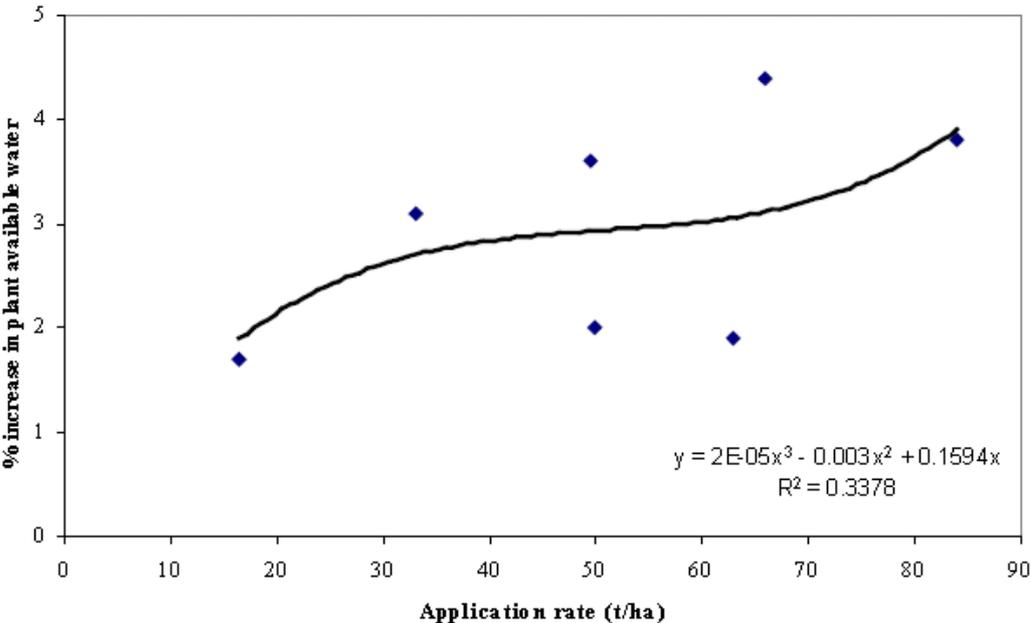
<sup>119</sup> A. M. Abu-Awwad (1998) Effect of Mulch and Irrigation Water Amounts on Soil Evaporation and Transportation. *J. Agron. Crop Sci.* 181: 55-59; A. M. Abu-Awwad (1999) Irrigation Water Management for Efficient Use in Mulched Onion. *J. Agron. Crop Sci.* 183: 1-7

<sup>120</sup> J. C. Buckerfield and K. A. Webster (1995) Earthworms, mulching, soil moisture and grape yields: earthworm response to soil management practices in vineyards, Barossa Valley, *South Australia. Australian and New Zealand Wine Industry J.* 11:47-53

layer.<sup>121</sup> This was used to estimate the quantity of water saved through the application of compost. If the same relationships are assumed (and clearly, they may not hold in the UK context), then it could be assumed that an application rate of 20 tonnes per hectare fresh matter would deliver a 2% increase in plant available water.

On the basis of work undertaken by Defra, where irrigation is applied, it was applied at a rate of 131,300,000 m<sup>3</sup> over an area of 147,270 ha. In dry years, the figures are estimated at 439,470,000 m<sup>3</sup> over 282960 ha.<sup>122</sup> An average of these figures gives an estimated 1,222m<sup>3</sup> of water used per hectare. A conservative estimate would be that the compost reduces this figure by 2%, or 24m<sup>3</sup>. On the other hand, rain water supplies some of the plant available water. If this is assumed to be 60% of the total, then the 2% increase in availability implies a reduced demand of 61m<sup>3</sup> of water.

Figure 17: Effect of Compost Used as Soil Conditioner on Soil Moisture (0-15cm layer)



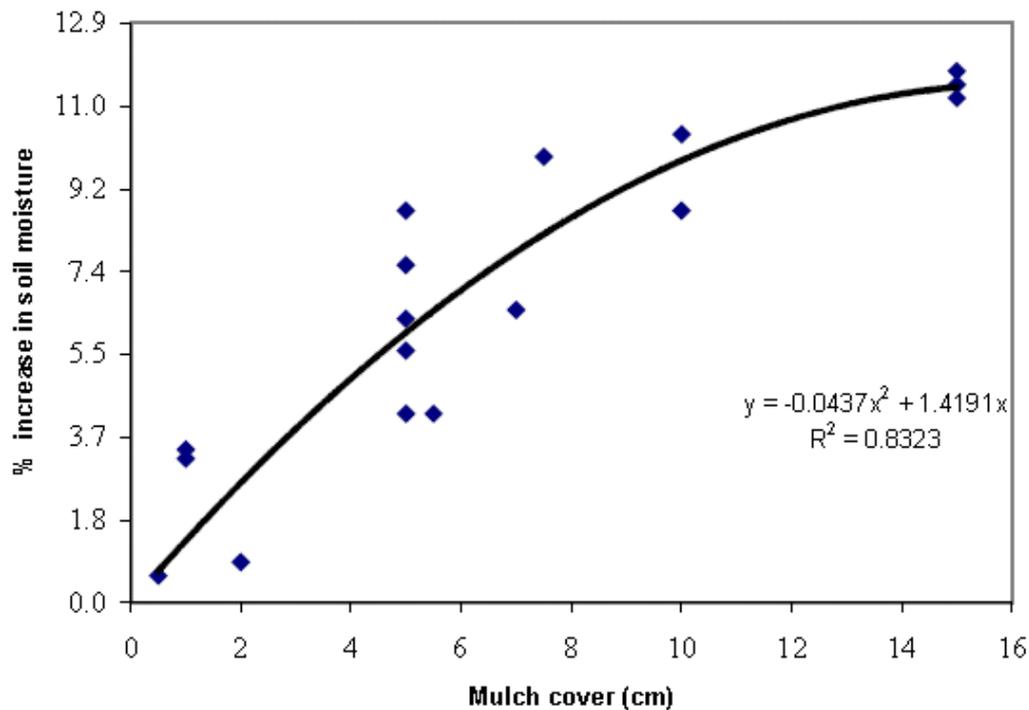
Source: G. Sharma and A. Campbell (2003) *Life Cycle Inventory and Life Cycle Assessment for Windrow Composting Systems*, Report for Recycled Organics Unit, University of New South Wales and NSW Dept. for Environment and Conservation, October 2003

<sup>121</sup> G. Sharma and A. Campbell (2003) *Life Cycle Inventory and Life Cycle Assessment for Windrow Composting Systems*, Report for Recycled Organics Unit, University of New South Wales and NSW Dept. for Environment and Conservation, October 2003

<sup>122</sup> E. K Weatherhead and K. Danert (2002) *Survey Of Irrigation Of Outdoor Crops in 2001 – England*, Research for Defra’s Climate Change and Demand for Water (CCDeW) project, October 2002.

The value of these savings is very difficult to estimate. The value of water is multi-dimensional. It extends beyond water *prices*, and includes values of water in terms of its 'use' and its 'non-use' values.<sup>123</sup>

Figure 18: Effect of Compost Used as Composted Mulch on Soil Moisture (0-15cm layer)



Source: G. Sharma and A. Campbell (2003) *Life Cycle Inventory and Life Cycle Assessment for Windrow Composting Systems, Report for Recycled Organics Unit, University of New South Wales and NSW Dept. for Environment and Conservation, October 2003*

One study reviewed a range of studies and gave figures for the western US as shown in **Error! Reference source not found.**<sup>124</sup> It suggested (amongst other things) that:

- irrigated agriculture accounts for a large proportion of water use, especially in many water-scarce areas, and the value of water for many low-value crops, such as food grains and fodder, is very low. Nevertheless, the value of water can be high (of the same order of magnitude as values in M&I end uses) where reliable supplies are used on high-value crops.
- The value of water for household purposes is usually much higher than the value for most irrigated crops. However, within household usages, the value

<sup>123</sup> See, for example, K. Turner, S. Georgiou, R. Clark, R. Brouwer and J. Burke (2004) *Economic Valuation of Water Resources in Agriculture: From the Sectoral to a Functional Perspective of Natural Resource Management, FAO Water Reports 27, 2004.*

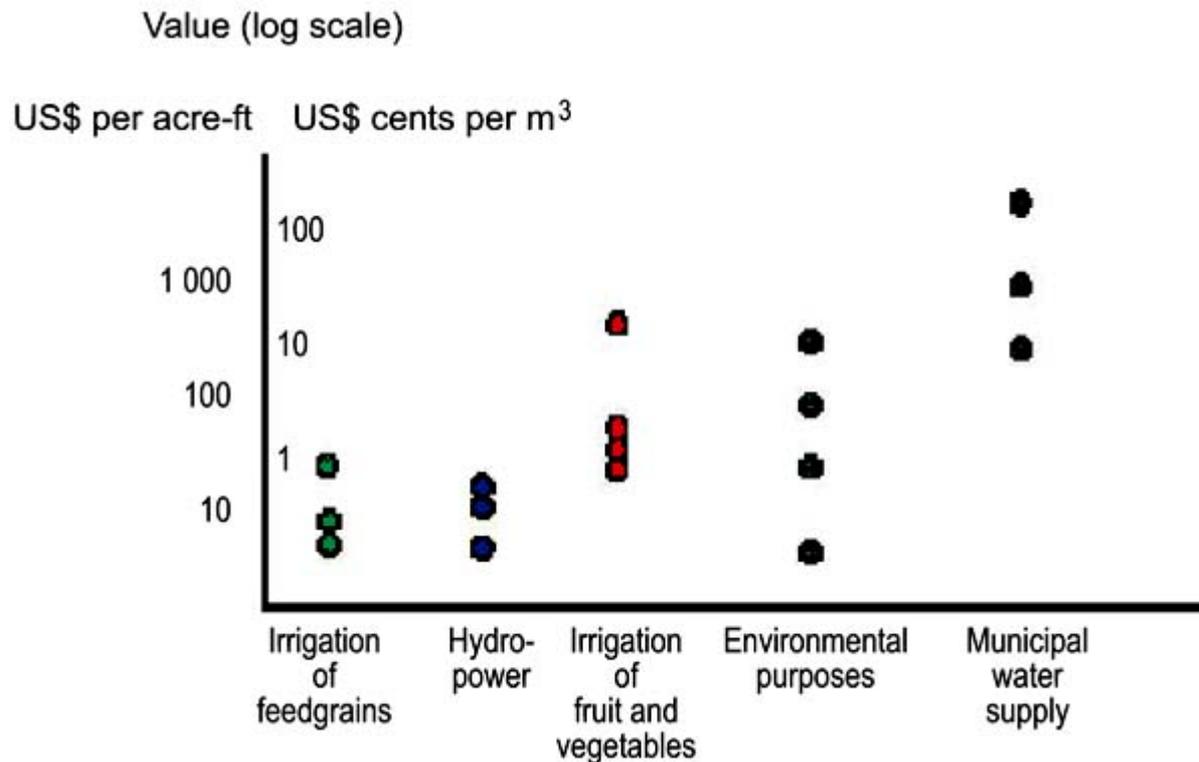
<sup>124</sup> J. Briscoe (1996) *Water as an Economic Good: the Idea and What it Means in Practice.* Paper presented at the World Congress on Irrigation and Drainage, Cairo, September 1996.

for 'basic human needs' is much higher than the value for discretionary uses (such as garden watering). Reliability of supply is an important factor in how the resource is valued.

- The value of water for industrial purposes is typically of a similar order of magnitude to that of supplies for household purposes;
- The value of environmental and ecological purposes varies widely but typically falls between the agricultural and municipal values (as shown in **Error! Reference source not found.**).

We could find no UK-based study which seeks to capture all use and non-use values of water. Most UK studies relate to the value of recreation, or to the value of wetlands, or to some measure of water quality. Avoiding abstraction can, in theory, contribute to each of these.

Figure 19: Typical Market and Non-market Values for Water Use in the Western United States of America



Source: J. Briscoe (1996) *Water as an economic good: the idea and what it means in practice*. Paper presented at the World Congress on Irrigation and Drainage, Cairo, September 1996.

In England and Wales, for agricultural abstractors, the *pricing* mechanism depends on the nature of the source, the seasonality of abstraction and the loss rates expected, as well as the region in which the abstraction occurs. Taking the 'charge factor' to be of the order 2 (average of unsupported and supported, multiplied by summer seasonality multiplied by loss factor for agricultural uses excluding trickle and spray

irrigation), then the cost per 1,000 m<sup>3</sup> would be of the order £20.56-48.54, depending upon the region.

Briscoe's figures from the US suggest that a value of US\$0.10 per m<sup>3</sup> might be a conservative estimate of the total economic value of water in the US context. UK-based agricultural values may be higher (at least, historically) as a consequence of price supports under the Common Agricultural Policy. The value of US\$0.10 per m<sup>3</sup> in 1995 translates into a value of £0.063 per m<sup>3</sup> in 1995 terms, or £0.81 per m<sup>3</sup> at current prices (converted using HM Treasury GDP deflators). This value would be equivalent to around £810 per 1,000 m<sup>3</sup>. This is some 30-40 times the existing price level.

Wessex Water looks at the replacement cost of abstractions in exceptional circumstances:<sup>125</sup>

*there are 15 rivers where we are assessing the impact of other abstractions. These impacts are unlikely to be resolved until after 2010. The figure in the accounts is an annualised estimated cost, based on necessary capital expenditure for remedying low flows in these locations. Within the accounts we have estimated that the assessment will show that 20MI/d of current abstractions should be replaced with alternative resources. Assuming a replacement cost of £2m per MI and discounted over 30 years at 6% this is equivalent to £2.9m.*

Given that 1 million litres of water is equal to 1,000 m<sup>3</sup> of water, the suggestion is that the capital investment required is £2 million per 1,000 m<sup>3</sup>. Discounted at 6% over 30 years, this suggests an annual cost of £145,297 per 1,000 m<sup>3</sup>.

It is clear that the range of values one could derive is enormous. This is a major source of uncertainty in this study. If values are as high as replacement costs indicate they may be, then essentially, this benefit may over-ride all others.

However in this study, we have used the transferred benefits from the US study. We feel this is too important a benefit to be assigned zero value, yet there is no reasonable basis upon which to base a UK estimate at present.

It is perhaps worth adding that no life cycle assessment actually takes into account water use. This is potentially – as this section highlights – a major omission from all LCAs, and one which ought to be addressed by practitioners.

### 6.10.1.1 Flood Risk

The issue of reduced flood risk is another area where benefits from compost utilisation *could* be high, and where they are (as with water savings) likely to increase in value over time. The basis for estimating any value on this is simply too shaky. However, it is clearly an area for further investigation, and those seeking to ensure drainage from new developments is unlikely to give rise to new problems would clearly do well to seek to ensure that surface run-off is minimised through attention to solid surfaces and to the quality of soil.

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<sup>125</sup> Environmental Accounts

<http://www.wessexwater.co.uk/strikingthebalance2005/finances/index.aspx?id=1191>

### A.6.10.2 Effects on Nitrate Leaching from Soil

Nitrogen from mineral fertiliser is the major source of nitrogen input in the EU. Excessive nitrogen surpluses can pose a threat to the environment leading to pollution of air, water and soil.

Traditionally, following essentially organic production methods of crop rotation and regular fallow periods together with the spreading of animal manures allowed the soil to recover some of its fertility. Today however, the main method used to restore nutrients and to increase crop yields is to apply mineral fertilisers.

Nitrogen in commercial fertilisers is readily soluble to facilitate uptake by crops, which in conjunction with excessive application can pose a threat to the environment and in some cases affect the fertility of the soil itself. Losses to the environment can be minimised if sustainable agricultural practices are followed and reasoned fertilisation is used (taking into account weather conditions to reduce the incidence of runoff and applying at the appropriate stage of crop growth, using appropriate doses).

Nitrogen when applied as uncomposted animal manures or inorganic nitrogen fertilisers also have the potential to volatilise and lose more than 50% of their nitrogen to the atmosphere within the first few days following application to land. When animal manures are spread nitrogen is lost to the atmosphere through volatilisation as ammonia or as the greenhouse gas N<sub>2</sub>O.

Nutrients that are not taken up by plants may be metabolised by micro-organisms in the soil which will improve soil fertility. However this is a slow process and there is a risk that soluble nutrients such as nitrate will run off into surface water or percolate into groundwater reservoirs.

Combined, excessive amounts of nitrogen and phosphorous can result in eutrophication in lakes, rivers and coastal areas, resulting in the proliferation of toxic blue-green algae. Soils can also be at risk of eutrophication, where excess nutrients deplete the soil of oxygen, resulting in a reduction of natural micro-flora and subsequent reduction in soil fertility.

Nitrogen supplied from compost is not immediately available. Approximately 40% is available in the first year following application, 20% in the second year and 10% in the third, slowly decreasing every subsequent year. Therefore, composting when managed correctly is a form of nitrogen conservation. As the most of the nitrogen in compost is not in a form that is immediately available to the soil, there is less risk of nitrogen volatilisation and nitrogen leaching; this is especially relevant in nitrate sensitive areas.

In recent trials, green waste compost has been applied at three times the maximum N application rates in the Code of Good Agricultural Practise without significantly increasing nitrate leaching. This should be related to the high percentage of slow-release N (organic N) that tends to be released over longer time frames than in the case of chemical fertilisation.<sup>126</sup>

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<sup>126</sup> L. Jackson, personal communication, based on series of reports, Researching the Use of Compost in Agriculture 1997-2001, HDRA Consultants.

The effects of displacing the equivalent quantity of nitrate fertiliser from the soil are not only that one avoids burdens associated with their manufacture. In addition, there is a reduction in the leaching of nitrate into groundwater.

As we have seen, 10 tonnes of dry matter applied in year one displace a quantity of N from commercial fertiliser in line with the schedule set out in **Error! Reference source not found.** Assuming that the 23% of nitrate lost is leached to groundwater, this would imply that the quantity of N being leached into groundwater follows the schedule outlined in **Error! Reference source not found.**

Work on economic valuation of nitrate pollution of groundwater is relatively scarce. There are a number of difficulties associated with this, not the least of these being the fact that such leaching as occurs today may only affect people a generation or more in the future.

**Table 31: Reduction In Nitrate Leached Into Groundwater Associated With 10 Tonnes Dry Matter Compost Application**

Year	Reduction in N leached (kg)
1	13.44
2	9.41
3	6.59
4	4.61
5	3.23
6	2.26
7	1.58
8	1.11
9	0.77
10	0.54

The cost of removing nitrate from groundwater is not insignificant. A recent study in the UK estimates the cost of nitrate removal at £18.8 million per year in the years 1992-1997 (capital expenditure) plus £1.7 million (operating expenditure).<sup>127</sup> The study estimated that 80% of nitrate originated from agriculture, giving a cost of removal of £16.4 million per annum.

Two studies have sought to elicit values for nitrate free water supplies. One, study in the UK based upon a willingness to pay study, estimated that households would pay €25.2 (2000 values) to guarantee water supplies with nitrate levels not exceeding 50mg/l.<sup>128</sup> Grossing up on the basis of 835,212 households,<sup>129</sup> the aggregate

<sup>127</sup> J. N. Pretty, C. Brett, D. Gee, R. E. Hine, C. F. Mason, J. I. L. Morrison, H. Raven, M. D. Rayment and G. van der Bijl (2000) An Assessment of the Total External Costs of UK Agriculture, *Agricultural Systems* (65) pp.113-136.

<sup>128</sup> N. Hanley (1990) The Economics of Nitrate Pollution, *European Review of Agricultural Economics*, 17, pp.129-51.

<sup>129</sup> This was the figure used by Stewart et al (1997) in their evaluation of the value for money of the UK Nitrate Sensitive Areas scheme (see Lisa Stewart, Nick Hanley and Ian Simpson (1997) *Economic Valuation of the Agri-environment Schemes in the United Kingdom*, Report to HM Treasury and the Ministry of Agriculture Fisheries and Food, September 1997).

willingness to pay for this reduction in NVZs is €21 million. This estimate does not, however, make it possible to quantify the cost per tonne of nitrate leaching into groundwater. A study carried out in Gotland in Sweden sought to elicit preferences for nitrate pollution in aquifers to be reduced to levels below the WHO recommended limit. This value was SEK 600 (1995 prices) per person per year.<sup>130</sup>

More recently, work by Brauer has estimated the value of nitrogen abatement by wetlands:

*Depending on the production system and the intended reduction level the replacement costs vary between 1-23 €/kg N (Hennies 1996 unpubl.). For the calculations in the presented study average costs of 2.56 €/kgN are assumed. To make this number comparable: for nitrogen removal in sewage plants marginal costs of 5-8 €/kgN are reported in Germany (Grünebaum 1993).*

Again, given the limited availability of such values, we have used a figure of £2.00 per kg N. These enable the development of estimates as to the value of the 'avoided leaching' associated with the nitrogen displacement.

These estimates are, again, uncertain. There is a lack of studies enabling an assessment of the effects of marginal increases in nitrogen being leached.

### A.6.11 A Note On Financial Savings From Avoided Fertiliser Applications

There is an interesting question as to whether the private savings which arise from the use of compost should be considered in an analysis of external costs and benefits of the use of compost. Theoretically, and in many cases, in practice also, such savings should not be included. People who buy compost would be expected to 'internalise' such savings in their decision as to whether or not to purchase the product.

However, there may be effects which arise from the use of compost which might not be obviously attributable to the use of compost itself. A good example would be the disease suppressing effect of compost. Reduced outlay on pesticides might not necessarily be linked to the application of compost (not least since it is not easy to know what the counterfactual scenario would have been). Arguably, the more people understand the benefits associated with compost, the smaller is the justification for considering the private savings as 'an external benefit'. In the world of perfect information, the effect is internalised in the decision making process (and arguably, the market for compost would improve where such benefits were understood and realised). We have not considered the private savings as external benefits in this study. However, for completeness, we estimate them here.

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<sup>130</sup> I.-M. Gren, H. H. Groth, and M. Sylvén (1995): Economic Values of the Danube Floodplains. *Journal of Environmental Management* 45:333-345.

## A.6.12 Unquantified External Benefits

### A.6.12.1 Effects On Soil Physical And Biological Properties

Soil physical properties affect crops both directly and indirectly. The structure, porosity, aeration and moisture holding capacity are part of the root environment and so have direct effects on crop growth and nutrient release from the soil reserves. The structure of the soil will determine suitability and timings of cultivation; the better the soil structure the more scope there is for tillage. An ideal soil for crop production has a highly stable structure which is not easily destroyed by cultivation, water movement or treading by livestock (poaching). Structural stability is increased by increasing organic matter levels; the naturally occurring 'gums' and 'polysaccharides' 'hold' the soil together.

As structure improves aeration, drainage and ability to provide water and nutrients to plants also increase. The soil organic matter may be fractionated chemically<sup>131</sup> or physically in an attempt to separate it into functionally distinct types. It is generally considered that the organic matter light fraction, containing recognisable plant and animal debris, is the source of most of the readily mineralisable nutrients. The heavy fraction, more closely associated with the soil minerals, contains organic matter that is more highly humified (containing very large organic molecules with strong bonds) and stable over very long time periods. As discussed above, the composting process may produce organic matter which contributes to both these categories in the soil.

Although compost is high in organic matter it has yet to be proven that beneficial changes in the soil structure have been achieved following its application. Trials at IACR Rothamstead have shown an improved efficiency of mineral fertilisers where soil organic matter is high and this has certainly also been seen in other trials.<sup>132</sup> There are many techniques for quantifying changes in soil physical characteristics (e.g. cohesion, sheer strength, water retention etc.) which may be brought about by organic matter changes but their interpretation is notoriously difficult. Sometimes trials have investigated other related soil features, as water retention and moisture intervals at different sucking forces; such features are primarily linked to the absorbing capacity of organic matter and to the porosity and its distribution between micro- and macroporosity.

While there may be debate about the actual nature of the soil improvements and the mechanisms whereby soil properties are affected, there is no doubt that high organic matter levels are generally associated with ease of soil management, better crop establishment and plant growth. Compost is a 'living' material and can only increase the density and diversity of the soil and microbiological populations which seem to then increase productivity of the soil.

It is clear that in practice one of the most important reasons why farmers are, and will, in future, be using compost is to improve soil structure, and it is therefore

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<sup>131</sup> M. Schnitzer (1982) Organic Matter Characterisation, in A. L. Page, R. H. Miller & D. R. Keeney (eds) (1982) (Methods of Soil Analysis, *Agronomy* 9 (2) pp 581-593.

<sup>132</sup> D. S. Jenkins (1991) The Rothamstead Long-term Experiments: Are They Still of Use? *Agronomy Journal* 83.

important this effect is studied in greater depth. The effect needs to be quantitatively measured in order to make recommendation on how to make the best use of compost for this purpose.

It is also worth mentioning that codes of practice for organic farming stress the need to compost the organic matter in order to both stabilise it (thus avoiding any undesired side-effect related to oxygen uptake and release of phytotoxic compounds during mineralisation) and to activate its biological diversity.

### A.6.12.2 Reduced Requirement for Liming

One of the effects of compost on soil is to act as a buffer against changing pH of the soil. One typical remedy for falling pH is to apply lime to the soil. Lime is occasionally acquired as a by-product of industrial processes but more typically it is a product of mining. The effect of compost, therefore, may be to avoid the extraction of lime, which would otherwise incur external costs. Furthermore, farmers outlays on lime would also be reduced.

Applications of green waste compost in trials at the Henry Doubleday Research Association (at rates to supply 250, 500 and 750 kg N/ha) were found to raise pH from 6.5 to between 6.8 and 7.4, with the highest pH resulting from the highest rate of compost application (supplying 750kg N/ha). This precluded the use of lime to maintain pH in this trial.<sup>133</sup>

Pot trials assessing performances of compost versus traditional peat-based growing media, have shown that compost has a much higher buffer capacity e.g. versus alkaline waters that tend very often to raise the pH causing a reduction in growth, flowering, and so on.<sup>134</sup> This effect is linked to the higher cation exchange capacity (per unit volume) of compost as opposed to peat, whilst the cation exchange capacity per unit weight tends to be quite similar.

We have no clear relationship between pH changes and compost applications. Therefore, we have not attributed any benefits to compost owing to this effect simply because the relationships which would enable such quantification are not available.

### A.6.12.3 Reduced Susceptibility to Soil Erosion

The condition of soil surface determines whether rainfall infiltrates the soil or simply runs off. Soil therefore regulates and partitions water. When water runs off land, it tends to carry soil particles. This results in costs to farms in terms of lost productivity and off-farm impacts such as damage to commercial and recreational fishing, increased pressure on water treatment facilities, increased flood damages and requirement for repairs from redredging damaged waterways.

It is increasingly recognised that off-farm costs of soil erosion are probably greater than on farm ones. The off-farm costs associated with soil erosion in the US due to

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<sup>133</sup> L. Jackson, personal communication, based on series of reports, *Researching the Use of Compost in Agriculture 1997-2001*, HDRA Consultants.

<sup>134</sup> E. Favoino and M. Centemero (1995), Impiego di compost nella vivaistica: esperienze applicative svolte nel corso del 1994; in attachment to *Notiziario della Scuola Agraria del Parco di Monza*, May 1995.

waterways alone were estimated at \$2-\$17 billion.<sup>135</sup> In the UK, a 1996 study estimated soil erosion impacts at between £23.8 - £50.9 million (1991 prices) with off-farm losses responsible for as much as 80% of this figure.<sup>136</sup>

In future, severe storms may generate the bulk of soil erosion losses, and this may be a possible 'positive feedback' associated with global warming in the future. Air-borne soil particles may also have impacts on human health, and their presence could be reduced through greater use of organic matter to bind soil into stable aggregates. Management factors play a role in reducing erosion, but so also does the soil texture and organic matter content.

It is difficult (for obvious reasons) to estimate the incremental reduction in soil erosion associated with applications of compost. However, the benefits associated with reduced soil erosion are potentially significant. We have not quantified them here.

#### A.6.12.4 Benefits from Improved Infiltration

Benefits from improved infiltration of water arise through reduced risk of flooding (and soil erosion – see above) and reduced requirement for irrigation water.

##### Reduced Irrigation Requirement

Water holding capacity can be increased by as much as 3-5% through application of soil organic matter. The avoided environmental burden is difficult to assess at the margin, though it is possible to place values upon water in specific contexts (and indeed, many argue for the use of tradable permits as an allocation mechanism to ensure efficient use of water). Other benefits relate to the increased survival rates of unmanaged young trees in dry periods.

It is difficult to quantify these savings, either in terms of environmental benefits or financial savings, because of the varying nature of the demand for water for agriculture across countries. The financial savings to be realised depend very much on the charging regime for water. In many countries, water for agriculture is still made available on a flat fee or per hectare basis. Consequently, there is no marginal benefit to be gained from reduced consumption. Such savings may become more important in the future, however, as it is likely that more and more countries will move towards marginal cost pricing for water resources.

##### Reduced Risk of Flooding

Again, quantification of any benefit here is extremely difficult. For a start, it is not always the case that floods impose costs. However, the degree to which compost reduces any risks, and associated costs, of flooding would be difficult to discern in

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<sup>135</sup> National Research Council (1989) Problems in US Agriculture, in *Alternative Agriculture*, Washington DC: National Academy Press; M. Ribaud (1989) Water Quality Benefits from the Conservation Reserve Programme, *Agricultural Economic Report No.606*, Washington DC: USDA Economic Research Service; D. Pimentel, C. Harvey, P. Resosudarmo, K. Sinclair, D. Kurz, M. McNair, S. Crist, L. Shpritz, L. Fitton, R. Saffouri and R. Blair (1995) Environmental and Economic Costs of Soil Erosion and Conservation Benefits, *Science* 267 (5201): pp.1117-1123.

<sup>136</sup> R. Evans (1996) *Soil Erosion and its Impact in England and Wales*, London: Friends of the EarthTrust.

anything other than a location-specific context (and the costs would have this character also).

### A.6.12.5 Improved Tilt

The fact that compost improves soil structure means that it is actually easier to work the soil with agricultural machinery. There are likely to be savings in fuel use resulting from this change in soil quality. This will be offset by fuel requirements to spread compost on the soil (depending upon what is being 'displaced'). No attempt has been made here to quantify any cost / benefits associated with changes in the fuel use requirement.

### A.6.12.6 Compost and Bioremediation

Compost and the composting process can be used successfully in the bioremediation of contaminated soils. In-situ remediation is commonly used whereby compost is used essentially as an inoculate to the contaminated soil, providing the micro organisms which break down the contaminants. This form of bioremediation is not suitable for all contaminants, but it has been proven to be successful in treating soils that have been contaminated with, for example: hydrocarbons, aromatic compounds, and aliphatic compounds. We have not quantified any of these benefits.

### A.6.12.7 Micro nutrients and Compost Application

Compost can provide a broad spectrum of nutrients; both macronutrients and micronutrients such as S, Mg, Ca, Fe and Zn. Conventional fertilisers provide merely the basic macronutrients (such as N, P and K), neglecting the minor elements beneficial to plant growth. There may also be links here to the presence of micronutrients in dietary uptake. These are important in the functioning of the endocrine system in humans.

## A.6.13 Unquantified External Costs of Compost Application

For the externality calculation, the air emissions are the key ones to which costs can readily be attached. However, there are other emissions from the process which are of concern.

### A.6.13.1 Leachate

Figures for the production of leachate from composting are given by the Environment Agency study. These are compared with those from COWI for an unlined landfill site (see **Error! Reference source not found.**).

In the COWI study, the presence of a leachate collection system is deemed to lead to no leachate emissions. Adequate control of leachate at composting sites would be expected to lead to a similarly sanguine prognosis, but the levels of leachate measured also suggest this is unlikely to be a major problem at compost sites. Indeed, water tends to be applied in the process and is then evaporated. Careful management enables the 'excess water' to be used in the process itself. Hence, there is little need for treatment.

Table 36: Leachate From Compost Plant Compared With That From Landfill

Quality	Kg/tonne MSW	kg/tonne MSW
COD	0.457	11,411
Chloride	0.152	349
Mg	0	1,127
Ni	7.94E-04	
Cd	0	
Cr	0	0.1
Cn	7.28E-05	0.1
Cu	0	1
Pb	5.96E-04	0.4
Hg	0	0.01
Zn	2.38E-04	75

Sources: Environment Agency (2000) *Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion*, Bristol: Environment Agency; and COWI (2000) *A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste. Final Report to DG Environment, the European Commission, August 2000.*

### A.6.13.2 Odour

Even the most well managed site will produce odours. However, effective operational management should prevent the formation of unpleasant odours which can be a direct result of mismanagement or ineffective odour control measures. It can be noted in passing that odour is a subjective quality and what may be an inoffensive pine-like odour to one person may be quite unacceptable to another.

Odours are emitted from the surface of open piles, windrows, maturation piles, storage piles, and stockpiles of amendments. Exhaust gases from controlled aeration systems also contain odorous compounds. Typically the most problematic odorous compounds at composting facilities include, ammonia, hydrogen sulphide, mercaptans, alkyl sulphides such as dimethyl sulphide and dimethyl disulphide and terpenes. These compounds are present in many composting feedstocks or are formed during the process through aerobic or anaerobic actions.

Effective operational management can help to control the formation of odours these include:

- Processing incoming feedstocks as soon as possible
- Managing the process properly, including:
  - ✓ ensure proper stabilisation of the biomass within the retention time in enclosed buildings, so as to ensure only odourless materials are present in the open curing stage

- ✓ avoid an early refining step to reduce particle size too far, which would hinder the diffusion of air through the material that still has to complete its biochemical transformation (smaller particle size would cause the compost to lose structure and make anaerobic decomposition more likely);
- Ensuring good housekeeping practices are followed, such as:
  - ✓ Preventing the formation of leakage puddles (e.g. ensuring proper slopes to paved surfaces)
  - ✓ Avoiding external stockpiling of coarse rejects from pre-process screening steps, as these would also contain a certain percentage of fermentable materials

Besides prevention, often composting facilities have to tackle odour issues through treatment of exhaust air, above all where they feature high capacities and/or short distances from dwellings. From this standpoint, a comprehensive treatment strategy should ensure:

- Withdrawal of exhaust air from the odorous sections of the process (tipping, storage of input fermentable materials, pre-treatment, early process steps. Sometimes also the curing section can be enclosed and exhaust air treated)
- Design of the withdrawal system to prevent any loss of exhaust air from windows, doors, etc.
- Fitting the facility with properly dimensioned abatement systems (for biofilters, see criteria in the sections below)
- Ensure proper maintenance of treatment technologies (e.g. watering the biofilter, preventing its compaction, etc.)
- The release of odorous areas from tipping gates, above all where deep bunkers are used.

A good number of facilities across Europe are currently showing that technologies can help running of composting activities even in most crowded areas, provided design and management of the plant consider odour problems with the proper care.

### A.6.13.3 Odour Treatment Using Biofilters

Biofilters are used primarily to treat odours.<sup>137</sup> Biofilters use a biologically active, solid media bed to absorb/adsorb compounds from the air and retain them for their

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<sup>137</sup> VOC abatement is also a requirement at MBT sites as mixed waste contains some potentially hazardous VOC's (paintings, solvents, etc.). The German Government has set a limit value for overall VOCs emissions at 55 grams/ton; this applies for the time being only to MBT facilities. At some composting facilities, the majority of VOC's are being produced by the biofilter itself in a natural way, as e.g. terpenes come from degradation of the wooden materials of the biofiltering media. This is why VOC's abatement would potentially be a misleading goal for composting plants. 101

subsequent biological oxidation. They work on a simple principle whereby odorous gas passes through the media.

Typically a manifold system or a plenum is used to distribute the gas through the media which is usually between 1-2 m deep. Sometimes a wet scrubbing system is installed before the biofilter in order to improve overall efficiency, by both stripping soluble odorous compounds and humidifying inlet air so as to prevent a fast drying out of the biofiltering media. Chemical scrubbers can also be used, with different chemicals fitting different types of odorous compounds to be removed.

Organic material is usually used as a biofilter media as it has a higher biological activity than soil. The properties of compost: high surface area, air and water permeability, water holding capacity, high microbial population and relatively low cost make it an ideal biofilter media. Lately surveys and applications have been increasingly focusing on the highest suitability of coarse activated organic materials, such as the wooden rejects from yard waste composting sites, as they can better withstand the tendency of the biofilter to shrink and give rise to short circuiting of air (this problem is one of major maintenance issues regarding biofilters; materials with a lack of structure can shrink in as little as a few weeks, while coarse materials can last up to 5 years or so).

Biofilter media can vary from locally available compost to specifically designed media mixtures containing ingredients such as: compost, soil, peat, bark, wood, lime (deemed to withstand the tendency to acidification related to the nitrification of ammonia), lime and polystyrene spheres.<sup>138</sup> The desired qualities of a biofilter are outlined in **Error! Reference source not found.**

The removal efficiency of a biofilter is determined by the gas residence time in the media bed (residence time is calculated by dividing the seconds in a hour – 3600 - by the specific loading rate, expressed in  $\text{cm}_{\text{air}}/\text{hr} \cdot \text{cm}_{\text{biofilter}}$  ). Effective residence times typically range from 30 to 60 seconds for most compost applications.

A large number of composting facilities across Europe use biofilters to control odours at composting facilities. Studies have reported high removal efficiencies for specific compounds such as  $\text{H}_2\text{S}$  (>99%), methyl mercaptan, dimethyl disulphide, dimethyl sulphide (>90%) and various terpenes (>98%).<sup>139</sup>

As a biofilter is a biological treatment process certain conditions must be maintained to ensure the viability of the microbial flora. Moisture and pH must be monitored and maintained in order to ensure effective operation. Dehydrated or waterlogged conditions will limit microbial activity and therefore the efficacy of the biofilter. The control of moisture actually drives most of maintenance in the short run, while the control of porosity and of the even distribution of inlet air is the main maintenance issue in the medium to long term.

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<sup>138</sup> Evidently, some of these such as the lime and carbon examples, work not through biological activity but through absorption.

<sup>139</sup> See, e.g. H. U. Hartenstein and E. R. Allen (1986) *Biofiltration, An Odour Control Technology For A Wastewater Treatment Facility*, Report to the City of Jacksonville, FL 1986.

**Table 32: Qualities of Biofilter Media**

Characteristic	Description
Filter media	Biologically active, but reasonably stable, Organic matter content >60% Porous and friable with 75-90% void volume Resistant to water logging and compaction Relatively low fines content to reduce gas headloss Relatively free of residual odour Specifically designed mixtures of materials may be desirable to achieve the above characteristics
Moisture content	50-80% by weight Provisions must be made to add water and remove bed drainage
Nutrients	Must be adequate to avoid limitations Usually not a problem with composting gases because of the high NH <sub>3</sub> content
PH	7 to 8.5
Temperature	Near ambient, 15-35 or 40°C
Gas pre-treatment	Humidification could prove to be useful in order to achieve near 100% inlet gas humidity Dust and aerosols may be removed to avoid media plugging, but for most biofilters this is not a problem (unless they have a tissue layer in the bottom)
Gas loading rate	<100 m <sup>3</sup> /h-m <sup>3</sup> , unless testing supports higher loadings
Gas residence time	30-60 seconds, unless testing supports shorter residence time
Media depth	>1m, < 2 m
Elimination capacity	Depends on media and compound (typically in the range 10 – 160 gm <sup>-3</sup> h <sup>-1</sup> *)
Gas distribution	The manifold must be properly designed to present a uniform gas flow to the media

*Source: Adapted from Haug, R. T. (1993) The Practical Handbook of Compost Engineering and Swanson W.J., and R. C. Loehr (1997) Fundamentals, Design And Operating Principles, J. Envir. Eng., 538-546, 1997.*

#### **A.6.13.4 Dusts and Bioaerosols**

Bioaerosols are micro-organisms and other tiny biological particles that are suspended in air. They are respirable and generally invisible. Dusts are small particles that are larger than bioaerosols. They are inhalable but not respirable and are visible. It should be noted that bioaerosols from the composting process contain the same micro-organisms as ones to which citizens are routinely exposed. They are present naturally and are essential to the recycling of nutrients in our gardens, parks and countryside.

The number of individual organisms necessary to cause a reaction varies according to the state of health of the person exposed to it. In the composting process, the levels

encountered are significantly higher.<sup>140</sup> Therefore it is imperative that steps should be taken to protect site operatives and residents in the surrounding areas.

Again, effective operational management can help to control the formation of bioaerosols and dusts these include:

- Ensuring that the optimum moisture content is maintained during the composting process;
- Ensuring that the compost is turned regularly;
- Maintaining good housekeeping; and
- Erecting bunding/planting trees around the perimeter of the site.

Most surveys have led to the following general assessment:

- Activities run at a composting facility expose workers to a certain load of dust particles and aerosols, above all while turning or moving dry, dust-like materials. Fresh food waste, for instance, is too wet to release dust;
- The risk is similar to those run by workers at earth-moving companies;
- Health risk management should include a prevention program for workers (as they currently do in many Member States), including:
  - ✓ Individual protection devices (dust masks to be worn during most dusty activities); and
  - ✓ Periodic health assessment;
- Nearby dwellers are not so exposed in most situations. Distances in the order of 200-300 metres are frequently enough for bioaerosols to fall down to background concentration of airborne micro organisms;
- Running operations in enclosed buildings sharply reduces the occurrence of risks in external spaces

We have not quantified any effect of micro-organisms on human health. Where these are problematic, we suspect that they have greatest effect on workers and in closed systems, and that their effect is relatively confined. The Table below outlines figures for micro-organisms at one of the sites in the UK Environment Agency investigation.

**Table 38: Presence of Micro-organisms at Compost Site**

Microorganisms Species	Shredding cfm/m <sup>3</sup>	Turning cfm/m <sup>3</sup>	Screening cfm/m <sup>3</sup>
Fungi	0.000004	0.0000001	0.000005

<sup>140</sup> E. J. Gilbert (1998) *Health and Safety at Composting Sites: A Guidance Note for Site Managers*, The Composting Association.

Aspergillus fumigatus	0.000001	0.0000001	0.000001
Total bacteria	0.0000001	0.0000001	0.000005
Streptococci	0.00005	0.0006	0.0001
Enterobacteria	0.00001	0.0001	0.001
Total Actinomycetes	0.000001	0.0000001	0.000005

Source: Environment Agency (2000) *Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion*, Bristol: Environment Agency

### A.6.13.5 Composting and Other Human Pathogens

Pathogens can be present in the feedstock or be introduced from the environment during the composting process, following which they can increase in number until levels are reached that are capable of causing harm. There are two main pathways through which pathogens from compost can cause problems to humans and animals. One is through inhalation of dust and minute particles known as aerosols, the other is by ingestion.

The process of composting itself can help to sanitise the final material through both pasteurisation (see **Error! Reference source not found.** for the time/temperature combinations suited to ensure sanitation) and the loss of starting hospitable biochemical features.

Compost applied directly to grazing crops should be free of unacceptable levels of pathogens. For application to salads and fruit that may be consumed raw these are also of concern. In these cases it would be a sensible precaution to have the compost checked for certain organisms thought to indicate the overall pathogen content. For these purposes it is often recommended to include one or more of the following:<sup>141</sup>

- *Salmonella* spp
- faecal *Streptococcus*
- faecal coliforms
- total coliforms
- viable nematode eggs

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<sup>141</sup> C. Johansson, E. Kron & S-E Svensson (1997). *Compost Quality and Potential for Use* AFR Report 154, Swedish Environmental Protection Agency.

Table 33 Temperature/Time Regimes Known To Be Effective During Composting Against A Range Of Pathogens Of Importance To Humans <sup>142</sup>

Disease	Organism	Lethal conditions
<b>Non spore-forming</b>		
Brucellosis	<i>Brucella abortus</i>	10 min: 60 °C
Cholera	<i>Vibrio cholerae</i>	15 min: 55 °C
Contagious abortion	<i>Vibrio fetus</i>	5 min: 56 °C
Diphtheria	<i>Corynebacterium</i>	10 min: 58 °C
Dysentery	<i>Shigella</i> spp	60 min: 55 °C
Food poisoning	<i>Salmonella</i> spp	20 min: 60 °C
Leptospirosis (Weil's)	<i>Leptospira</i> spp	10 min: 50 °C
Plague	<i>Yersinia pestis</i>	5 min: 55 °C
Staphylococcal infections	Staphylococci	30 min: 60 °C
Streptococcal infections	Streptococci	30 min: 55 °C
Tuberculosis	<i>Mycobacterium</i>	10 min: 60 °C
Typhoid fever	<i>Salmonella typhi</i>	20 min: 60 °C
<b>Spore-forming bacteria</b>		
Anthrax	<i>Bacillus anthracis</i>	10 min: 100 °C
Botulism	<i>Clostridium botulinum</i>	5 hr: 100 °C/5 min: 120
Gas gangrene	<i>Clostridium</i> spp	6 min: 105 °C
Tetanus	<i>Clostridium tetani</i>	3-25 min: 105 °C
<b>Viruses</b>		
Foot and mouth disease		30 min: 56 °C
Scrapie		withstands 2 hr 100 °C
Serum hepatitis		10 hr: 60 °C
Swine fever		1 hr: 78 °C
<b>Intestinal worms</b>		
Round worm	<i>Ascaris limbridoides</i>	1 hr: 55 °C
Tape worm	<i>Taenia saginata</i>	few minutes: 55 °C

There are therefore several points to be considered when applying compost to agricultural land: reduction of infection brought in with feedstock; process management to eliminate any presence in feedstock or coming from the environment; and protection of users against inherent pathogens. Provided all precautions are taken, the risk to humans or animals is believed to be minimal.

<sup>142</sup> CEN/TC 223 WG2 (1995)

### A.6.13.6 Plant Pathogens / Diseases

Of paramount importance to compost users, particularly agriculture and horticulture, is the potential for introduction of plant disease in compost. Applying infected material to the soil without any prior treatment would have consequences for susceptible crops before such times as the plant pathogens can be broken down in the soil. As with human pathogens, process controls during composting can reduce the incidence of disease in the finished material.

Potentially infected material may be included in peelings and other kitchen and garden waste, or in crop residues. It is therefore unrealistic to expect to eliminate plant disease from feedstock, especially when raw materials are collected from civic amenity sites or household collections. As illustrated in **Error! Reference source not found.**, it is possible to destroy many disease causing pathogens through controlled management of the composting process. Some viruses and spore forming fungi such as *Fusarium*, *Phytophthora* and *Pythium* are hard to inactivate and can then persist in the soil for years.<sup>143</sup> Nonetheless, surveys have been focusing increasingly on the “disease suppressive” features of composted products, i.e. the capability of a biologically activated material to hinder pathogens through competition and antibiosis (see also section 6.0). Such features are increasingly being exploited also on a commercial level.

### A.6.13.7 Other Compost Related Problems

#### Vermin

All organic waste is attractive to vermin, i.e. rats, mice, flies, and birds. However, good management practices can reduce their occurrence at a composting facility.

#### Flies

There is no disputing that flies are attracted to organic material, however they should not become a problem at a well managed composting facility. It should be noted that there will always be flies and other insects at a composting facility, indeed they are an intrinsic part of the process. However they should not be present in such numbers as to cause a problem. There are several steps that can be taken to ameliorate the problem of flies if they do occur.

Probably the most effective method of controlling flies at a composting facility is to ensure that the windrows are turned regularly, no less than once a week during the first six weeks of the process (in some instances it may be necessary to turn more frequently i.e. 2-3 times a week). Frequent turning serves a number of purposes. It disrupts the flies, and should destroy larvae and eggs. It also ensures that the outside of the windrow is exposed to high core temperatures, which will also destroy fly eggs and larvae.

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<sup>143</sup> C. Johansson, E. Kron & S-E Svensson (1997) *Compost Quality and Potential for Use* AFR Report 154, Swedish Environmental Protection Agency.

Table 34 Inactivation Regimes For A Range Of Pathogens Of Importance To Growers

Pathogen	Inactivation regime
<b>Fungus with resting spores</b>	
<i>Phytophthora infestans</i> (potato blight)	2 - 3 wks: 47 - 65 °C
<i>Phytophthora cryptogea</i>	2 - 3 wks: 64 - 70 °C
<i>Fusarium oxysporum</i>	30 minutes: 57.5 - 65 °C
<b>Fungus with sclerotia</b>	
<i>Sclerotinia sclerotiorum</i> (white mould)	2 - 3 wks: 64 - 70 °C
<b>Bacteria</b>	
<i>Pseudomonas syringae pv phaseolicola</i> (halo blight of bean)	4 days: 35 °C
<i>Erwinia amylovora</i> (fire blight of pome fruits)	7 days: 40 °C under optimum conditions
<b>Viruses</b>	
Tobacco necrosis virus (TNV)	72 - 96 hrs: 55 °C/24 - 48 hrs - 70 °C
Tobacco mosaic virus (TMV)	survived 6 wks: 50 - 75 °C
Tomato mosaic virus (ToMV)	inactivated by biological decomposition
Tobacco virus (TRV)	survived 68 °C
<b>Nematodes</b>	
<i>Heterodera rostochiensis</i>	50 - 55 °C
<i>Aphelenchoides fragariae</i>	1 hr: 50 °C/4 hrs - 44 °C
<i>Ditylenchus dipsaci</i>	1 hr: 50 °C/4 hrs - 44 °C
<i>Meloidogyne hapla</i>	1 hr: 50 °C/4 hrs - 44 °C

Source: Johansson, C., E. Kron & S-E Svensson (1997). *Compost Quality and Potential for Use AFR Report 154*, Swedish Environmental Protection Agency.

Fresh feedstock should also be shredded and/or mixed as soon as possible as the action of shredding and mixing will destroy some larvae. The structural changes to the material will also make it less desirable to flies.

In static processes, i.e. where no turning is planned, a layer of mature compost (approximately 15 cm's) can be used to cover the youngest windrows, this will act as a biofilter which will help to reduce odour it will also prevent flies from getting to the fresher more desirable material underneath.

## Birds

Birds are attracted to flies and organic material at compost sites, although problems with larger birds such as seagulls are rarely an issue and tend to be associated with composting facilities located at landfill sites. Smaller birds are occasionally found at composting sites but usually not in sufficient numbers to be considered a problem. Managing the process and ensuring that the site is kept clean and free from debris should help to prevent the conditions which attract both flies and birds from occurring.

## Rats

Rats can be attracted to the organic matter at a composting facility and also the storage piles of finished product as these make an ideal location to build a nest. Good site management can help to control rat populations.

### A.6.13.8 Disamenity

We are not aware of any study looking at the disamenity associated with compost plants. There is, therefore, no basis for quantification. Certainly, these would be expected to deviate more significantly from zero where one or more of the following are true:

- The composting process is poorly managed (and odours are prevalent);
- The compost plant accepts inappropriate materials (i.e. those which are likely to give rise to problems in the context of the process technology being used);
- The plant is of a significant scale, so that visual intrusion becomes an issue, as do transport movements (though this may imply some double counting where transport externalities are considered); and
- The composting process occurs in close proximity to housing.

All of these increase the potential for significant disamenity.

### A.6.13.9 Heavy Metal Concentrations

Concern has been expressed regarding the danger of heavy metals (potentially toxic elements) contaminating soils as a result of compost application. Once they have been added to soils, crop removal and other loss mechanisms are relatively insignificant.<sup>144</sup> Heavy metals may be directly toxic to plants or passed through the food chain to humans. The uptake of heavy metals by plants depends upon a number of factors including the nature of the crop, the pH of the soil and other specific factors relating to the metals concerned. Although heavy metals can be easily measured in composts there is no consensus on how or what standards should be set. Indeed there is considerable variation both across Europe and within individual countries (see section 3.6 of the Main Report)

It is not only the standards that vary; there seems to be no consensus among researchers concerning the accumulation of heavy metals in soils, the uptake of

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<sup>144</sup> D. E. Baker and M. C. Amacher (1982) Nickel, Copper, Zinc and Cadmium, Page, A.L Miller R.H & Keeney, D.R (eds.) (1982) Methods of Soil Analysis, *Agronomy* 9(2): pp. 323-336.

heavy metals by plants or the consequences of heavy metals once in the food chain (perhaps unsurprisingly given the range of issues affecting uptake).

One study investigated the differences in heavy metal uptake between various vegetables and concluded that lettuce, spinach and celery took up heavy metals more readily than many others.<sup>145</sup> After a 10 to 15 year period of application the study found the heavy metal content of the soil, especially of cadmium, had increased. Clearly, however, this depends upon the loading implied by the compost application. Furthermore, overall, he still regarded compost application as having a positive effect on the soil in terms of structural improvements and nutritional benefits.

Another study proposed that even though heavy metals may be added to the soil in the composts they were not always in a form available to plants.<sup>146</sup> The study compared the leachable fraction of heavy metals in different composted municipal solid wastes, showing that although these composts may contain high concentrations of elements only small percentages were leachable (with the exception of nickel). From this they concluded that only those leachable amounts were of concern. Using the SCE (sequential chemical extraction) and SRC (synthetic acid-rainwater cascade extraction) protocols they illustrated that the total elemental contents did not correlate well to the leachable fraction, thus demonstrating that the chemical form rather than the total content of an element is more important in determining its availability for plant uptake or leachability into groundwater.

This opinion is disputed by many others, as it is argued that the chemical form depends much more on the conditions of the site (redox potential, in turn being affected by the depth of the groundwater and climatic features, soil pH, etc.) than on the form of the element in the compost to be used. Soil conditions themselves are subject to change with time (not least through application of organic matter), and this suggests the need for precaution when considering the accumulation of heavy metals in the soil, with no difference between different chemical forms.

It would seem that for each heavy metal, each vegetable and each soil type combination uptake of heavy metals is likely to be different. Although it is generally believed that composted materials 'lock' heavy metals into their structure some researchers have reported different conclusions. There is clearly much work to be carried out on the contamination of soils and food by heavy metals from composts. Further field studies are needed to test the heavy metals concentrations in a variety of vegetables grown in compost amended soils.

It is worth stressing that heavy metal loadings from composted materials will be lower where waste materials are separated at source. In addition, it is also worth reflecting on the fact that conventional fertilisers also contain heavy metal fractions. In particular, phosphate rock deposits contain varying levels of cadmium for which decadmiation processes exist, but these are not 100% effective (see **Error! Reference source not found.**).

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<sup>145</sup> H. Poletschny (1992) *Kompostverwertung im Landbau aus der Sicht des Verbandes Deutscher Landwirtschaftlicher Untersuchungs-und Forschungsanstalten (VDLUFA)*. Kongressband.

<sup>146</sup> S. E. Tisdell and V. T. Breslin (1995) Characterisation and Leaching of Elements from Municipal Solid Waste Compost, *Journal of Environmental Quality* 24:827 -833.

Table 35 Cadmium Contents Of Main Commercial Phosphate Rocks According To Different Sources.

Origin	Cadmium content (mg per kg P <sub>2</sub> O <sub>5</sub> )	
	(1)	(2)
<b>Igneous</b>		
Kola (Russia)	< 13	0.25
Pharlaborwa (South Africa)	< 13	0.38
<b>Sedimentary</b>		
Florida (USA)	23	24
Jordan	< 30	18
Khouribga (Morocco)	46	55
Syria	52	22
Algeria	60	
Egypt	74	
Bu-Cra (Morocco)	100	97
Nahal Zin (Israel)	100	61
Youssofia (Morocco)	121	120
Gafsa (Tunisia)	137	173
Togo	162	147
North Carolina (USA)	166	120
Taiba (Senegal)	203	221
Nauru	243	

Sources: (1) Davister, A. (1996). *Studies and Research on Processes for the Elimination of Cadmium from Phosphoric Acid*. In: OECD (1996) *Fertilizers as a Source of Cadmium*. Organisation for Economic Co-operation and Development, Paris, p. 21-30; (2) Demandt, I. (1999). *The World Phosphate Fertilizer Industry. Research Project 'Environmental Regulation, Globalization of Production and Technological Change', Background Report No. 10*. The United Nations University, Institute for New Technologies, Maastricht, July 1999.

The European approach, as mirrored in the regulations adopted by most Member States, tends to be to seek to preserve soil quality, preventing a rapid accumulation of heavy metals. Composted materials with higher concentrations of heavy metals are deemed to be applied in most Member States under controlled conditions and after licensing. The American approach on the other hand often tends to determine through risk-assessment the 'no-risk load' of each heavy metal carried onto the soil by compost. This has given raise to sharply different limit values, with European ones much tighter than American ones.

#### A.6.13.10 Trace Elements and Compost<sup>147</sup>

Many metals are naturally present in minute amounts in the soil and water. These 'trace' elements occur as a result of the weathering of rocks. They can be leached

<sup>147</sup> See P. B. Woodbury (1998) *Municipal Solid Waste Composting: Potential Effects of Heavy Metals in Municipal Solid Waste Composts on Plants and the Environment*, BOYCE Thompson Institute for Plant Research at Cornell University.

into groundwater or surface water and taken up by plants, released as gases into the atmosphere or bound semi-permanently by soil components such as clay or organic matter.

Metals can arise in the waste stream from a variety of sources such as, batteries, consumer electronics, ceramics, house dust, paint chips, plastics and used motor oils. It is inevitable that some contamination of the compost feedstock by these materials will occur, albeit in small quantities. On the other hand, source separation sharply reduces the occurrence of such contaminants.

In small amounts many of these trace elements such as: boron, zinc, copper and nickel are vital for plant growth. However, in large amounts they may inhibit plant growth. Trace elements such as: arsenic, cadmium, lead and mercury are of concern primarily due to their potential to harm soil organisms and animals and humans who may eat contaminated plants or soil. Soil properties such as pH and cation exchange capacity also effect how plants react when metals are present.

#### Elements Of Concern Primarily To Plant Health

Excess boron can decrease plant growth, however like other trace elements, boron is more likely to be deficient in soils than to cause toxicity. Most boron in compost is water soluble and leaching of the compost prior to application may eliminate the problem of toxicity.<sup>148</sup>

#### Elements Of Primary Concern To Animal And Human Health

Cadmium, lead and mercury can be harmful to animals and humans at relatively low concentrations. The degree of uptake of cadmium is, to a certain extent, species specific. Mushrooms, spinach and other leafy vegetables readily uptake cadmium. Studies suggest that plants take up very little lead from soils and even with substantial additions of compost the increased uptake of lead by crops is minimal. Conversely, there is evidence to suggest that the application of composts can reduce the uptake of lead by plants, as the organic matter in compost binds the lead and decreases its availability to plants. The concentration of mercury in compost is usually low and therefore there is little significant uptake by plants.

#### Elements Of Minor Concern

Arsenic, chromium, copper, nickel and zinc are unlikely to cause problems for plant, animal or human health as they are not usually found in high enough concentrations or are not readily taken up by plants. Arsenic is not readily taken up by plants and therefore is unlikely to pose a problem. Chromium is usually only present in trace amounts in compost, in addition it is usually present in a form that is not readily taken up by plants.

Long term studies have shown that even with substantial applications of compost, there is very little increase in the copper content of plants, the organic matter in compost also serves to bind the copper and therefore reduce its availability to plants. Nickel is toxic to plants. However, it tends to be present in very small quantities. Therefore it is unlikely to restrict growth. This also tends to be the case with zinc, and some zinc may be beneficial to Zn-lacking soils and crops.

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<sup>148</sup> Leachate should be captured and treated and not allowed to drain into soil/watercourses.

### Long-Term Concerns

As organic matter decomposes the concentration of metals in compost (and therefore in the soil it is applied to) increases. The available data suggests that if large amounts of composts are applied to agricultural soils, half of the organic matter may decompose within 10 –20 years. Metal concentrations in the soil are unlikely to exceed the concentration present in the original compost unless very large amounts of compost high in organic matter are applied. Over a period of time, metals generally become less available to plants and other organisms, providing there are not changes in soil pH (pH decreases) or there are prolonged periods of flooding.

### Potential Benefits Of Trace Elements In Compost

Soils that have been farmed intensively for many years may be deficient in elements such as boron, zinc and copper, the application of compost could mitigate such deficiencies. It is also possible that the application of compost may reduce harm to plants by ‘tying-up’ trace pollutants and potentially toxic organic compounds.

## A.7.0 HOME COMPOSTING

### A.7.1 Defra Health Effects Study

The Defra Health Effects study has no information concerning emissions from home composting.

### A.7.2 Air Emissions from Home Composting

Only one study of relevance has been carried out in the UK.<sup>149</sup> This study looked at emissions from a number of home composting sites. The level of emissions was found to be very low for those gases for which measurements were sought.

It is not possible to translate the estimates of emissions into emissions 'per tonne' of home composted material. The emphasis was on the measurement of concentrations rather than emissions per tonne.

The rate of emissions from home composting facilities is likely to be much slower than at larger facilities. Furthermore, the ratio of the surface area to the volume is likely to be much greater. These two factors suggest that it may be the case that emissions from home compost heaps are more likely to be subject to 'biofilter-type' abatement as the outer layers act in this way to abate, at least to some extent, the emissions from the degrading material.

It may be that this explains the very low level of emissions of methane recorded in the experiment, which was carried out at 12 volunteer home compost sites. The authors state:

*The analysis for methane has resulted in only two readings at one of the sites (out of 112 readings) being above the detection limit of the equipment used (0.1 %) and further tests are underway with more sensitive equipment to ascertain levels more precisely. The CO<sub>2</sub> readings have measured values 32 times out of the 112 readings. It is likely that the methane emitted is very low and does not pose any significant environmental contribution to global climate change, but results from the more accurate equipment may shed some light on this.*

It seems possible that the outer layers of the composting mass act as an oxidation layer, so reducing emissions to the atmosphere. The low level of CO<sub>2</sub> detected is probably simply testimony to the lower rate of activity in many compost heaps, as well as limitations of the detection equipment used.

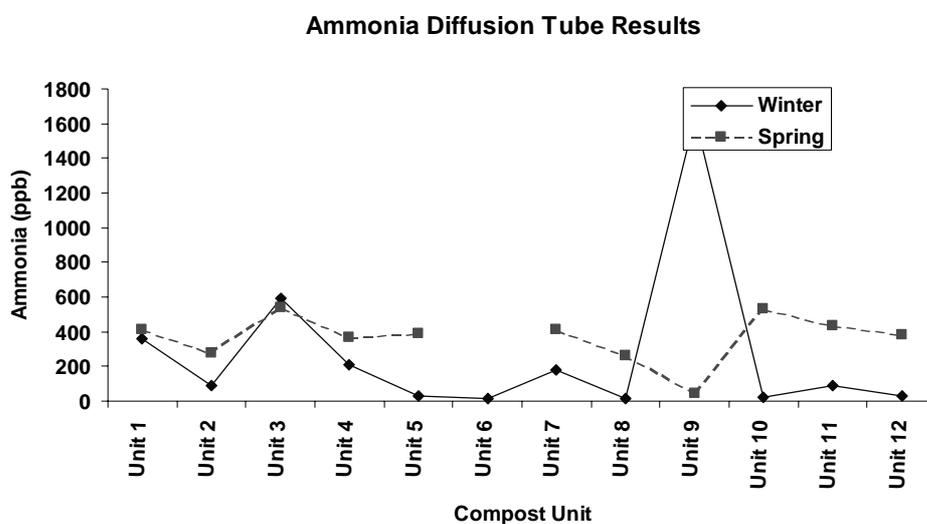
Given the significance (for the analysis) of ammonia emissions, the emissions of this gas are of particular significance, and the degree to which these escape into the atmosphere, and subsequently give rise to *additional* secondary particulate

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<sup>149</sup> P. A. Wheeler and J. Parfitt (u.d.) *Life Cycle Assessment of Home Composting*, Environment Agency R&D Report CLO329.

formation, is of considerable interest. The emissions detected from the experiments are shown in **Error! Reference source not found.**

Figure 20: Concentrations of Ammonia in Airspace Above Home Compost Heaps



Source: P. A. Wheeler and J. Parfitt (u.d.) *Life Cycle Assessment of Home Composting*, Environment Agency R&D Report CLO329.

These show low level concentrations in the space above the composting material. One heap – where chickens were kept by the householder – shows elevated levels of emissions relative to the other units. This is most likely a reflection of the presence of chicken manure – a rich source of nitrogen - in the feedstock (and this may lend some support to the approach of using a transfer factor for the percentage of N in the feedstock being emitted as ammonia, as was discussed in the composting module).

It is not entirely clear:

- How much ammonia is emitted per unit weight of material home composted (and the relationship to the nitrogen content of the input feedstock); and
- what happens to the ammonia once released into the headspace above the heap. It is possible that the fate of the material varies depending upon the actual container used. For example, ammonia is quite soluble and so in plastic container systems, where moisture is often retained as condensate, some ammonia may be dissolved in the moisture without ever escaping into the atmosphere. This is less likely to occur in heaps where the heap is more open to the atmosphere, though even here, there may be means of containment / keeping additional moisture off the heap which affect the extent to which ammonia escaping from the composting material actually escapes the composting site.

Hence, there is some uncertainty about the level of ammonia emissions and they way they are likely to be controlled.

The study also looked at emissions of VOCs. The authors state the following:

*In the measurement made here ketones were not detected in any significant quantities. The most significant emission was limonene, which is presumably from citrus fruit peel, ethanol has been also detected at up to 60 ppb from the fermentation of the organic wastes. Overall this suggests a somewhat different composting process to that which occurs at commercial composting plants, as a low temperature anaerobic process generates ethanol. A wide range of other compounds have been detected at very low levels, such as benzene and toluene that are presumably from the general environment rather than the composting process.*

The point being made concerning the nature of the process is an interesting one. There is likely to be a range of micro organisms acting in different ways in different compost heaps. Consequently, home composting emissions may be particularly hard to quantify with accuracy. Evidently, management of the process (including, critically, the mix of materials in the feedstock) is likely to have an impact upon emissions.

### A.7.3 This Study

The impacts of different emissions (other than GHGs) are likely to be related to their contribution to increasing concentrations of the pollutants to which receptors are exposed. What little evidence there is available appears to suggest that such contributions will not be significant. Consequently, to the extent that home composting increases the emissions of various pollutants, the fact that these are in relatively small amounts, and are likely to be dispersed, suggests that the contributions to impacts on human health are unlikely to be significant. These are likely to be greatest for those engaged in the process themselves, and it is not clear that they would pose much by way of threat in this respect either, though (as with larger processes) turning heaps is likely to give rise to some bioaerosol emissions. The impact of these seems likely to be localised, and basic precautions seem likely to be adequate to ensure that participants in home composting avoid major problems associated with their release.

Consequently, for the emissions of home composting we have assumed:

- CO<sub>2</sub> emissions as for larger scale processes. The lack of detectable measurements in the study discussed above may reflect the fact that emissions occur over a longer period of time in most home composting processes;
- Methane emissions as for large scale processes but with an assumed oxidation rate of 60% through the biomass;
- Ammonia emissions calculated as previously, but with the mass of material assumed to remove 60% of the ammonia;
- VOCs as for larger processes; and
- N<sub>2</sub>O as for larger processes.

In terms of offsets, we have assumed that similar benefits apply in terms of water use, but none of the agriculture-related benefits apply. Instead, the compost is assumed to displace peat-based products. This is unlikely to be strictly true since in some cases, displacement may be of mulch, or other soil improvers.

## A.8.0 LANDFILL

### A.8.1 DefraHealth Effects Report

The most recent UK data for emissions from landfill was presented in the report on Health Effects of Waste Management.<sup>150</sup> That report looked at emissions from landfills and in particular at atmospheric emissions and the fate of leachate associated with the operation of the landfill.

One issue with the study is that, like many other studies, it makes no attempt to understand the profile of the emissions across time, though it recognises the fact that emissions will occur over an extended period of time. In that sense, it resembles many life cycle studies which typically use arbitrary cut-off periods to ‘truncate’ the quantity of emissions at some point in time (which is usually a period of 100 years).

For the purpose of cost-benefit analyses, this is not an adequate approach. It is not appropriate to assume, effectively, a zero discount rate up to a given point in time and, implicitly, an infinite one for the period thereafter. As soon as one considers non-zero discount rates, the time profile of emissions assumes considerable significance. This is why the conventional life-cycle assumption – that biogenic carbon dioxide emissions can be ignored for the sake of the analysis – is inadequate for an economic analysis.

In the report from HM C&E assessing the external costs of landfills, a footnote acknowledges this, although the study itself simply made use of the figures from the Health Effects report:

*Emissions from Landfill will typically take place over a number of years (and any emissions taking place at present will be as a result of wastes land filled over previous years). The analysis does not model the time profile of, or discount, future landfill emissions.*

The significance of the assumption has been explored elsewhere by us.<sup>151</sup>

As regards the Health Effects report, a laudable attempt was made to understand the different emissions associated with fugitive emissions, flared gases, and gases used for the purpose of energy generation. The approach then undertaken was to multiply

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<sup>150</sup> Enviro, University of Birmingham, RPA Ltd., Open University and Maggie Thurgood (2004) *Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes*, Final Report to Defra, March 2004.

<sup>151</sup> See Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC Research & Consulting (2002) *Economic Analysis of Options for Managing Biodegradable Municipal Waste*, Final Report to the European Commission; D. Hogg (2003) External Costs and Benefits of Composting and Anaerobic Digestion, in L. Marmo and H. Langenkamp (eds) *Biological Treatment of Biodegradable Waste: Technical Aspects*, European Commission Joint Research Centre; D. Hogg (2004) Costs and Benefits of Bioprocesses in Waste Management, in P. Lens, B. Hamelers, H. Hoitink and W. Bidlingmaier (eds.) (2004) *Resource Recovery and Reuse in Organic Solid Waste Management*, pp.95-121; D. Hogg (2005) Costs and Benefits of Residual Waste Options, Presentation to the ORBIT/ECN Conference, November 17-18, 2005, Luxembourg.

the concentration of the pollutants in each type of gas by the quantity of gas emitted (related to the process concerned). This is a somewhat awkward approach when applied, as in the report, to the modelling of case studies in which the quantity of any gas emitted per tonne of waste relates back to the relative proportions of gas assumed to be fugitive, flared or used for energy recovery. As a consequence of this, the two case studies represented in the model appear to imply radically different raw gas volumes even though the purpose of the study was (presumably) to model the emissions from similar waste where the biogas is sent for different treatments. What is presented as Case Study 1 has an implied biogas generation of around 181m<sup>3</sup> whilst Case Study 2 has a generation of 128 m<sup>3</sup> (our calculations), relating to carbon contents of 83kg and 56 kg respectively. Both these figures are on the low side of what the study suggests are typical figures for the emissions of biogas per tonne of waste, which it puts at 200m<sup>3</sup> ± 74m<sup>3</sup>, and the lower of the two figures barely falls within this range (so it would appear to be 'extreme' by the standards of the study).

This is not simply a problem of measurement and variability. It arises more from the fact that the nature and quantity of gas escaping as fugitive or being flared or collected varies across the life of the landfill so that 'snapshot pictures' of emissions cannot give a clear representation of what happens in the landfill. There is no way of escaping this problem without adopting an approach which allocates emissions to the time profile of the release of the landfill gases.

That having been said, this is an extraordinarily difficult process to model and there remains considerable disagreement concerning the possible rates of capture of landfill gas over the lifetime of a landfill (as opposed to 'in a given year'). Recent controversy in the United States over this issue (in the context of debates concerning Joint Implementation under the Kyoto Protocol) has led one credible source to suggest that lifetime captures of biogas may be as low as 19%.<sup>152</sup>

The issue of gas capture efficiencies is one which critically influences the performance of landfills in any external cost assessment. Anderson suggests that the high rates of capture posited by bodies such as the USEPA are based upon high estimates of instantaneous captures of gas rather than of lifetime efficiencies. He argues that '*only 32.1% of a landfill's lifetime gas emissions occur when there is a functioning gas collection system in place*', and that as a consequence, lifetime methane captures are likely to be closer to 19%. This contrasts with estimates in various documents where gas capture rates are assumed to be much higher. For example, the USEPA itself notes that newer closed landfills with impermeable membranes and active gas controls may have an efficiency of 90% based upon '*the consistent reporting of near zero surface emission measurements*'.<sup>153</sup> In some respects, this may support the contention of critics that what are being reported are instantaneous gas capture rates rather than lifetime figures. In the EU context, the work by Smith et al suggests:<sup>154</sup>

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<sup>152</sup> P. Anderson (2005) *The Landfill Gas Recovery Hoax*, Abstract for 2005 National Green Power Marketing Conference.

<sup>153</sup> USEPA (2004) *Direct Emissions from Municipal Solid Waste Landfilling*, Climate Leaders Greenhouse Gas Inventory Protocol – Core Module Guidance, October 2004

<sup>154</sup> A. Smith, K. Brown, S. Ogilvie, K. Rushton and J. Bates (2001) *Waste Management Options and Climate Change*, Final Report to DG Environment, European Commission.

*'Estimates vary as to the efficiency with which gas collection systems in landfills gather the methane formed in waste. Gas collection networks installed around the periphery of a site to prevent landfill gas migrating into neighbouring land have very limited efficiency in reducing emissions from across the site. On the other hand, modern, well-designed systems with gas wells installed throughout a landfill site may collect between 70 – 90% of the gas that is formed. We have no data on this parameter so our weighted average is derived from estimates of the likely efficiency in each EU country based on our knowledge of landfill practices in the EU.'*

*We also include an estimate of the proportion of gas which is vented without combustion on sites with gas collection. This represents the gas collected during the first few years of site operation, and during the later years when the peak of methane production is over. During this period the concentration of methane in the gas collected is too low for combustion without the addition of another fuel such as natural gas, and therefore the gas collected is often vented without combustion. Although there is little data on this parameter, we estimate a figure of 10%.*

The study also looked at three cases (to explore variation). The parameters used to characterise these are shown in **Error! Reference source not found.**

**Table 36: Landfill Site Characteristics Used by Brown et al**

	Case 1	Case 2	Case 3
	Limited collection	Best practice	Restoration layer
Gas collection efficiency	20%	80%	80%
Methane oxidised	10%	10%	90%
Landfill gas used	0%	60%	60%
Power gen efficiency	0%	30%	30%

Source: A. Smith, K. Brown, S. Ogilvie, K. Rushton and J. Bates (2001) *Waste Management Options and Climate Change, Final Report to DG Environment, European Commission.*

**Table 37: Assumptions from Brown et al Concerning Average Performance of EU Landfills**

Parameter	Value
Fraction of Landfill Carbon Decaying to Methane	50%
Percentage of Waste in Sites with Gas Control	68%
Gas Collection Efficiency	54%
Percentage of Uncollected Methane Oxidised	10%
Percentage of Collected LFG Vented Without Combustion	10%
Percentage of Collected LFG Used for Energy	60%
Percentage of Collected LFG Flared	30%
LFG Electricity Generation Efficiency	30%

Source: A. Smith, K. Brown, S. Ogilvie, K. Rushton and J. Bates (2001) *Waste Management Options and Climate Change, Final Report to DG Environment, European Commission.*

## A.8.2 Air Emissions from Landfills

### A.8.2.1 Greenhouse Gases

In this work, we have used a model which is based upon a first order decay model of landfill gas generation. The model is based around one which is recommended for use by countries when reporting their greenhouse gas inventories to the IPCC.

The gases which are emitted in any one year are assumed to be related to the quantity of methane or CO<sub>2</sub> produced, depending upon whether one is considering raw gas or gas once combusted. The model allows for specification of the following parameters:

1. Gas collection rate – reflecting the discussion above, we have used two rates, one representing what is frequently cited in the literature as good practice (70%), the other, reflecting what critics argue is a more realistic figure (30%);
2. Oxidation rate of uncaptured gas – the model also allows for varying estimates of the rate at which methane is oxidised through the landfill cap or through leachate.<sup>155</sup> The USEPA suggests a range between 10% to 25%, with clay soils being at the lower end and top soils at the higher end. A default figure of 10% is proposed. This is the same figure proposed by Brown et al in a study for what was then DETR. We have used the figure of 10%;<sup>156</sup>
3. Quantity of captured gas which is used to generate energy – we have used a default figure of 70% in both cases;
4. Efficiency of engine in generating energy. This is taken to be 35%.

Fugitive emissions and airborne dust from landfill activities during the operational stage of the landfill are not considered here. Some studies have addressed the issue of emissions from the landfill construction and engineering phases and it is frequently argued that these tend to be small relative to the total emissions for modern large-scale landfills.<sup>157</sup> On the other hand, they do occur, by definition, at an earlier time, so where one is discounting emissions (rather than adopting a perspective which is wholly insensitive to time, as in traditional life-cycle studies), they would assume greater importance from an economic perspective.

### A.8.2.2 First Order Decay Model

From the landfilling of a tonne of residual municipal waste, emissions will occur over time as the material degrades. Different fractions of waste will degrade at different

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<sup>155</sup> Note that critics such as Anderson do not state the extent to which, at the lower capture rates for methane, they believe the uncaptured methane might be converted to carbon dioxide.

<sup>156</sup> K Brown et al (1999). *Methane Emissions from UK Landfills*. A report for the UK Department of the Environment, Transport and the Regions.

<sup>157</sup> Gregory, R. G. and A. J. Revans (2000) Part One, in *Environment Agency (2000) Life Cycle Inventory Development for Waste Management Operations: Landfill*, Project Record P1/392/3, Bristol: Environment Agency.

rates. Typically, this has been modelled using first order decay functions with different materials being characterised by different time constants for their rates of decay. The equation used in this analysis is, therefore:

$$Emissions = \sum DOC * FCD * e^{-kt}$$

Where DOC is the degradable organic carbon fraction, FCD is the fraction of dissimilable organic carbon, and k is the material specific time constant driving the decay kinetics.

In this work, we have looked at the decay of materials and used the following parameters to characterise the decay functions (see **Error! Reference source not found.**). These are based on work undertaken by the IPCC and by AEA Technology for the European Commission.<sup>158</sup>

**Table 38 Parameters in The First-Order Decay Functions Used In Modelling Landfill Gas Emissions**

Model Factors	k	Degradable Organic Carbon	Fraction of Organic Carbon Dissimilated
Paper	0.03	40%	35%
Textiles	0.03	20%	30%
Miscellaneous combustibles	0.03	28%	35%
Kitchen waste	0.20	15%	75%
Garden waste	0.10	22%	50%
Average putrescibles		19%	64%
Fines	0.06	9%	60%
Wood	0.03	28%	35%

*Source: Figures were based upon a review of IPCC (u.d.) Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (available at IPCC website); A. Smith, K. Brown, S. Ogilvie, K. Rushton and J. Bates (2001) Waste Management Options and Climate Change, Final Report to the European Commission, DG Environment, July 2001. For the most part, AEA figures were used except for the degradable organic carbon fraction of paper (IPPC value used). For kitchen waste and garden waste, the degradable organic carbon fractions were adjusted downwards to reflect the estimated carbon content of these wastes. For garden waste, this may vary with the seasons.*

These parameters have been validated to some extent through assessing the implied methane emissions from the materials and cross-checking against work undertaken

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<sup>158</sup> IPCC (u.d.) Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (available at IPCC website); A. Smith, K. Brown, S. Ogilvie, K. Rushton and J. Bates (2001) Waste Management Options and Climate Change, Final Report to the European Commission, DG Environment, July 2001.

in the United States and by the Environment Agency.<sup>159</sup> On the basis of the decay functions, emissions of methane are allocated to specific years. This is not strictly accurate since, as is well-known, the production of different gases in landfills varies over time and methanogenic phases do not commence immediately. It is also worth noting that the first order decay model was reviewed recently by Gregory on behalf of Defra and suggestions for changes to the UK's approach to reporting landfill emissions were made on that basis.<sup>160</sup>

Also important here is the assumption concerning landfill gas capture rates. In our model, this figure is constant in each year. In practice, the gas capture rates will vary over time (according to the way in which the landfill is operated). This is another simplifying assumption.

In this study, we have assumed that a specific fraction of each of the biodegradable materials is released as methane. The model then calculates the generation of methane and carbon dioxide and allocates emissions to each year of operation. In doing so, the following (variable) parameters are used:<sup>161</sup>

1. The overall efficiency of gas collection;
2. The percentage of uncaptured methane which is oxidised (to carbon dioxide) in leachate and at the cap;
3. The proportion of captured methane used to generate energy (with the remainder used for flaring).

This approach clearly incurs errors relative to the actual situation. However, in assessing the impact of specific waste fractions and in attempting to incorporate some of the dynamics of landfill gas generation, this is an improvement upon earlier attempts to model the external costs of landfill emissions (which implicitly assume all emissions occur 'at once').<sup>162</sup>

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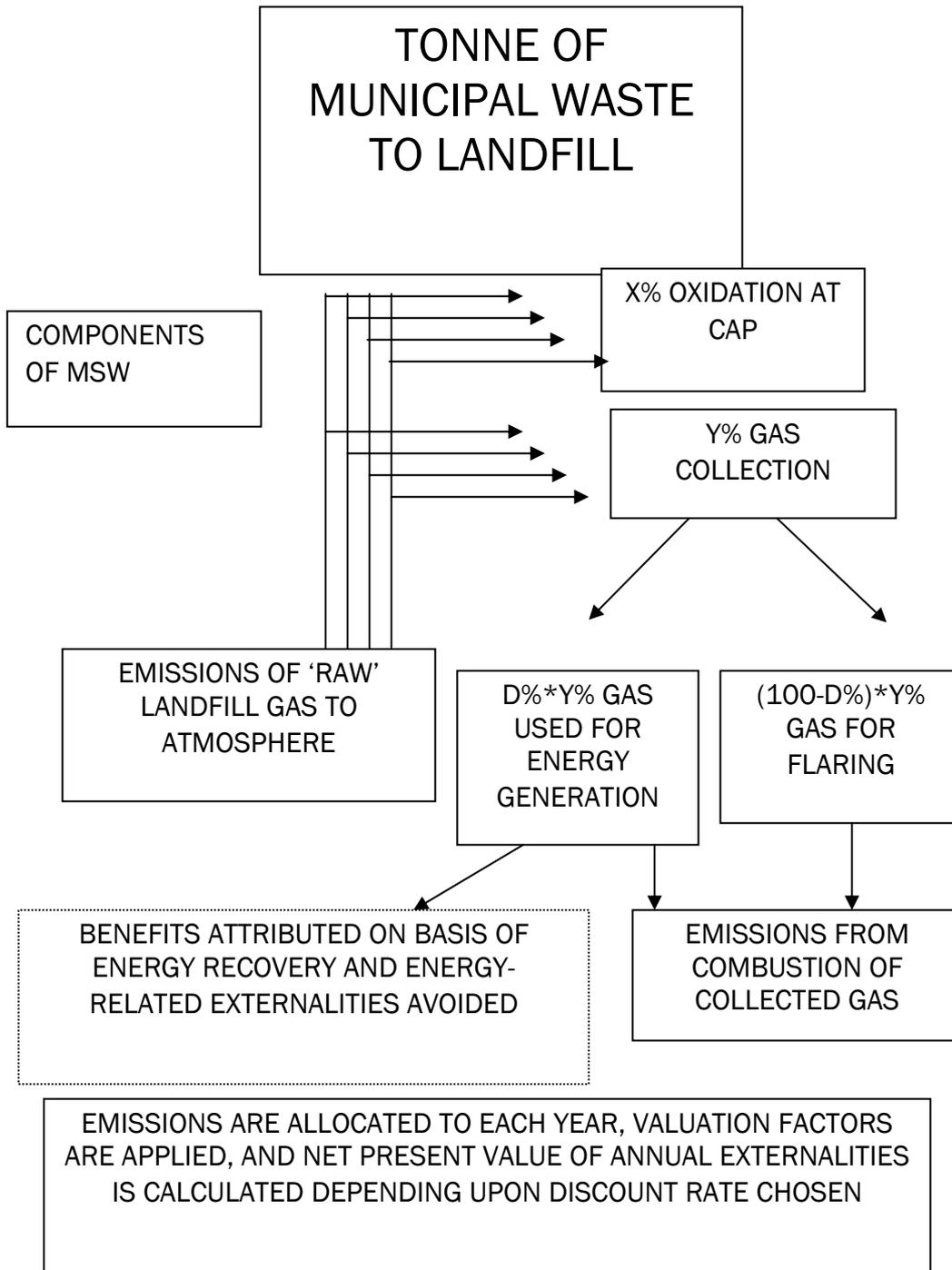
<sup>159</sup> Barlaz, M. (1997) *Biodegradative Analysis of Municipal Solid Waste in Laboratory-scale Landfills*, EPA 600/R-97-071, Washington, DC: USEPA. Gregory, R. G. and A. J. Revans (2000) Part One, in Environment Agency (2000) *Life Cycle Inventory Development for Waste Management Operations: Landfill*, Project Record P1/392/3, Bristol: Environment Agency.

<sup>160</sup> LQM (2003) *Methane Emissions from Landfill Sites in the UK*, Report for Defra, January 2003.

<sup>161</sup> Note again that these parameters are kept constant across time. A more detailed treatment would assess the potential for capture etc. across time in the landfill. This might show, for example, relatively small amounts captured in the earlier and later years.

<sup>162</sup> Recently, a French study has been brought to our attention which highlighted the significance of discounting the emissions. See J. Mery and S. Bayer (2005) *Comparison of External Costs Between Dry Tomb and Bioreactor Landfills: Taking Intergenerational Effects Seriously*, *Waste Management & Research* 23, pp.514-26.

Figure 21 Modelling of Landfill Gas Emissions in the Study



### A.8.2.3 Trace Emissions of Other Gases

Methane emissions to the atmosphere and methane emissions captured are both used to estimate, on a proportional basis, emissions of different trace gases in a given year using the relative composition of gas outlined in **Error! Reference source not found.** below. This is done by normalising the concentrations (by weight) so that:

- Where gas is flared, the emissions of other gases are calculated with reference to the studies by Enviros et al and White et al. This is done by calculating the CO<sub>2</sub> content of flared gas and calculating the emissions of other gases through the quantities relative to CO<sub>2</sub> as specified in the two studies mentioned;
- A similar approach is used to calculate fugitive emissions, but in this case, the other emissions are calculated relative to the calculated quantity of methane emissions; and
- For gas which is emitted from the gas engine, the emissions of other gases are calculated using the quantities estimated in other studies relative to calculated CO<sub>2</sub> emissions.

There are some inconsistencies in this approach, the principal one being that the White et al data make little allowance for changes in the level of oxidation of methane through the cap of the landfill site. Our model incorporates this as a variable. It is important to appreciate here that oxidation may appear not only at the cap (and typical estimates in the literature are 10%), but also in the leachate, so that total oxidation of methane to carbon dioxide may be greater than is sometimes suggested.

### A.8.2.4 Landfill Fires

In addition to these 'biodegradation emissions', we have followed the approach of Finnvenden et al and modelled the occurrence of landfill fires.<sup>163</sup> Landfill fires frequently take place, 0.5 to 1 times per year and landfill. Emission factors for domestic waste landfill fires from Fliedner<sup>164</sup> and Sundqvist<sup>165</sup> are used. These are allocated to the combustible part, which is also suitable in this study. Only the emissions of chlorobenzene, dioxins, PAH, PCB and Hg are documented. Fliedner and Sundqvist assume that only 25% of the emissions from landfill fires will actually become dispersed as air emissions, since the rest will fall back down on to the landfill and this figure is also used here.

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<sup>163</sup> G. Finnvenden, J. Johansson, P. Lind and A. Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, FMS: Stockholm.

<sup>164</sup> A. Fliedner (1999). *Organic Waste Treatment in Biocells. A Computer-based Modeling Approach in the Context of Environmental Systems Analysis*. Master of Science Degree Thesis. Royal Institute of Technology, Stockholm.

<sup>165</sup> J.-O. Sundqvist (1999). *Life Cycle Assessments and solid waste - Guidelines for solid waste treatment and disposal in LCA*. IVL, Swedish Environmental Research Institute, Stockholm.)

Table 39 Air Emissions from Landfilling, Including Landfill Gas and Flare/Engine Exhaust (all Amounts are Given in Mg/Nm<sup>3</sup> Of LFG).

	Fugitive Ratio to CH <sub>4</sub>	Flaring Ratio to CO <sub>2</sub>	Generation Ratio to CO <sub>2</sub>	Source
Methane	1	0.001818	0.005714	Enviros
Carbon dioxide	1.733333	1	1	Enviros
Carbon monoxide	3.03E-05	4.09E-04	4.09E-04	White et al
Hydrogen sulphide	4.66E-04	1.69E-08	1.69E-08	White et al
Hydrogen chloride	2.67E-06	8.64E-05	1.14E-05	Enviros
Hydrogen fluoride	5.33E-07	1.82E-05	1.14E-05	Enviros
Chlorinated HC	8.10E-05	5.10E-06	5.10E-06	Enviros
Dioxins and furans	0	3.36E-13	5.43E-13	Enviros
Total Particulates	0	3.64E-05	0.00002	Enviros
Nitrogen oxides	0	0.000455	0.002571	Enviros
Sulphur dioxide	0	0.000545	0.0002	Enviros
Cadmium	0	0	2.86E-07	Enviros
Chromium	7.12E-08	1.25E-08	1.25E-08	White et al
Lead	2.00E-08	2.49E-09	2.49E-09	White et al
Mercury	1.41E-08	2.49E-09	4.57E-09	Enviros
Zinc	1.68E-07	6.64E-11	6.64E-11	White et al
Nickel	0	0	3.71E-08	Enviros
Arsenic	0	0	4.57E-09	Enviros
Total VOCs	0.000333	7.73E-06	0	Enviros
Non-methane VOCs	0	8.64E-06	8.57E-05	Enviros
1,1-dichloroethane	0.000036	0	0	Enviros
Chloroethane	1.33E-05	0	0	Enviros
Chloroethene	1.47E-05	0	0	Enviros
Chlorobenzene	0.000032	0	0	Enviros
Tetrachloroethene	0.000044	3.64E-08	5.71E-07	Enviros
Poly-chlorinated biphenyls	0	0	0	White et al
Benzene	3.2E-06	0	0	Enviros

Source: Adapted from White, P.R. Franke, M, and Hindle, P (1995) *Integrated Solid Waste Management: A Lifecycle Inventory*, Blackie Academic & Professional, Chapman and Hall and Enviros Aspinwalls (2002) *Comparison of Emissions from Waste Management Options*, Enviros Aspinwalls for NSCA June 2002.

### A.8.3 Emissions of Leachate to Soil and Water

In the externality assessment, we have used the assumptions made in the COWI report, though in practice, these have (as that study notes) significant limitations.<sup>166</sup> COWI indicate the considerable variation in the composition of leachate. Emission

<sup>166</sup> COWI (2000) *A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste*. Final Report to DG Environment, the European Commission, August 2000

factors in grams of pollutant per tonne of MSW landfilled were estimated by COWI. The average leachate concentrations used to determine best estimates for emission factors represent averages over 20-30 years, the period of time for which data is available from landfill sites, and give an indication of the order of magnitude of the emissions. These are shown in **Error! Reference source not found.** and **Error! Reference source not found.** These data have been used to characterise raw leachate emissions.

Bez et al assumed that of leachate, 90% would be captured, and 10% would become diffused pollutants. Nielsen and Hauschild made an assumption that 80% of the leachate from the landfill would be collected with the remaining 20% leaking to aquatic recipients.<sup>167</sup>

A submodule of municipal sewage treatment is used to characterise the environmental impact of the effluent treatment plant. Bez et al used estimated purification efficiencies at the plant to characterise the material. They also used figures to estimate the degree of retention of key pollutants on sewage sludge. They assumed that reduction in BOD yields biological raw sludge which generates digester gas under anaerobic pre-treatment. For each kg BOD, 0.3m<sup>3</sup> digester gas was assumed to be produced, and is used to generate electricity.

Finnvenden et al assumed the collected leachate was transported to a municipal waste water treatment plant. The leachate treatment plant removal factors were taken from Fliedner.<sup>168</sup> Reduction of nitrogen is achieved through conversion to nitrogen gas released to the atmosphere. 50% of the nitrate is converted, the remaining 50% of the nitrate is assumed to be emissions to water. Purifying the leachate demands 0.001 MJ/ kg water and 2 litres of leachate is assumed to arise from 1 kg of waste. This figure is used for all fractions of waste. The energy used for purification is assumed to be electricity and the source of this electricity is coal.

The sludge from the leachate treatment is landfilled. When landfilling the sludge, the leachate purification step is excluded. To get the carbon content of the sludge the COD value is divided by three as a rule of thumb (Björklund 1998) and the carbon is assumed to be easily degradable. The amount of carbon in the sludge is very small compared to the carbon content of the waste handled and therefore this approximation will not be of importance.

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<sup>167</sup> J. Bez, M. Heyde and G. Goldhan (1998) Waste Treatment in Product Specific Life Cycle Inventories, An Approach of Material-Related Modelling, Part II: Sanitary Landfill. *Int. J. LCA* **3**, 100-105; P. Nielsen and M. Hauschild (1998). Product Specific Emissions from Municipal Solid Waste Landfills. *Int. J LCA* **4**:158-168.

<sup>168</sup> A. Fliedner (1999) *Organic Waste Treatment in Biocells. A Computer-based Modelling Approach in the Context of Environmental Systems Analysis*. Master of Science Degree Thesis. Royal Institute of Technology, Stockholm.

Table 40: Emission Factors for Leachate from MSW Landfills (all factors in g/tonne MSW landfilled).

g/tonne	Best estimate*		Low estimate	High estimate
	L1	L2	L2	L2
<b>Organic substances:</b>				
TOC	-	2,177	5	4,350
BOD	-	4,277	3	8,550
COD	-	11,411	21	22,800
Organic N	-	189	2	375
<b>Inorganic macrocomponents:</b>				
Total P	-	2	0.02	3
Chloride	-	349	23	675
Sulphate	-	582	1	1,163
Hydrogen carbonate	-	595	92	1,098
Sodium	-	583	11	1,155
Potassium	-	281	8	555
NH4-N	-	169	8	330
Calcium	-	541	2	1,080
Magnesium	-	1,127	5	2,250
Iron	-	413	0.5	825
Manganese	-	105	0.005	210
Silicate	-	6	1	11
<b>Heavy metals:</b>				
As	-	0.1	0.002	0.2
Cd	-	0.03	0.00002	0.1
Cr	-	0.1	0.003	0.2
Co	-	0.1	0.001	0.2
Cu	-	1	0.001	2
Pb	-	0.4	0.0002	1
Hg	-	0.01	0.00001	0.02
Ni	-	1	0.002	2
Zn	-	75	0.005	150

Note: Best estimates are calculated as the arithmetic mean of the high and low estimates.

Source: COWI (2000) A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste. Final Report to DG Environment, the European Commission, August 2000.

**Table 4.1 Emission Factors for Specific Organic Compounds in Leachate from MSW Landfills (all factors in g/tonne MSW landfilled).**

g/tonne	Best estimate*		Low estimate	High estimate
	L1	L2	L2	L2
<b>Aromatic Hydrocarbons:</b>				
Benzene	-	122	0.2	245
Toluene	-	923	0.2	1,845
Xylene	-	263	1	525
Ethylbenzene	-	96	0.2	192
Trimethylbenzene	-	19	1	38
Naphthalene	-	20	0.02	39
Diethylphthalate	-	50	2	99
Di-n-butylphthalate	-	2	1	2
Butyl-benzyl-phthalate	-	1	1	1
<b>Chloromatic Hydrocarbons:</b>				
Chlorobenzene	-	8	0.02	17
1,2-Dichlorobenzene	-	2	0.02	5
1,4-Dichlorobenzene	-	1	0.02	2
1,1-Dichloroethane	-	3	0.1	7
1,2-Dichloroethane	-	1	0.9	1
1,1,1-Trichloroethane	-	286	0.02	572
trans-1,2-Dichloroethylene	-	7	0.2	13
cis-1,2-Dichloroethylene	-	35	0.2	71
Trichloroethylene	-	56	0.1	113
Tetrachloroethylene	-	19	0.02	38
Methyl chloride	-	5	0.2	10
Chloroform	-	5	0.2	11
Carbon tetrachloride	-	1	1	1
<b>Phenols:</b>				
Phenol	-	90	0.2	180
Ethyl phenols	-	45	45	45
Creosols	-	158	0.2	315
<b>Pesticides:</b>				
MCCP	-	7	0.3	14
2,4-D	-	0.5	0.2	1
<b>Other:</b>				
Acetone	-	330	1	660
Tetrahydrofuran	-	33	1	65
Methylethylketone	-	503	17	990
Tri-n-butylphosphate	-	27	0.2	54
Triethylphosphate	-	2	2	2
Camphor	-	Identified		
Fenchone	-	4	3	5

*Note: Best estimates are calculated as the arithmetic mean of the high and low estimates.*

*Source: COWI (2000) A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste. Final Report to DG Environment, the European Commission, August 2000.*

## A.8.4 Odours

Landfill odours often prompt complaints from community members. People may also have concerns about health effects associated with these odours and other emissions coming from the landfill. At low-level concentrations — typically associated with landfill gas — it is unclear whether it is the constituent itself or its odours that trigger a response. Typically, these effects fade when the odour can no longer be detected. **Error! Reference source not found.** below describes the biology behind detecting odours.

Landfill gas odours are produced by bacterial or chemical processes and can emanate from either active or closed landfills. These odours can migrate to the surrounding community. Potential sources of landfill odours include sulphides, ammonia, and certain NMOCs, if present at concentrations that are high enough. Landfill odours may also be produced by the disposal of certain types of wastes, such as manures and fermented grains.

- **Sulphides.** Hydrogen sulphide, dimethyl sulphide, and mercaptans are the three most common sulphides responsible for landfill odours. These gases produce a very strong rotten-egg smell — even at very low concentrations. Of these three sulphides, hydrogen sulphide is emitted from landfills at the highest rates and concentrations. Humans are extremely sensitive to hydrogen sulphide odours and can smell such odours at concentrations as low as 0.5 to 1 part per billion (ppb). At levels approaching 50 ppb, people can find the odour offensive.
- **Ammonia.** Ammonia is another odorous landfill gas that is produced by the decomposition of organic matter in the landfill. Ammonia is common in the environment and an important compound for maintaining plant and animal life. People are exposed daily to low levels of ammonia in the environment from the natural breakdown of manure and dead plants and animals. Because ammonia is commonly used as a household cleaner, most people are familiar with its distinct smell. Humans tend to be less sensitive to the odour of ammonia than they are to sulphide odours. The odour threshold for ammonia may be between 28,000 and 50,000 ppb. Landfill gas has been reported to contain between 1,000,000 and 10,000,000 ppb of ammonia, or 0.1% to 1% ammonia by volume. Concentrations in ambient air at or near the landfill site are expected to be much lower.
- **NMOCs.** Some NMOCs, such as vinyl chloride and hydrocarbons, may also cause odours. In general, however, NMOCs are emitted at very low (trace) concentrations and are unlikely to pose a severe odour problem.

**Error! Reference source not found.** lists some of the common landfill gas components and associated odour thresholds.

**Table 48: Common Landfill Gas Components and Their Odour Thresholds**

Component	Odour Description	Odour Threshold (parts per billion)
Hydrogen Sulphide	Strong rotten egg smell	0.5 to 1
Ammonia	Pungent acidic or suffocating odour	1,000 to 5,000
Benzene	Paint-thinner-like odour	840
Dichloroethylene	Sweet, ether-like, slightly acrid odour	85
Dichloromethane	Sweet, chloroform-like odour	205,000 to 307,000
Ethylbenzene	Aromatic odour like benzene	90 to 600
Toluene	Aromatic odour like benzene	10,000 to 15,000
Trichloroethylene	Sweet, chloroform-like odour	21,400
Tetrachloroethylene	Sweet, ether-or chloroform-like odour	50,000
Vinyl Chloride	Faintly sweet odour	10,000 to 20,000

Many people may find the odours emitted from a landfill offensive or unpleasant. In reaction to the odour, some people may experience nausea or headaches. Although such responses are undesirable, medical attention is usually not required. Often, symptoms such as headaches and nausea fade when the odour goes away. However, the effects on day-to-day life can be more lasting.

In the US, families living close to a landfill in Connecticut described frequent odour events as overwhelmingly disruptive. One family reported being awakened during pre-dawn hours by a flood of nauseating air that persisted for 2 or more hours. The loss of sleep and the frustration from the frequent odour events greatly added to the level of stress in the family's life. Although landfill odours may not be associated with long-term adverse health effects or illness for most people, the added disruption and stress of day-to-day activities can greatly impact quality of life.

The impact of landfill gas odours on sensitive populations such as people with pre-existing respiratory illnesses is not well documented or understood. A study conducted on Staten Island, New York showed an increase in self-reported wheezing among asthmatics living near a landfill on days when they reported odours. The study suggests that odours in and of themselves may trigger respiratory effects among asthmatics. This preliminary conclusion may be confounded by other environmental triggers for respiratory response in asthmatics, such as dust mites, animal dander, tobacco smoke, and outdoor air pollution.

### **A.8.5 Other Reported Health Effects**

In addition, there has been some discussion about the health effects of landfill. There are still uncertainties in the available knowledge as to how landfills affect human health. Further work is being undertaken in this context.

## A.8.6 Disamenity

The reader is referred to the main report for a discussion of this issue.

## A.9.0 INCINERATION

In this study, we have used our own proprietary model, for which the incineration module is described below.

### A.9.1 Air Emissions

The model allows for choice of abatement equipment. For particulate removal, the choices are:

- None;
- Electrostatic precipitator; or
- Bag filter.

For treatment of acid gases, the model allows for a choice to be made between:

- None
- Dry scrubbing
- Semi-dry scrubbing; and
- Wet scrubbing.

For NO<sub>x</sub> removal, the model allows for a choice of:

- None;
- SNCR; and
- SCR.

Finally, for dioxin removal (as well as SCR), the model allows for a choice between:

- None; and
- Activated carbon.

In practice, most UK incinerators are based around use of:

- Bag filters;
- Semi-dry scrubbing;
- Flue gas recirculation with / without SNCR; and
- Activated carbon.

This model, using SNCR (to ensure compliance with the Incineration Directive) is used as the 'baseline' UK performer.

Calculations of air emissions are discussed below.

### A.9.1.1 Carbon-related Air Emissions

Carbon-related emissions are constant across all configurations and are calculated as follows:

- Emissions of CO<sub>2</sub>, CO and CH<sub>4</sub> are taken from Erichsen and Hauschild.<sup>169</sup> These are calculated based upon the carbon input of waste.
- Emissions of Chloroethene and Dichloroethane are taken from work by Enviros for the NSCA.<sup>170</sup> These are fixed factors expressed in mg/tonne waste.

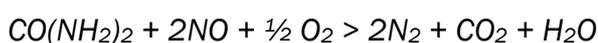
### A.9.1.2 Nitrogenous emissions

Nitrogenous emissions are taken from Erichsen and Hauschild<sup>171</sup> and are specific to the nature of flue gas cleaning chosen. Emissions of NO, NO<sub>2</sub>, N<sub>2</sub>O and N<sub>2</sub> are each calculated assuming process- and feedstock-related factors. In other words, for each, the emissions are calculated partly on a 'per tonne of waste' basis, and partly on the basis of the N content of the waste.

Emissions of ammonia are taken to be dependent on the nature of flue gas cleaning used. Selective Non-catalytic Reduction (SNCR) removes NO, through direct chemical reaction with a reagent. Compounds used in this process include ammonia and urea.

A problem with this process is called "ammonia slip," which is a measure of the amount of ammonia that fails to react. Since it does not react with lime in the slurry reactor and is a volatile chemical, ammonia will pass through the flue gas purification system and exit as a stack emission. Ammonia also has the potential to combine with hydrogen chloride (HCl) (generated from combustion of chlorinated compounds) to form an ammonium chloride plume as it exits the stack.

Ammonia slip tends to be more difficult to minimize when ammonia itself is used as the reagent. Therefore, urea has become the reagent of choice. It removes NO, through the following reaction:



Urea is most effective at high temperature, and is therefore injected directly above the combustion zone. Urea typically has a removal efficiency of between 50 and 75%, but in some cases has removed up to 90% of NO<sub>x</sub>. Some ammonia slip has been observed, but **Error! Reference source not found.** illustrates how removal efficiency increases while ammonia slip decreases as temperature rises.

At typical operating temperatures, ammonia slip is of the order 5% or less.

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<sup>169</sup> Hanne L. Erichsen and Michael Hauschild (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.

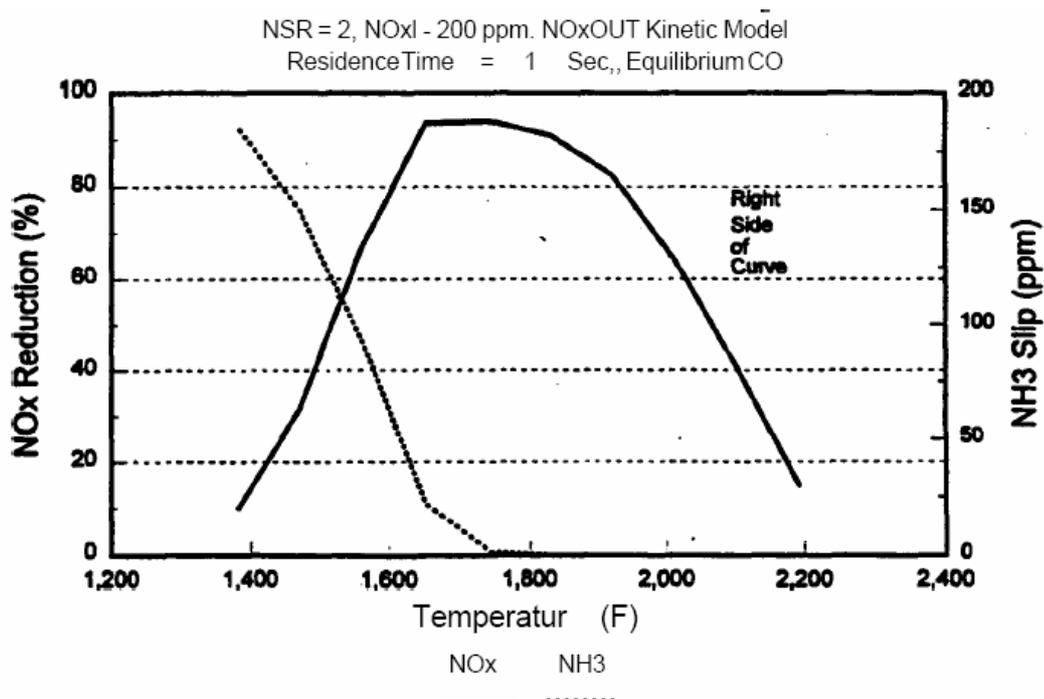
<sup>170</sup> Enviros Aspinwalls (2002) *Comparison of Emissions from Waste Management Options*, Enviros Aspinwalls for NSCA June 2002.

<sup>171</sup> Hanne L. Erichsen and Michael Hauschild (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.

Ammonia emissions are estimated in the following way:

- The stoichiometric requirements for urea (SNCR) and ammonia (SCR) are calculated on the basis of NO and NO<sub>2</sub> in raw gas and are taken from Erichsen and Hauschild and the BREF note;<sup>172</sup>
- An additional 2% of the N in the compounds used is assumed to be emitted to the atmosphere as ammonia where SNCR is used. The figure of 2% is an estimate only, though this gives figures in line with the figures in, for example, the study by VITO.<sup>173</sup>

Figure 22: Relationship between NO<sub>x</sub> Reduction Achieved and Operating Temperature, with Resultant Ammonia Slip



### A.9.1.3 Other Acid Gases

Emissions of HCl, HF and SO<sub>2</sub> are calculated using transfer factors based on the content of Cl, F and S of the waste, and these vary with the nature of flue gas cleaning. Again, these are taken from Erichsen and Hauschild.<sup>174</sup>

<sup>172</sup> Hanne L. Erichsen and Michael Hauschild (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; European Commission (2005) *Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration*, Final Draft, May 2005.

<sup>173</sup> VITO (2000) *Vergelijking van Verwerkingsscenario's voor Restfractie van HHA en Niet-specifiek Categorie II Bedrijfsafval*, Final Report.

<sup>174</sup> Hanne L. Erichsen and Michael Hauschild (2000) *Technical Data for Waste Incineration - Background for Modeling of Product Specific Emissions in a Life-cycle Assessment Context*, Elaborated

#### A.9.1.4 Dioxins and PCBs

For chlorinated dioxins, emissions are deemed to be process related (calculated per tonne of waste) but are specific to the flue gas cleaning configuration used. These are taken from Erichsen and Hauschild.<sup>175</sup>

For PCBs, the emissions are deemed to be process related (calculated per tonne of waste) but are not specific to the flue gas cleaning configuration used. These are taken from the Enviro report for the NSCA.

#### A.9.1.5 Particulate Matter

The emissions of particulate matter are based upon data from the Enviro report for the NSCA.<sup>176</sup> These figures are graded somewhat to reflect the different flue gas cleaning systems. The speciation across particle size group is based on an AEA report cited in evidence provided in the context of the Belvedere Inquiry. Particulate matter is, therefore, split between PM<2.5 microns, and the PM which is >2.5 microns but <10 microns.<sup>177</sup>

#### A.9.1.6 Metals

For metals, the transfer into the flue gas was modelled using transfer factors from work by Hellweg.<sup>178</sup> Hellweg used transfer factors for incinerators with:

- Electrostatic precipitator only (no flue gas purification);<sup>179</sup>
- Average Swiss gas purification system;<sup>180</sup> and
- Modern gas purification system.<sup>181</sup>

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as part of the EUREKA project EUROENVIRON 1296: LCAGAPS, sponsored by the Danish Agency for Industry and Trade, April 2000.

<sup>175</sup> Hanne L. Erichsen and Michael Hauschild (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.

<sup>176</sup> Enviro Aspinwall (2002) *Comparison of Emissions from Waste Management Options*, Enviro Aspinwall for NSCA June 2002.

<sup>177</sup> In Nick Gamble (2003) *Electricity Act 1989: Section 36, Riverside Resource Recovery Facility, Belvedere*, Proof of Evidence of Nick Gamble, 04 June 2003, S0641-0056-0009NSG Evidence.

<sup>178</sup> Stefanie Hellweg (2000) *Time- and Site-dependent Life-cycle Assessment of Thermal Waste Treatment Processes*, Diss. ETH No.13999, Zurich.

<sup>179</sup> P. Zimmermann, G. Doka, F. Huber, A. Labhardt, and M. Menard (1996) *Ökoinventare von Entsorgungsprozessen, Grundlagen zur Integration der Entsorgung in Ökobilanzen*, ESURreihe, 1/96, Institut für Energietechnik, ETH Zürich, Zürich, 1996.

<sup>180</sup> P. Zimmermann, G. Doka, F. Huber, A. Labhardt, and M. Menard (1996) *Ökoinventare von Entsorgungsprozessen, Grundlagen zur Integration der Entsorgung in Ökobilanzen*, ESURreihe, 1/96, Institut für Energietechnik, ETH Zürich, Zürich, 1996.

<sup>181</sup> P. Zimmermann, G. Doka, F. Huber, A. Labhardt, and M. Menard (1996) *Ökoinventare von Entsorgungsprozessen, Grundlagen zur Integration der Entsorgung in Ökobilanzen*, ESURreihe, 1/96, Institut für Energietechnik, ETH Zürich, Zürich, 1996; L. Morf, E. Ritter, and P.H. Brunner, *Güter- und Stoffbilanz der MVA Wels*, Technische Universität Wien, Wien, 1997; H. Belevi, *Environmental Engineering of Municipal Solid Waste Incineration*, v/d/f, Dübendorf, 1998.

These transfer factors have been applied in the model to demonstrate the transfer of the elements concerned to the atmosphere based on their content in the feedstock waste.

## A.9.2 Transfer of Substances to Bottom Ash (slag) and to Air pollution Control (APC) Residues

Figures for total organic carbon, carbonated carbon and nitrogen are taken from Erichsen and Hauschild.<sup>182</sup> These are related to input C and input N as appropriate. These are assumed not to vary with the nature of flue gas treatment. A figure for dioxins is taken from Erichsen and Hauschild.<sup>183</sup> The review in the original study shows that measurements give wide variations in dioxins in solid residues. Hence, there is considerable uncertainty in these figures. The figure for bottom ash is not deemed to vary with flue gas treatment, but the figure for fly ash clearly does.

The transfer of Chlorine, Fluorine and Sulphur into the slag and into APC residues is, however, assumed to vary with flue gas treatment. For consistency (in respect of mass flows), we have used figures from Erichsen and Hauschild.<sup>184</sup> These are related to the amount of Cl, F and S in the input waste.

Again, in order to maintain a consistent mass balance in the analysis, the transfer of metals into bottom ash and fly ash is taken from Hellweg's study.<sup>185</sup>

## A.9.3 Water Pollution

The degree to which an incinerator is likely to lead to discharges to water is related to the nature of flue gas cleaning used. Water pollution can arise from the use of wet scrubbing for the removal of acid gases. The transfer of different elements into waste water and sludges is taken from Erichsen and Hauschild.<sup>186</sup> These factors are only relevant where wet scrubbing is used. The transfer factors relate to the quantity of each material in the input material.

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<sup>182</sup> Hanne L. Erichsen and Michael Hauschild (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.

<sup>183</sup> Hanne L. Erichsen and Michael Hauschild (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.

<sup>184</sup> Hanne L. Erichsen and Michael Hauschild (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.

<sup>185</sup> Stefanie Hellweg (2000) *Time- and Site-dependent Life-cycle Assessment of Thermal Waste Treatment Processes*, Diss. ETH No.13999, Zurich.

<sup>186</sup> Hanne L. Erichsen and Michael Hauschild (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.

## A.9.4 Input Materials

The input materials to an incinerator depend critically upon the flue gas cleaning system used. The inputs for all but the SNCR DeNO<sub>x</sub> system are adapted from Erichsen and Hauschild.<sup>187</sup> The figures in that report give outturns which are well below what is observed. The figures for urea for SNCR DeNO<sub>x</sub> are calculated using the adapted figures and taking into account the stoichiometric relations for the reactions concerned.<sup>188</sup>

## A.9.5 Input Energy

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);
- Changes in the LHV of input waste (necessitating use of fuel to maintain minimum combustion temperatures).

The Draft BREF note for Incineration gives figures of:<sup>189</sup>

- **Electricity use:**  
62 kWh per tonne – 257 kWh per tonne, average 142kWh per tonne
- **Heat demand:**  
72 GJ thermal energy per tonne – 3,366 GJ thermal energy per tonne, average 433 GJ thermal energy per tonne.

These are far higher than figures suggested in, for example, reports by Erichsen and Hauschild,<sup>190</sup> at 46 kWh electricity per tonne, though this figure reflects only the

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<sup>187</sup> Hanne L. Erichsen and Michael Hauschild (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.

<sup>188</sup> VITO (2000) *Vergelijking van Verwerkingsscenario's voor Restfractie van HHA en Niet-specifiek Categorie II Bedrijfsafval*, Final Report.

<sup>189</sup> European Commission (2005) *Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration*, Final Draft, May 2005.

<sup>190</sup> Hanne L. Erichsen and Michael Hauschild (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.

operation of gas cleaning equipment. VITO give the following consumption of energy for processes with and without SCR:<sup>191</sup>

- Natural gas: 7.2m<sup>3</sup> per tonne
- Oil 4kg per tonne
- Electricity Use (per tonne): 80kWh with SNCR, 85 kWh with SCR
- Electricity production (per tonne): 480 kWh with SNCR, 475 kWh with SCR

The reason for lower generation using SCR is that the metal oxide bed (the catalyst) cannot be located at the point in the flue gas stream where the temperature is optimal, since this occurs prior to the removal of many contaminants that can poison the bed. Therefore, the SCR system must be placed just prior to the stack, and requires the 200 ° F flue gas to be reheated at additional electrical cost.<sup>192</sup>

In our view, the lower figures for net electrical efficiency being found in other parts of Europe reflect the fact that flue gas abatement standards have been tighter for longer. More commonly quoted UK figures – such as those quoted by Fichtner – appear to envisage high generation efficiencies with little allowance for additional energy use in the plant itself (to reflect the energy demand of more advanced flue gas treatments). In this study, we have used the estimates from the VITO study.

## A.9.6 Energy Efficiency

The model allows the user to specify the energy efficiency of the process. The model allows for both heat and electricity to be recovered. In terms of understanding the equivalent fuel needed to generate the same amount of energy, we follow the Draft BREF note for Incineration in using 91% for heat generation, and we use a calculated figure of 40% for electricity generation.

It should be noted that net efficiencies (i.e. net of own usage) are frequently much lower than the levels quoted by consultants. For example, Fictner quotes a ‘realistic range’ for net electrical efficiency of 19-27%. Yet the Draft BREF note gives no case where the net export of electricity exceeds 18%, and for new facilities, a typical figure of around 400kWh per tonne (around 18% net efficiency) seems appropriate.<sup>193</sup> Taking into account typical usage of energy by the plant itself (see above), gross efficiencies of around 22% seem realistic.<sup>194</sup>

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<sup>191</sup> VITO (2000) *Vergelijking van Verwerkingsscenario's voor Restfractie van HHA en Niet-specifiek Categorie II Bedrijfsafval*, Final Report.

<sup>192</sup> 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.

<sup>193</sup> European Commission (2005) *Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration*, Final Draft, May 2005.

<sup>194</sup> 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%.

## A.9.7 Recovered Materials

The model allows the user to define the rate of recovery of steel and aluminium from the input material (as a percentage). Figures for steel recovery in the literature vary around 67%, this is the default figure used here. This is assumed to be recovered from the slag.

## A.9.8 Effects of Landfilling APC Residues (pollution of land and groundwater)

The modelling actually calculates the emissions to water from landfilling air pollution control residues. The landfilling of fly ash and bottom ash has been modelled by Björklund, Hellweg and GUA et al.<sup>195</sup> Both Hellweg and GUA et al use time-dependent transfer factors. For Hellweg, these are related to three scenarios regarding leachate generation:

1. Measurements at landfill sites (model option 1)
2. Leaching tests (model option 2)
3. Only on the composition of the landfill regarding all ingredients as future emissions (model option 3).

This is not so dissimilar to the approach of GUA et al (which is extended to gaseous emissions from landfills also).

In the modelling, for fly ash residues, we have used Hellweg's figures (for short and long-term emissions to water). Bottom ash is assumed to be recycled into construction materials.

### A.9.8.1 External Costs of Landfilling APC Residues

The external costs associated with the landfilling of APC residues are excluded in the 'Defra' scenario. In the 'low other' scenario, we have used the value from a recent Dutch study.<sup>196</sup> In the 'high other' scenario, the value is taken from a recent paper by Dijkgraaf and Vollebergh.<sup>197</sup>

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<sup>195</sup> A. Björklund, A. (1998). *Environmental systems analysis waste management*. Licentiate Thesis. Royal Institute of Technology, Stockholm. S. Hellweg (2000) *Time- and Site-Dependent Life-Cycle Assessment of Thermal Waste Treatment Processes*, Dissertation submitted to the Swiss Federal Institute Of Technology; GUA, AWS and IFIP (2000) *Bewertung Abfallwirtschaftlicher Maßnahmen mit dem Ziel der Nachsorgefreien Deponie (BEWEND)*, Final Report, Vienna, November 2000. An English summary is available – G. Doberl, R. Huber, P. Brunner, M. Eder, R. Pierrard, W. Schonback, W. Fruhwirth, H. Hutterer (2000) *Long-term Assessment of Waste Management Options – a New, Integrated and Goal-Oriented Approach*.

<sup>196</sup> Bartelings, H., P. van Beukering, O. Kuik, V. Linderhof, F. Oosterhuis, L. Brander and A. Wagtenonk (2005) *Effectiveness of Landfill Taxation*, R-05/05, Report Commissioned by Ministerie von VROM, November 24, 2005.

<sup>197</sup> E. Dijkgraaf and H. R. J. Vollebergh (2004), Burn or Bury? A Social Cost Comparison of Final Waste Disposal Methods, *Ecological Economics*, 50, 233-247

## A.10.0 MODELLING OF EXTERNAL COSTS

The modelling of external costs is based upon the application of unit damage costs to the calculated emissions associated with the different processes, as discussed in the preceding Appendices.

### A.10.1 Air Emissions

The air emissions quantified are set out in **Error! Reference source not found.** below. They are set out alongside the sources for the so-called Low, Medium and High estimates of the external costs. The low end values are effectively replicating the unit damage costs used in the estimates of external costs and benefits made by HM Customs & Excise.<sup>198</sup> For reasons explained in the main text, these are likely to understate the externalities from waste management, not least since they cover a small range of pollutants, and because the unit damage costs are lower than used in the majority of other studies.

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<sup>198</sup> HM Customs & Excise (2004) *Combining the Government's Two Health and Environment Studies to Calculate Estimates for the External Costs of Landfill and Incineration*, December 2004; these were based upon damage costs estimated in *Enviros* and EFTEC (2004) *Valuation Of The External Costs And Benefits To Health And Environment Of Waste Management Options*, Final Report for Defra, December 2004.

Table 42: Low and High Damage Costs Used, and Sources (€/tonne)

	Low	Medium	High	Source and Notes
CO <sub>2</sub>	£21.97	£21.97	£42.47	Clarkson & Deyes – Low and Medium are the same since the low figures reflect Defra’s base case (the mid-estimate based upon Clarkson and Deyes)
CH <sub>4</sub>	£365.32	£365.32	£706.28	Clarkson & Deyes – Low and Medium are the same since the low figures reflect Defra’s base case (the mid-estimate based upon Clarkson and Deyes)
N <sub>2</sub> O	£6,589.62	£6,589.62	£12,739.93	Clarkson & Deyes – Low and Medium are the same since the low figures reflect Defra’s base case (the mid-estimate based upon Clarkson and Deyes)
NH <sub>3</sub>		£11,724.14	£34,482.76	Defra gives no figures and CAFÉ Medium and High
VOCs	£684.38	£758.62	£2,206.90	Lower end from Enviros and EFTEC, Medium and High are lower and higher end results reported for UK under CAFÉ (AEAT)
PM2.5 Efw	£40,388.75	£40,388.75	£75,862.07	Lower end from Enviros and EFTEC, Medium is also lower end from Enviros and EFTEC (since this is marginally higher than the lower end from CAFÉ), High is higher end result reported for UK under CAFÉ (AEAT)
PM2.5 landfill	£1,054.87	£1,054.87	£1,981.36	Lower end from Enviros and EFTEC, Medium is lower end from Enviros and EFTEC (since, when lower end CAFÉ is factored down by the ratio of PM2.5 for Efw:PM2.5 for Landfill calculated from Enviros and EFTEC, this is marginally higher than the lower end from CAFÉ), High is higher end result reported for UK under CAFÉ (AEAT), though factored down by the ratio of PM2.5 for Efw:PM2.5 for Landfill calculated from Enviros and EFTEC
SO <sub>x</sub>	£3,026.71	£4,551.72	£13,103.45	Lower end from Enviros and EFTEC, Medium and High are lower and higher end results reported for UK under CAFÉ (AEAT)
NO <sub>x</sub>	£1,005.47	£2,689.66	£6,896.55	Lower end from Enviros and EFTEC, Medium and High are lower and higher end results reported for UK under CAFÉ (AEAT)
CO		£2.00	£7.00	Lower and Higher end from COWI
Cd		£13,793.10	£65,517.24	Lower and Higher end from COWI
Cr		£91,724.14	£660,689.66	Lower and Higher end from COWI
Cu		£42.00	£42.00	High is from EFTEC, Medium is 50% of this value
Hg		£10,993,944.00	£21,987,888.00	Lower and Higher end from COWI
Ni		£2,068.97	£13,793.10	Lower and Higher end from COWI
Pb		£610,000.00	£1,220,000.00	High is from CSERGE and EFTEC. Medium is 50% of this value
Dioxin		£1,600,000,000.00	£13,000,000,000.00	Lower and Higher end from COWI
As		£111,724.14	£805,517.24	Lower and Higher end from COWI

Sources: COWI (2000) *A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste. Final Report to DG Environment, the European Commission, August 2000*; AEAT Environment (2005) *Damages per tonne Emission of PM<sub>2.5</sub>, NH<sub>3</sub>, SO<sub>2</sub>, NO<sub>x</sub> and VOCs from Each EU25 Member State (excluding Cyprus) and Surrounding Seas, Report to DG Environment of the European Commission, March 2005*; R. Clarkson and K. Deyes (2002) *Estimating the Social Costs of Carbon, Government Economic Service Working Paper, January 2002*; Enviro and EFTEC (2004) *Valuation Of The External Costs And Benefits To Health And Environment Of Waste Management Options Final Report for Defra, December 2004*; CSERGE and EFTEC (1998), *Life Cycle Research Programme for Waste Management: Damage Cost Estimation for Impact Assessment, Report to the Environment Agency*.

## A.11.0 MODELLING CHARACTERISTICS OF INPUT WASTE

The model seeks to understand the effects of different treatments on the waste which is sent for treatment. It does this by:

1. Estimating the composition of all waste in the area being modelled;
2. Subtracting from this the materials deemed likely to be recycled / composted (and, where possible, a 'steady state' is modelled, though it is recognised that composition will change over time);
3. Estimating the resultant composition of residual waste; and
4. Deriving an ultimate analysis of the total waste by multiplying the composition by the characteristics of the waste in terms of moisture, calorific value, content of minerals, and metals content.

These steps result in an ultimate analysis for the residual waste.

The sources for the ultimate analysis are shown in Table 43. It must be appreciated that although we have sought to obtain data from reputable sources, the categories into which waste was classified do not all readily lend themselves to ready conversion into ultimate components (and there are discrepancies across the different sources). Hence, we have made some assumptions regarding some of the waste categories:

- Soil – taken to be composed of 50% garden waste and 50% glass (approximating a sand / organic mix).
- Sanitary – taken to be composed of 25% nappy, 25% ferrous metal and 50% plastic.
- Hazardous (mostly cat litter) – taken to be 80% glass and 20% kitchen waste.
- Textiles – for the biological characteristics, 25% garden and 25% paper (remaining 50% assumed non biological);
- Miscellaneous (hoover dust) – taken to be 50% textiles, 50% fines.

These approximations no doubt compromise the quality of the analysis, but data on such materials does not exist in, for example, the national household waste analysis programme (hence, some of the datasets for organic materials come from overseas).

Table 43: Data Sources for Ultimate Analysis and Composition of Materials in Residual Waste

Material	Source of Data	Year	% in Residual Waste
Paper	NHWAP	1992/3	17.05
Cardboard	NHWAP except moisture content - Sundqvist et al. 2000, Appendix 2	1993 / 2000	4.67
Plastic film	NHWAP	1993	0.56
PET plastic	RVF 1996, p 19-20 and appendix 2	1996	6.51
HDPE plastic	Björklund 1998, Appendix G, p G-2	1998	0.86
PVC plastic	Zevenhoven et al. 1995, p 8 to 9	1995	1.10
Other dense plastic	SAH.Moorman et al: Emissies uit bijstoken, verbranden en vergassen van niet-gevaarlijke afvalstromen in vergelijking tot BLA en AVI. Hoofdrapport, Delft(Netherlands),Centrum voor Energiebesparing en schone technologie,CE-00.5713.01, 69 p(2000)	2000	0.02
Textiles	NHWAP	1993	4.86
Hazardous [cat litter] (assumes 80% glass, 80% kitchen waste).	NHWAP & (Sundqvist et al. 2000, Appendix 2).	1993 & 2000	3.37
Kitchen waste	(Sundqvist et al. 2000, Appendix 2).	2000	2.87
Garden waste	Phyllis (Energy research centre of the Netherlands, database for biomass and waste), average composition of group of materials - Flowers / garden plants	2004	36.05
Soil			3.69
Inert	Magnus Dalemo, "The modelling of an anaerobic digestion plant and a sewage plant in the orware simulation model", Swedish university of agricultural sciences, report 213, issn 0283-0086, 1997	1996	0.52
Glass	NHWAP	1993	0.72
Ferrous metal	NHWAP	1993	6.91
Non-ferrous metal	NHWAP	1993	2.36
Sanitary			0.89
Nappies	NHWAP	1993	0.20
WEEE (assumes 30% flame retardent plastic, 20% glass, 50% ferrous metal)	Phyllis - European brominated flame retardant industry panel (EBFRIP) (ID-number 2179) - average - European brominated flame retardant industry panel (EBFRIP) shredded and sieved to particle sizes < 1 cm	2004	4.52
Natural timber	Phyllis (Energy research centre of the Netherlands, database for biomass and waste), average composition of group of materials - untreated wood	2004	0.56
Treated wood	Phyllis (Energy research centre of the Netherlands, database for biomass and waste), average composition of group of materials - treated wood	2004	0.15
Miscellaneous [hoover dust], (assumes 50% textile,, 50% fines).	NHWAP	1993	1.20